CRANFIELD UNIVERSITY

Thomas Denis Collin

# Towards sustainable fats, oils and greases (FOG) management: From waste to resource

School of Water, Energy and Environment
STREAM IDC

Engineering Doctorate
Academic Year: 2015 - 2019

Supervisor: Prof. Bruce Jefferson
Associate Supervisor: Dr. Jitka MacAdam
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This thesis is submitted in partial fulfilment of the requirements for the degree of Engineering Doctorate

# Abstract

Fats, oils and greases (FOG) are by-products of cooking and food preparation originating from households, food service establishments (FSEs) and industrial food facilities. Under UK legislation, industries are the only sources of FOG monitored by water authorities under trade effluent consents. In addition, whilst all FSE kitchens must be fitted with an “effective mean of grease removal”, experience within the water sector has demonstrated that very few premises were managing their effluents to minimise FOG emissions. Critically, large volumes of FOG are entering drainage systems resulting in impacts both in the sewers (formation of fatbergs) and downstream at the treatment works (reduced treatment efficiency), and in turn contributing to high operational costs for water utilities. With changing food habits and projected population growth, FOG-related problems will only become an increasingly worrying operational and financial burden for the water industry. Yet, FOG can be a valuable resource for energy recovery with the potential to offset operational costs and improve the overall sustainability of wastewater treatment.

This thesis establishes an evidence base by assessing the potential of alternative FOG management options in order to provide guidance to water utilities for improving current practices. A comprehensive study identified the contribution of current kitchen practices to FOG emissions from domestic and commercial sources, suggesting the need for educational campaigns to raise awareness on the problem. Production rates and quality of FOG from different sources were benchmarked clarifying variations amongst these wastes. The potential of FOG for energy recovery, via biogas generation from anaerobic digestion, was assessed through laboratory-scale studies. The occurrence of FOG at the treatment works was investigated, and the performance of enhanced treatment for its removal was further studied. Finally, data produced during this project was used to develop a business case for the implementation of more sustainable approaches.

**Keywords:** Anaerobic digestion; fatberg; sewer deposits; sewage treatment works (STW); dissolved air flotation (DAF); grease separator

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# List of abbreviations

| | |
|---|---|
| ANOVA | Analysis of variance |
| AMPTS | Automatic methane potential test system |
| $BOD_5$ | Biochemical oxygen demand |
| CapEx | Capital expenditure |
| CE | Chemically-enhanced |
| CHP | Combined heat and power |
| CST | Capillary suction time |
| DAF | Dissolved air flotation |
| DS | Dry solids |
| DWF | Dry weather flow |
| EHO | Environmental health officer |
| EPS | Extracellular polymeric substance |
| FAME | Fatty acid methyl ester |
| FFA | Free fatty acid |
| FHRS | Food hygiene rating scheme |
| FOG | Fats, oils and greases |
| FSE | Food service establishment |
| GTW | Grease trap waste |
| GRU | Grease removal unit |
| HEM | Hexane extractable material |
| HHV | Higher heating value |

| | |
|---|---|
| $IC_{50}$ | Half maximal inhibitory concentration |
| ICP-MS | Inductively coupled plasma mass spectrometry |
| LCFA | Long chain fatty acid |
| LHV | Lower heating value |
| OpEx | Operational expenditure |
| PST | Primary sedimentation tank |
| SDAC | Sewer drainage area catchment |
| SOR | Surface overflow rate |
| SPS | Sewage pumping station |
| SS | Suspended solids |
| STP | Standard temperature and pressure |
| STW | Sewage treatment works |
| tCOD | Total chemical oxygen demand |
| TFA | Total fatty acid |
| TH | Thermal hydrolysis |
| TWUL | Thames Water Utilities Ltd. |
| UCO | Used cooking oil |
| VFA | Volatile fatty acid |
| VS | Volatile solids |

# 1 Introduction

## 1.1 Background

Fats, oils and greases (FOG) are by-products of food preparation entering sewer systems through a variety of sources including households, food service establishments (FSEs) and industries. Uncontrolled discharges have attracted increased attention from both water infrastructure operators concerned about the obstruction of sewer lines and the general public as high profile sewer blockages (fatbergs) appear in media headlines (Engelhaupt, 2017; Moss, 2018). FOG impacts both sewerage systems, reducing their effective capacity and leading in some cases to sewer flooding, and sewage treatment works (STWs), hampering treatment efficiency (He et al., 2017; Wallace et al., 2017). As a consequence, it is estimated that UK water utilities spend between £15M and £50M per annum reactively managing FOG discharges (Williams et al., 2012). Ultimately, with changing food habits (Paddock et al., 2017) and projected population growth (Office for National Statistics, 2017a), FOG-related problems will only become an increasing burden in the future for the water industry. However, FOG represents an energy source and their use as substrates for biogas production through anaerobic digestion (Davidsson et al., 2008; Kabouris et al., 2009a) or biodiesel production (Canakci, 2007) has been documented in literature. Accordingly, FOG can also be viewed as a value proposition if appropriately managed. Yet, due to its long-chain fatty acids (LCFAs)-rich composition, further care is required when treated in anaerobic digesters for instance as it can hamper process efficiency (Alves et al., 2009).

The management of FOG can occur at all parts of the catchment, from source to STWs, through a wide range of approaches but this variety of techniques often leads to a paucity of clarity as to the overall best management strategy. The following section aims to describe current practices within the industry for the management of FOG.

## 1.2 An operational and financial burden

The accumulation of FOG in sewers has attracted increased attention from both water infrastructure operators concerned about the obstruction of sewers and the general public as high profile sewer blockages appear in media headlines (Engelhaupt, 2017; Moss, 2018). Once allowed to solidify and/or deposit in sewer lines, such discharges tend to form large assemblages (often called fatbergs) thereby reducing a sewer’s effective capacity and leading, in some cases, to sewer flooding (He et al., 2017). These deposits are generally removed from the sewers either reactively or pro-actively (planned sewer cleaning in known hotspot areas). This is achieved either through hand-removal in large diameter sewers or using high-pressure jetting equipment (e.g. wet well of sewage pumping stations). The latter generally breaks down deposits into smaller pieces either remaining in the networks or extracted alongside large volumes of sewage by vacuum tankers and further disposed at STWs. As of today, there has been limited focus on energy recovery from these materials. It was posited that calcium present in FOG deposits could provide a positive effect on its degradation in anaerobic digesters (Hatamoto et al., 2007). Further research on the potential of these wastes for energy recovery would encourage their utilisation, and thus drive sewer cleaning beyond networks protection.

FOG not only causes pipe blockages within the sewerage system but also disrupts settlement and clarification processes at STWs, thus hindering treatment efficiency (Wallace et al., 2017). In addition, FOG exerts an extra load of organic matter onto the secondary aerobic treatment stage thereby increasing the overall aeration demand. Whilst the LCFAs can be consumed under both aerobic and anoxic conditions, kinetic studies showed that these fatty acids were degraded at a much slower rate than sugars and other substrates (Chipasa and Mędrzycka, 2006; Novak and Kraus, 1973). Consequently, FOG can accumulate within the reactors potentially enhancing the risk of foaming through stimulating the growth of filamentous microorganisms (Lefebvre et al., 1998). At the treatment works, in order to protect downstream biological treatments, FOG can be separated from the wastewater through a preliminary step. The valorisation of the collected FOG has been reported by Silvestre et al. (2011) and found to increase methane yields by 138% in anaerobic digesters with the addition of 23% of FOG on VS basis to sewage sludge. Using this type of FOG as co-substrate is desirable from an energy generation point of view, however separation through a preliminary treatment step is not always a viable option for the treatment of municipal wastewater as inclusion requires the installation of additional assets (Pastore et al, 2014).

Alternatively, upgrading conventional primary sedimentation tanks (PSTs) would help avoiding the accumulation of FOG in biological processes as well as accommodating for increased loading (e.g. population growth). In relatively recent years, enhanced primary treatment has been introduced through the use of coagulant dosing prior to sedimentation to increase solids and/or phosphorus removal to meet stricter effluent discharge consents. FOG removal rates were reported at 47-47% for conventional sedimentation (Gehm, 1942; Loehr and de Navarra Jr., 1969; Murcott, 1992) and 59-71% using chemical dosing (Kuo and Goh, 1992; Murcott, 1992). However, how this relates to FOG management has not been explored in these studies but would be needed to inform an end-of-pipe strategy.

As an alternative to conventional primary treatment, the use of dissolved air flotation (DAF) could be considered to improve FOG capture at STWs. DAF is commonly used in drinking water and industrial waste treatment and works by injecting air saturated pressurised water into the tank. This results in the formation of a large mass of small bubbles (40-60 μm) which combines with the solids reducing their density and causing them to float to the surface where they are removed (Edzwald, 2010). The technology is particularly effective against low density solids and hence it is posited to offer a real potential for FOG removal at STW. To illustrate, in FOG-rich industrial wastewaters, removal levels of 89 and 98% have been reported from slaughterhouse effluents (Al-Mutairi et al., 2008; Karpati and Szabo, 1984; Travers and Lovett, 1985), 60% from dairy wastewaters (Monroy et al., 1995) and up to 97% in effluents from meat-manufacturing plants (El-Awady, 1999). In comparison, only a few studies have reported FOG removal efficiencies using DAF in urban wastewaters, ranging from 28% up to 72% (Kuo and Goh, 1992; Levy et al., 1972). The paucity of reported municipal cases reflects the combination of increased maintenance/operational complexity and energy demand associated with bubble generation. However, since then, technologies have become available with more optimised recycle systems and new methods of forming microbubbles (Crossley and Valade, 2006). Further research would be needed to determine whether these advancements support the upgrading of STWs with DAF for FOG management.

From sewers, to pumping stations and finally to STWs, FOG is an operational and financial burden for water utilities. Whilst FOG is thought removed alongside sewage sludge in primary treatment at STWs, it is posited that energy generation through anaerobic digestion only equates to a small fraction of operational costs engaged. With a commitment to deliver affordable water and wastewater services to customers, improving the management of FOG may help the water sector not only to reduce costs but also to achieve wider social and environmental challenges. Consequently, the management of FOG at source has the potential to avoid downstream impacts in the networks and at the treatment works. Under section 111 of the Water Industry Act 1991, UK water companies can prosecute any premises discharging "any matter likely [..] to interfere with the free flow of [wastewater]" (UK Parliament, 1991). In turn, this can result in substantial fines or even imprisonment for offenders. Where the water company has incurred costs in dealing with the detrimental effects on the sewers, it can take legal action to recover these costs. To achieve compliance, a wide range of approaches have been developed, often being dependant on the source of FOG: domestic, commercial or industrial.

## 1.3 Managing FOG from commercial sources

### 1.3.1 FOG sources

In commercial kitchens (i.e. FSEs), there are two main FOG streams: used cooking oils (UCOs) which are primarily wasted oils from deep fat fryers and FOG generated during washing up activities and cleaning of kitchen appliances. Ribau et al. (2018) estimated volumes of UCOs arising for 23 countries to vary between 0.9-8.0 kg.capita$^{-1}$ per year. Other estimates developed in the US were within a similar range, 1.4-9.6 kg.capita$^{-1}$ per year (Wiltsee, 1998). In the UK, the recycling of UCOs is encouraged through financial incentives from suppliers of fresh oils for use as a biodiesel feedstock (Smith et al., 2013) mainly driven by the Animal By-Products Regulations (Public Health England, 2013). Whilst there is no clear evidence regarding the discharge of UCOs in the sewers, it is posited that these oils are not the main cause of deposition in the networks, but other sources are (e.g. washing-up activities and cleaning of appliances). Techniques are commercially available to minimise these emissions. Still, their relative contribution to the FOG problem has not been studied in depth and is often limited to theoretical estimations with insufficient data published from field validation (Table 1-1). Thus, in designing and implementing appropriate control measures with a focus on energy recovery, one can understand the value of such dataset.

### 1.3.2 Remediation techniques

Minimisation of FOG discharges can occur through a combination of kitchen management and staff training and source-based remediation techniques in the form of physical separation or the use of biological additives (Wallace et al., 2017).

Physical remediation is achieved using grease abatement devices to retain FOG from wastewater based on the difference of density between oil and water. Depending on the unit size and its location in FSEs, grease separators are commonly categorised into: (1) grease interceptors referring to large underground trapping systems, or smaller indoor systems subcategorised into (2) grease traps and (3) grease removal units (GRUs). Grease interceptors and grease traps are passive systems requiring their entire content to be pumped out producing wastes, also referred to as grease trap waste (GTW), with large volumes of water (95%) whilst grease and solids layers only account for 3% and 2% (Chan, 2010; Kabouris et al., 2009b, 2009a; Miot et al., 2013; Robbins et al., 2011). In turn, this reduces the value proposition of FOG as further treatment is required to remove the excess water. Switching to GRUs addresses this challenge as the FOG component is collected separately. Yet, there is genuine paucity of information on the character of GRU-collected FOG and is currently assumed to be similar to UCOs (Wallace et al., 2017). In the UK, field experience has demonstrated that only 10-20% of FSEs currently manage their effluents using grease separators (ECAS, 2016; Thames Water Utilities, 2018). Whilst legislations exist broadly referring to FOG, the Building Regulations 2000 is the only piece providing recommendations for “a grease separator complying with BS EN 1825-1:2004 and designed in accordance with BS EN 1825-2:2002 or other “effective means of grease removal” to be installed in FSEs (Table A 1). Critically, this standard refers to large separators unsuitable for many FSEs (e.g. located in narrow town centres). Further to this, in absence of clarified statement, “other effective means” is left to interpretation and it is posited that FSEs will opt for the more convenient and less expensive technique to the detriment of effective systems.

By contrast to physical remediation, biological additives aim at degrading FOG entering sewerage systems or material already deposited using enzymes and/or microorganisms. The degradation of FOG by bio-additives works by the initial enzymatic breakdown of the parent triglyceride compounds followed by biological utilisation of the breakdown products (i.e. fatty acids). The process is sensitive to the balance of other components such as easily accessible carbons which reduce enzyme production and the requirement for appropriate levels of nitrogen to promote growth (Drinkwater et al., 2015; Gurd, 2018). Numerous commercial products are available on the market but years of experience with inconclusive trials within water utilities have led to scepticism regarding their efficacy (Mattsson et al., 2014; Mosholi and Cloete, 2018; Shaffer and Steinbach, 2007). Biological additive dosing remains an active area of research and its impact reduces the potential value proposition of FOG and as such is outside the scope of the current thesis.

### 1.3.3 Enforcement and compliance

In the UK, the enforcement of FSEs is relatively recent and, so far, has been mainly conducted by water utilities concerned with the accumulation of FOG in sewers (Brockett, 2016; Hackett, 2018). Enforcement often relies on visiting premises in critical areas (e.g. suffering from high sewer blockage rate) and raising awareness on the FOG issue. Generally, it will take a couple of visits before a FSE reaches compliance. The UK food industry is shared amongst private, chain and franchise businesses. As such, in an effort to reach out to a larger number of premises, some UK water companies have started interacting with the chains and franchises.

By contrast to a one-size-fits-all type of program, this type of approach targeting FSEs hotspot make costs more reasonable. Interestingly, in the UK, the Food Hygiene Rating Scheme (FHRS), developed as a partnership initiative between local authorities and the Food Standards Agency, could represent an interesting synergistic approach for the water sector to optimise FOG enforcement. Environmental Health Officers (EHOs) routinely conduct inspections of FSEs to ensure required hygiene standards are met. Problems arising from discharges of FOG into the drains resulting in failure to comply with the Food Hygiene Regulations could therefore result in prosecution or an emergency prohibition order preventing trading on the premise. Still, EHOs are mainly interested in food hygiene and do not enforce FOG management. On the contrary, there has been reports of cases where EHOs have identified grease separators as a potential health hazard and recommended their removal (Drinkwater et al., 2017; Grenz and Patel, 2007). Further to this, the fragmentation in local government structures could translate in significant differences in terms of practices between councils. Still, it is posited that a joined-up approach could benefit both local councils and water utilities. To drive further cooperative actions, research is required to understand drivers and barriers to implementation.

FOG discharges from FSEs are not an isolated to the UK, and programmes have been implemented around the world to minimise their impacts on sewers (Table A 2). These approaches present similarities as local codes of practices were often amended by municipalities so that the installation and maintenance of grease abatement devices were made a legal requirement. In some cases, such as Ireland (Dublin), New Zealand and Australia, stricter approaches were undertaken requiring FSEs to apply for a permit to discharge to sewers (i.e. as is the management of industrial-scale activities in the UK, section 1.5). Regular inspections are regarded by many authorities as a critical component of FOG management, providing the opportunity to promote best management practices (Helms and Dulac, 2016; Seiler, 2016). During these inspections, FSEs are required to present grease separators' maintenance records. Standards generally recommend a cleaning frequency of once a week to once in several weeks (ASME, 2000; CSA Group, 2012; Plumbing and Drainage Institute, 2012). In practice, many municipalities require cleaning on a 90-day basis or following a 25% rule (i.e. cleaning is required before the grease and solid layers reach 25% of the total volume of the GTS) (Helms and Dulac, 2016; Seiler, 2016). In Singapore, for instance, premises are advised to determine their optimum cleaning frequency by monitoring their discharges but are recommended to clear and clean their equipment at least once every two weeks. As these interventions are generally run by water authorities, information tend to be scarce in literature regarding their actual cost effectiveness. Wallace et al. (2017) reported a decrease of FOG blockages by 90% after implementation of the FOG programme in Dublin. In Dallas, a 95% reduction of sewer overflows was estimated post-implementation (Helms and Dulac, 2016). In Australia, City West Water and South East Water noticed a reduction by more than 50% of sewer blockages within five years of programme implementation (Alam, 2003). Consequently, managing FOG from FSEs at source would provide significant benefits for the UK water industry. Still, evaluating the associated costs with such approaches is needed to facilitate decision making. This is of importance as FOG can also be used as an energy source providing commercial opportunities beyond sewer protection (as discussed in section 1.6).

## 1.4 Managing FOG from domestic sources

FOG from domestic sources is a mixture of vegetable oil used for frying or in food preparation and residual animal fats. These wastes generally enter the drainage system during washing up activities but can be further minimised by placing any food scraps in kitchen bin or food waste caddies along with dry wiping plates and pans (Foden et al., 2017).

In the UK, local authorities have a statutory duty to ensure that household waste is collected and disposed of. Practices, in terms of household waste management, vary on a local authority basis. Still, the current provisions of local councils for the management of FOG has been of little interest. Instead, interventions are mainly conducted by water authorities in localised areas with the joint aim of reducing FOG and unflushable items entering the sewers (Anglian Water, 2014; Foden et al., 2017; Georges et al., 2017; Yorkshire Water, 2015). Outside the UK, it is common to find residential FOG programmes build around the provision of drop-off containers made available for the disposal of domestic FOG (Table A 2). By contrast to FSEs, these interventions are often of a lower priority for water companies possibly due to the perceived lower contribution of households to the FOG problem. Whilst there is a clear paucity of published validated data, theoretical estimates developed in the UK ranged between 0.6 and 4.4 kg.year$^{-1}$ per household being significantly lower than other theoretical estimates produced for FSEs (Table 1-1). To illustrate, the lowest FOG production rates for FSEs were evaluated for “club organisation” at 5 kg.year$^{-1}$ per premise by Doherty (2009). However, at the highest, FSEs production rates are equivalent to 1,000 households (Table 1-1). Despite demographic differences within populations (e.g. social class, ethnicity, religion) known to influence eating patterns (Murcott et al., 2013) and posited to affect FOG generation rates, it is very likely that the impact of households is not negligible in densely populated areas such as London. Critically, the paucity of available data associated to domestic FOG limits the ability to understand the scale of the contribution it makes and the impact its management could have.

## 1.5 Managing FOG from industrial sources

Industrial sources of FOG, often associated with dairy, food and meat processing plants, are the only emissions monitored and controlled by UK water utilities through trade effluent consents. These permits impose limits on volumes of FOG allowed to be discharged in the sewers. Based on consented maximum discharge flows allowed for each property obtained from the Thames Water Utilities Ltd. (TWUL) trade effluent register, it was estimated that FOG emissions from an industrial facility are ten times higher than that of a FSE (Table 1-1). To meet these discharge consents, grease interceptors or dissolved air flotation (DAF) units are common practice within the industry (Katsuyama, 1979; Kosseva, 2013). Consequently, with pre-treatment in place and on-going monitoring, industrial sources are assumed to have a lower contribution to the accumulation of FOG in sewerage systems.

**Table 1-1** Quantities of FOG from domestic, commercial (FSE) and industrial sources estimated in literature.

| Waste | Source | kg.year$^{-1}$.premise$^{-1}$ | kg.capita$^{-1}$.year$^{-1}$ | Reference |
|---|---|---|---|---|
| **Domestic (UCO)** | Estimates from EU Member States | | 3.5 | European Biomass Industry Association (2015) |
| **Domestic (UCO)** | Austrian UCO collection from households | | 0.2 to 1.0 | Ortner et al. (2016) |
| **Domestic (FOG)** | Estimates of UK household food and drink wastes | 2.6 | 1.1 | Quested et al. (2013) |
| **Domestic (FOG)** | Estimated FOG disposed to sewers in the UK | 0.6 to 4.4 | 0.3 to 1.9 | Gelder and Grist (2015) |
| **Domestic (FOG)** | Estimates from the Capital Regional District in Canada | 5.6 | 2.6 | Blanc and Arthur (2013) |
| **FSE (FOG)** | Based on FSE size and FOG management practices | 17.0 to 115.8 (small)<br>54.3 to 370.4 (medium)<br>108.6 to 740.9 (large) | | Gelder and Grist (2015) |
| **FSE (FOG)** | Volumes of FOG collected from FSEs in Dublin | 5 to 10,768 | | Doherty (2009) |
| **FSE (GTW)** | Estimates of GTW from 30 US metropolitan areas | 355 to 7,918 | 0.7 to 12.0 | Wiltsee (1998) |
| **FSE (GTW)** | Calculated from GTW sampling at 39 US FSEs | 50 to 10,117[a] | | Kennedy/Jenks Consultants (2011) |

|  |  |  |  |  |
|---|---|---|---|---|
| **FSE (GTW)** | GTW collected from FSEs and STWs in Finland |  | 6.6 | van der Veen (2013) |
| **FSE (GTW)** | GTW collected by two Australian water authorities | 12,060 and 12,240 |  | Alam (2003) |
| **FSE (UCO)** | Estimates of UCOs produced in 23 countries |  | 0.9 to 8.0 | Ribau et al. (2018) |
| **FSE (UCO)** | Data from UCO collectors in London | 1,919 |  | Smith et al. (2013) |
| **FSE (UCO)** | Estimates of UCOs from 30 US metropolitan areas | 926 to 5,825 | 1.4 to 9.6 | Wiltsee (1998) |
| **Industrial** | Dairy processing facility | 223 to 145,080[b] |  | TWUL trade effluent consents |
| **Industrial** | Food processing facility | 56 to 308,016[b] |  | TWUL trade effluent consents |

[a] Estimated as dry solids; [b] Calculated from maximum discharge flow allowed and limit set on FOG concentration at 300 mg.L$^{-1}$

## 1.6 From nuisance to energy

Once captured, FOG materials have been historically diverted to landfills or, prior the introduction of the Animal By-Products Regulations, used in animal feed (e.g. UCOs) (Public Health England, 2013). Other routes have included land application, composting or rendering for lubricants or soaps (Ragauskas et al., 2013). Due to its lipid-rich nature, FOG is an attractive substrate for energy recovery. To date, the two most studied applications have been conversion to biodiesel and biogas generation through anaerobic co-digestion. This section aims to provide an overview of the FOG-to-energy routes.

### 1.6.1 Biodiesel production

Biodiesel is an alternative to petroleum-based fuel mainly composed of fatty acid methyl esters (FAMEs) produced either from the acid- or alkali-catalysed transesterification of fats and oils (Van Gerpen, 2005; Math et al., 2010; Tu, 2015; Zhang et al., 2003) with conversion yields up to 95% (Table 1-2).

**Table 1-2** Biodiesel conversion of UCOs and GTW.

| Waste | Alcohol[a] | Catalyst (%w/v) | T (˚C) | Time (min) | Yield | Reference |
|---|---|---|---|---|---|---|
| **FSE (UCO)** | Methanol (7.8-8.1) | KOH (0.7%) | 30-50 | 80-90 | 88-90% | Phan and Phan (2008) |
| **FSE (UCO)** | Ethanol (30%) | KOH (1.2%) | 35 | 30 | 78.5% | Allawzi and Kandah (2008) |
| **FSE (UCO)** | Methanol (7.5-1) | Sodium methoxide (1%) | 60 | 60 | >95% | Alcantara et al. (2000) |
| **FOG (GTW)** | Methanol (9:1)<br>Methanol (6:1) | $H_2SO_4$ (0.5%)<br>KOH (1.2%) | 95<br>80 | 60<br>30 | 94.1% | Park et al. (2010) |
| **FOG (GTW)** | Step 1: Methanol (43%)<br>Step 2: Methanol (26%) | $H_2SO_4$ (2.5%)<br>KOH (1%) | | 240<br>60 | 95.5% | Karnasuta et al. (2007) |

[a] Reported as alcohol to oil volume ratio

The composition of feedstocks is critical for biodiesel conversion as parameters including water and free fatty acid (FFA) contents can negatively affect transesterification reactions leading to a more laborious and expensive process (Sanford et al., 2009). It is generally admitted that feedstocks with over 3% FFAs will require an additional treatment step to esterify FFAs to methyl esters using an acid catalyst such as sulfuric acid (Dorado et al., 2002; Van Gerpen, 2005).

UCOs from FSEs are commonly used as biodiesel feedstock as they exhibit low water and FFA contents (Cheah et al., 2016; Hailei and Hui, 2014; Sanford et al., 2009; Sanli et al., 2011; Supple et al., 2002). Studies focusing on the utilisation of domestic FOG as biodiesel feedstock have been mainly limited to waste frying oils (Berrios et al., 2010; Hailei and Hui, 2014), further posited to be an easily processed material resembling FSE UCOs. By contrast, GTW, reported with water content up to 80% wt. (Canakci, 2007; Kabouris et al., 2009b; Miot et al., 2013; Parry et al., 2008) and FFA concentrations around 30% (Canakci, 2007; Karnasuta et al., 2007; Kobayashi et al., 2016) requires water to be removed and an additional treatment converting the FFAs prior to the transesterification reaction (Van Gerpen, 2005). In the UK, conversion into biodiesel from FOG substrates is already well implanted with 77% of biodiesel produced from UCOs (UK Department for Transport, 2019). Yet, no research has been published on the use of FOG deposits as biodiesel feedstock.

### 1.6.2 Anaerobic co-digestion

Anaerobic digestion is a well-established process converting organic matter into biogas and digestate, further utilisable as fertiliser, through multiple sequences of metabolic pathways in the absence of oxygen. The resulting methane-rich biogas can be used as a fuel in a combined heat and power (CHP) plant or for further processing (i.e. gas to grid). This treatment has been widely deployed within the water industry for the safe treatment and disposal of sewage sludge improving renewable energy utilisation.

Anaerobic co-digestion, consisting in the joint treatment of two of more substrates with complementary characteristics, is a desirable option to enhance current biogas generation and organic matter degradation. FOG is a valuable co-substrate as lipids, which are its most prominent components, are capable of yielding more methane, at 903.9-1,101.2 mL.g volatile solids (VS) added$^{-1}$, than both proteins and carbohydrates, respectively at 302.5-407.3 and 191.8-359.3 mL.g $VS_{added}^{-1}$ (Labatut, 2012). The potential of GTW for co-digestion with sewage sludge has been reported by many authors as summarised by Long et al. (2012). Davidsson et al. (2008) showed that when sewage sludge and GTW (10-30% of total VS load) were co-digested under mesophilic conditions, methane yields increased up to 27%. Similarly, Kabouris et al. (2009) showed that up to 48% of GTW (of the total VS load) could be digested with a mixture of primary sludge and thickened waste activated sludge with no inhibitory effects on the process, with a three-fold increase in the methane yield (Table 1-3).

**Table 1-3** Anaerobic co-digestion of FOG and sewage sludge in semi-continuous mesophilic reactors at laboratory scale.

| Source of FOG | OLR (kgVS.$m^{-3}$.$d^{-1}$) | HRT (days) | Max.FOG added (% VS) | Max. methane yield (mL.g $VS_{added}^{-1}$) | Increase in methane yield from control (%) | Reference |
|---|---|---|---|---|---|---|
| **GTW** | 2.4-2.7 | nr | 10 | 360 | 24 | Davidsson et al. (2008) |
| **Polymer-dewatered FOG** | 4.3 | 12 | 48 | 473 | 66 | Kabouris et al. (2009a) |
| **Un-dewatered GTW** | 2.9 | 15 | 46 | 641 | 40 | Yalcinkaya and Malina (2015b) |
| **Fat-rich waste from meat processing plant** | 2.9 | 10 | 18 | 288 | 37 | Grosser and Neczaj (2016) |
| **GTW** | 4.2 | 20 | 68 | 620 | 50 | Liu and Buchanan (2011) |

OLR: organic loading rate; HRT: hydraulic retention time; nr: not reported.

Despite these reported benefits, FOG has also been associated with inhibitory actions from LCFAs which are lipids with 14 or more carbons making up the majority of the FOG. Process failures have been reported at high FOG loading rates as LCFAs accumulates in the reactor causing sludge flotation, digester foaming and blockages of pipes and pumps (Girault et al., 2012; Luostarinen et al., 2009; Noutsopoulos et al., 2013). The adsorption of LCFAs onto cell walls has been proposed as the main mechanism to explain this inhibition, further leading to direct toxicity and/or limiting substrate transport (Pereira et al., 2005).

The toxicity of these fatty acids is dependent on their concentrations and their types (Dasa et al., 2016). Several authors have reported their toxicity as the fifty percent inhibition concentration ($IC_{50}$) which is defined as the concentration causing a 50% relative activity loss. The lowest $IC_{50}$ were reported for oleic (C18:1) and linoleic acids (C18:2) respectively at 50 and 30 $mg.L^{-1}$ (Alves et al., 2001; Lalman and Bagley, 2000). In comparison, other main LCFAs identified in FOG were found less inhibitory with $IC_{50}$ reported at 1,100 $mg.L^{-1}$ for palmitic acid (C16:0) (Pereira et al., 2005), 1,500 $mg.L^{-1}$ for stearic acid (C18:0) (Shin et al., 2003) and for linolenic acid (C18:3) at 500 $mg.L^{-1}$ (Prinst el., 1972). Thus, different strategies have been proposed to overcome the inhibition caused by LCFAs including enzymatic pre-treatments (Bouchy et al., 2012; Cirne et al., 2007) and the application of pulse-feeding procedures (Palatsi et al., 2009; Ziels et al., 2017). The physicochemical characterisation of the waste to be co-digested, including fatty acid composition, is therefore recommended in order to minimise possible detrimental effects on the digestion process. However, most of the research to date has focused on GTW with very little consideration to other types of FOG (e.g. domestic, sewer deposits, accumulations at STWs) or more recent advancements in FOG separation techniques (e.g. GRUs). This needs to be addressed when developing a business case for the management of FOG as it will support optioneering.

In the UK, water companies possess the largest asset base of anaerobic digesters. However, there is an additional complication for anaerobic co-digestion as FOG and sewage sludge fall under different regulatory regimes implying that the co-digestate produced is still a waste under the revised Waste Framework Directive requiring potential expensive permitting for its disposal to land or treatment (Iacovidou et al., 2012). However, alternatives to land-based uses for the digestate, such as thermal treatment (e.g. gasification or pyrolysis), are being under serious consideration within the water sector and, as such, could help addressing this shortcoming.

Despite the reported benefits from co-digestion, studies have not taken into consideration advancements in sludge treatment. In recent years, the water sector has significantly invested in sludge pre-treatment to overcome slow degradation rates from the initial substrate degradation step (i.e. hydrolysis). One of the most commonly applied techniques is thermal hydrolysis (TH) which solubilises organic matter making substrates more available to microorganisms. However, TH has been shown to be unsuccessful at degrading FOG into more biodegradable substrates (Charuwat et al., 2018; Cuetos et al., 2010) suggesting that this treatment would need to be bypassed. Adding FOG into TH sludge then poses a potential risk as the easily degradable material may limit uptake of the more complex FOG compounds and hence may restrict the value proposition of FOG through this route.

### 1.6.3 Other recovery routes

Incineration can be applied to FOG due to their high energy content. However, the large volumes of water found in GTW would require to be reduced prior to the process possibly not favouring this recovery route (Awe et al., 2017). More recently, the conversion of FOG into gaseous and liquid fuels achieved through pyrolysis or thermal-catalytic cracking has been reported by several authors (Pratt et al., 2014; Sim et al., 2017; Strothers et al., 2019). Notwithstanding, more research has been suggested to better understand reaction mechanisms and identify optimum operating parameters in order to increase commercial viability (Sim et al., 2017).

## 1.7 Aims and objectives

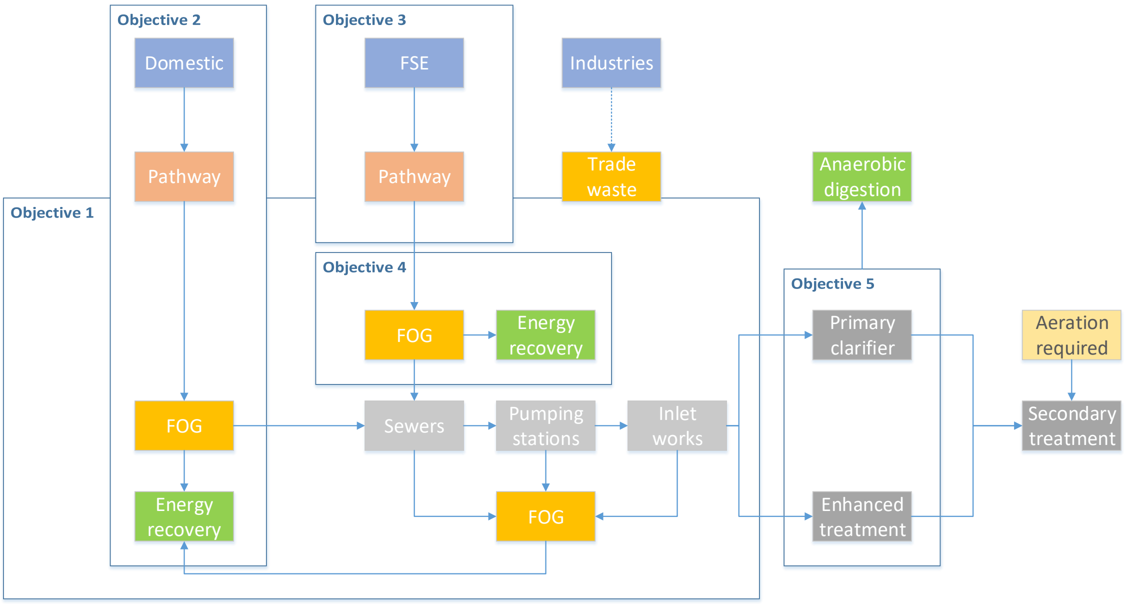

**Figure 1-1** Thesis structure in relation to FOG management.

The aim of this thesis is to quantify the different FOG streams in a wastewater catchment and to characterise their physicochemical properties and digestibility with sewage sludge. In doing so, the thesis will establish a scientific evidence base in order to assess the potential of alternative FOG management options and provide guidance to water utilities for improving current practices with more sustainable approaches. The aim of this project has been achieved through the completion of the following objectives:

1. To clarify the variation amongst FOG wastes collected at different locations in the wastewater catchment (i.e. source, networks and STWs), in terms of their physicochemical properties, through laboratory analyses, and biomethane potentials using batch reactors.
2. To collect FOG from a set number of households to generate quantitative and qualitative data on FOG from domestic sources to inform decision making for energy recovery from these materials in relation with current regulatory framework.
3. To produce a comprehensive understanding of the contribution of current practices in kitchens to FOG discharges.

4. To use GRUs to quantify volumes of FOG collectable from FSEs and investigate the utilisation of FOG in anaerobic digesters treating TH sewage sludge.
5. To use historical datasets to determine the performance of conventional primary clarifiers at STWs in removing FOG and to investigate the potential of enhanced primary treatment as an alternative treatment in order to understand the business case for end-of-pipe FOG strategies.
6. To examine the current economic impact of FOG in a wastewater catchment and develop a business case for the collection of FOG at source.

## 1.8 Thesis structure

This thesis is presented as a series of Chapters, 2 to 6, formatted in the style of papers for publication. All the papers were written by the primary author Thomas Denis Collin, reviewed and edited by Rachel Cunningham of Thames Water Utilities Ltd. (TWUL) (UK) and Prof. Bruce Jefferson of Cranfield University (UK) and proofread by Dr. Jitka MacAdam of Cranfield University (UK). All experimental work was designed and completed by Thomas Denis Collin. Chapter 2 was completed with the contribution of Dr. Raffaella Villa of De Montfort University (UK). The work conducted on “Evaluating the potential of domestic FOG for energy recovery” (Chapter 3) was assisted by Melani Deb of TWUL (UK). The survey of FSEs presented in Chapter 4 was originally designed by Anna Cermakova of Cranfield University (UK) who interviewed half of the respondents whilst the contribution of the other half was brought by Thomas Denis Collin. In addition, support was provided by Prof. Paul Jeffrey of Cranfield University (UK). Experimental work conducted as part of Chapter 6 was assisted by Mohammed Qasim Asghar of TWUL (UK). The contribution of the main author and the collaborators is further detailed in Table 1-4.

Chapter 2 presents a comparison of the FOG wastes collectable in a wastewater catchment (source, sewer systems and STWs) in terms of their physicochemical properties, determined through laboratory analyses, and anaerobic biodegradability using batch reactors, and provides an energy assessment of these wastes (Chapter 2, paper 1 – published in *Waste Management*: Collin, T. D., Cunningham, R., Jefferson, B. and Villa, R. Characterisation and energy assessment of fats, oils and greases (FOG) wastes at catchment level).

Chapter 3 is designed to quantify and characterise FOG from domestic sources. Collection using reusable containers was identify as a cost-effective method to achieve this. The FOG production rates from 31 households were recorded over the course of a one-year study. The collected wastes were further described in terms of their physicochemical characteristics through laboratory analyses. Anaerobic batch tests, at laboratory scale, were conducted to determine the biomethane potential of this type of FOG and evaluate its digestibility with sewage sludge. A survey of local authorities was conducted to understand the place of FOG in household waste management (Chapter 3, paper 2 – in preparation: Collin, T. D., Cunningham, R., Deb, M., Villa, R., MacAdam, J. and Jefferson, B. Evaluating the potential of domestic fats, oils and greases (FOG) for energy recovery).

To date, current knowledge on FOG from FSEs has been often limited to reports of accumulation in the sewers by water utilities. To address this knowledge gap, a questionnaire-administered survey was developed exploring how FOG is currently perceived and managed by FSEs operators with the aim to bring insights on activities conducted in kitchens contributing to FOG discharges. These results are used to identify potential risks to suggest further efforts to improve FOG management (Chapter 4, paper 3 – in preparation: Collin, T. D., Cermakova, A., Cunningham, R., MacAdam, J., Villa, R., Jeffrey, P. and Jefferson, B. Towards a risk register for improved management of fats, oils and greases (FOG) discharges from food outlets).

GRUs separate the FOG component from kitchen effluents and, in turn, allow an easier quantification of these wastes in comparison to passive systems. These devices were installed at 14 FSEs, and the collected wastes were further characterised in regard to their physicochemical properties. The potential of this FOG for anaerobic co-digestion with sewage sludge (non- and thermo-hydrolysed) was studied in semi-continuous conditions to mimic full-scale operations and assess the process performance (Chapter 5, paper 4 – in preparation: Collin, T. D., Cunningham, R., Villa, R., MacAdam, J. and Jefferson, B. Separation of fats, oils and greases (FOG) from kitchen effluents: Waste characterisation, production rates and potential for anaerobic co-digestion with thermo-hydrolysed sewage sludge)

Chapter 6 documents the occurrence of FOG at 15 STWs over a period of five years to evaluate the performance of conventional and chemical-dosed primary clarifiers to remove FOG. With the aim to further increase FOG removal to avoid downstream overloading of biological treatments, two DAF pilot-scale systems are trialled as an alternative primary treatment. These results are further used to develop an economic analysis assessing the feasibility of end-of-pipe techniques for FOG management (Chapter 6, paper 5 – published in *Science of the Total Environment*: Collin, T. D., Cunningham, R., Asghar, M. Q., Villa, R. MacAdam, J., and Jefferson, B. Assessing the potential of enhanced primary treatment clarification to manage fats, oils and greases (FOG) at wastewater treatment works)

Findings from the previous Chapters were then used to perform an economic analysis describing the current contribution of FOG to operational costs in a case study based on TWUL catchment area (chapter 7). The costs and benefits associated with FOG control programmes at source were further discussed.

Chapter 8 provides a discussion to reflect on the overall outcomes of this thesis and how they relate to the wider context of FOG management.

Finally, Chapter 9 summarises the overall conclusions of this thesis and recommends future areas of focus to drive the development of sustainable approaches for FOG management.

**Table 1-4** Contribution of the author and co-authors to each chapter.

| Chapter | Collaborator (affiliation) | Contributions |
|---|---|---|
| **2** | Thomas Denis Collin (Cranfield University) | Reviewed literature; designed experiments, collected the wastes, conducted solids analyses and anaerobic batch testing; extracted and formatted FSEs data; analysed the data; redacted the paper. |
| | Rachel Cunningham (TWUL), Dr. Raffaella Villa (De Montfort University), Prof. Bruce Jefferson (Cranfield University) | Reviewed and edited. |
| **3** | Thomas Denis Collin (Cranfield University) | Reviewed literature; designed experiments; measured quantities of FOG collectable; conducted solids analyses and anaerobic batch testing; surveyed local councils' websites; analysed the data (including data from previous TWUL trial); redacted the paper. |
| | Rachel Cunningham (TWUL) | Provided insights on the experimental design; reviewed and edited. |
| | Melani Deb (TWUL) | Helped conducting the survey on FOG practices; assisted with the collection of FOG. |
| | Dr. Jitka MacAdam (Cranfield University), Dr. Raffaella Villa (De Montfort University), Prof. Bruce Jefferson (Cranfield University) | Reviewed and edited. |
| **4** | Thomas Denis Collin (Cranfield University) | Reviewed literature; surveyed 68 FSEs; analysed the data; redacted the paper. |

| | | |
|---|---|---|
| | Anna Cermakova (Cranfield University) | Designed the questionnaire; surveyed 39 FSEs. |
| | Prof. Paul Jeffrey (Cranfield University) | Provided support and guidance with the design of the survey. |
| | Rachel Cunningham (TWUL), Dr. Jitka MacAdam (Cranfield University), Dr. Raffaella Villa (De Montfort University), Prof. Bruce Jefferson (Cranfield University) | Reviewed and edited. |
| **5** | Thomas Denis Collin (Cranfield University) | Reviewed literature; designed and conducted experiments (FOG collection and anaerobic co-digestion); analysed the data; redacted the paper. |
| | Dr. Jitka MacAdam (Cranfield University), Dr. Raffaella Villa (De Montfort University), Prof. Bruce Jefferson (Cranfield University) | Reviewed and edited. |
| **6** | Thomas Collin (Cranfield University) | Reviewed literature; analysed data from historical sampling; conducted experimental work on with the flotation plants; redacted the paper. |
| | Mohammed Qasim Asghar (TWUL) | Organised the installation of the flotation plants (as part of another TWUL trial); assisted with the operation of the pilot plants. |
| | Dr. Jitka MacAdam (Cranfield University), Dr. Raffaella Villa (De Montfort University), Prof. Bruce Jefferson (Cranfield University) | Reviewed and edited. |

# 2 Characterisation and energy assessment of fats, oils and greases (FOG) wastes at catchment level

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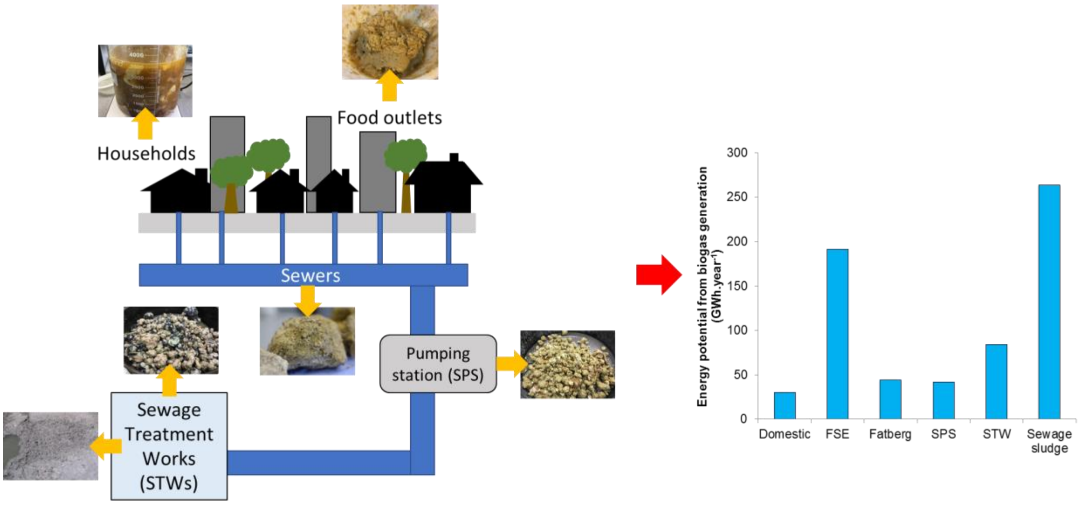
## Graphical abstract



## Abstract

Several of the waste materials that have a negative impact on the sewer system are produced by fats, oils and greases (FOG) discharged from commercial and domestic kitchens. These materials accumulate at different points in the sewer catchment, from kitchens to sewers, pumping stations and sewage treatment works (STWs), and comprise oily wastewater, floating agglomerates and hard deposits. Despite their detrimental effects, these waste materials have a high calorific content and are an ideal feedstock for energy recovery processes. So far, the overall volume of each type of waste and their physical-chemical properties in relation to their collection point are unknown. However, from a management point of view, knowledge on each feedstock quality and volumes is necessary to develop an economic viable solution for their collection and for energy recovery purposes. In this study, FOG wastes collected from households, food service establishments (FSEs), sewage pumping stations, sewers and STWs, were compared to sewage sludge in terms of organic contents and energy potentials. As expected, FOG recovered at source (households and FSEs) were ‘cleaner’ and had a higher energy content. Once mixed with wastewater the materials changed in composition and lost some of their energy per unit mass. Our results showed that around 94,730 tonnes per year of these materials could be recovered from the Thames Water Utilities’ catchment, one of the most populated in the UK. These materials could produce up to 222 GWh.year$^{-1}$ as biogas, close to what is produced with sewage sludge digestion and around 19% of the company energy needs. Finally, even with over six million households in the catchment, the results showed that most of the FOG waste was produced by FSEs (over 48,000 premises) with an estimated average of 79,810 tonnes.year$^{-1}$ compared to 14,920 tonnes·year$^{-1}$ from private households. This is an important outcome as recovery from FSEs will be cheaper and easier if the company decides to implement a collection system for energy recovery.

**Keywords:** Anaerobic digestion; energy from waste; fatberg; sewer deposits; sewage sludge

## 2.1 Introduction

Fats, oils and greases (FOG) discharged from households and food service establishments (FSEs) have been identified as one of the major contributors to blockages in sewer networks and the formation of sewers’ fatbergs (Engelhaupt, 2017). Developing effective FOG management strategies has therefore become a priority for many water authorities, including Thames Water Utilities Ltd. (TWUL), the largest water utility in the UK, which comprises more than six million households in its catchment. These materials accumulate at different points in a sewerage catchment, from kitchens drains to sewers, pumping stations and sewage treatment works (STWs), and they comprise oily wastewater, floating agglomerates and hard deposits. Despite their detrimental effects on the sewer network, FOG-rich wastes have a high calorific content and can be an ideal feedstock for energy recovery processes. An assessment of each material’s quality and volume is necessary to evaluate the economic viability of collecting and using FOG wastes for energy recovery. Thus far, most of the research has focused on used cooking oil (UCO) harvested from FSEs for biodiesel production (Wallace et al., 2017) or grease trap waste (GTW) for the production of biogas in anaerobic digestion (Long et al., 2012). The potential of GTW FOG waste co-digestion with sewage sludge has been reported by many authors, as summarised by Long et al. (2012). Davidsson et al. (2008) showed that when sewage sludge and GTW (10-30% of total volatile solids load) were co-digested under mesophilic conditions, methane yields increased up to 27%. Similarly, Kabouris et al. (2009) showed that up to 48% of GTW (of the total volatile solids load) could be digested with a mixture of primary sludge and thickened waste activated sludge with no inhibitory effects on the process, with a three-fold increase in methane yields. However, little attention has been given to other FOG wastes available in the sewerage catchment, such as fatbergs from sewers, or floating deposits from pumping stations or STWs. The use of these energy-rich materials as co-digestion substrates could offer water utilities a double economic advantage by disposing of unwanted waste and increasing their renewable energy production. Understanding the processing potential of these different FOG-rich materials could help define and drive a more sustainable FOG management at catchment level. For instance, the overall volume of each type of waste and their physical-chemical properties, in relation to their collection point, are still unclear. Furthermore, no attempt has been made to study FOG collected from households, which some authors believe to be one of the major contributors towards FOG discharges in sewer networks (Foden et al., 2017). Wallace et al. (2017) suggested that grease removal units (GRUs) produce a waste similar to UCOs and with fewer impurities than GTW, but no work to date has intended to characterise this waste. Lastly, most of the research conducted on FOG has focused on explaining the mechanisms of formation of FOG deposits (Keener et al., 2008) and very few have reported their potential for energy recovery. This paper aims to clarify the variation among these substrates in regards to their physicochemical properties and biomethane potential as well as to provide an assessment of their volumes and their energy potential within TWUL catchment.

## 2.2 Material and methods

### 2.2.1 Inoculum and substrates

Digested sludge, used as inoculum in batch tests, was obtained from a full-scale anaerobic digester treating municipal sewage sludge. Six FOG wastes were used in this study: (1) domestic FOG (Domestic) collected from 31 households (located in different catchment areas). The samples were blended, heated to 35˚C and sieved to remove any large food particulates. (2) FOG sample from a FSE grease removal unit (GRU). (3) FOG deposit (Fatberg) was manually excavated during the clean-up of a sewer in London (2-3 kg sample). Fat balls samples were collected from two locations: (4) a sewage pumping station (SPS) and at (5) the inlet of a STW (SPS and STW respectively). The FOG deposit and fat balls samples were grinded to produce finer and more homogeneous samples. (6) Floating scum (Floating scum) accumulating at the inlet of a STW was collected and further analysed. Sewage sludge (Sewage sludge), pre-treated through a thermal hydrolysis process, was used as a comparison material.

### 2.2.2 Analytical methods

The physical appearance (i.e. texture and colour) of the different FOG wastes was qualitatively assessed. Dry solids (DS) and volatile solids (VS) were determined according to standard methods (APHA, 2005).

A chemical characterisation of the main organic fractions (e.g. lipids, carbohydrates, proteins and fibres) was performed on each material. Fibres were measured as the organic matter remaining after samples were de-fatted and digested successively with acid and alkali under controlled conditions (Horwitz, 2003). Proteins were determined either with the Dumas method using Leco FP528 (Sciantec Analytical, 2018a) or as total Kjeldahl nitrogen (Sciantec Analytical, 2018b) respectively for solid and semi-solid samples respectively. Lipids were measured using a modified Wiebul acid hydrolysis method (Sciantec Analytical, 2018c). Carbohydrates were estimated as the remaining fraction.

Methylated fatty acids profiles were obtained by gas-liquid chromatography using a free fatty acid phase column of dimensions 25m x 0.20mm ID and detection by flame ionisation detector. Fats and oils were trans-esterified to fatty acid methyl esters by heating under reflux for two hours with a mixture of methanol and sulfuric acid in toluene. The resulting methyl esters were extracted using a small volume of n-hexane. The n-hexane solution was dried using anhydrous sodium sulphate and then transferred to a chromatography vial (Sciantec Analytical, 2018d).

Theoretical biogas production was calculated from the organic components of the materials considering proteins, carbohydrates and lipids respectively yield 496, 415 and 1,014 mL $CH_4$.g $VS^{-1}$ (Angelidaki and Sanders, 2004).

Calorific values were determined experimentally in terms of the higher heating value (HHV) using a calorimeter (Parr model 6100) equipped with a 1108CL oxygen bomb; solid samples were pelletised whereas semi-solid samples were freeze dried (Sciantec Analytical, 2018e). The lower heating values (LHV) were estimated from the measurement of calorific values by subtracting the heat of vaporisation of water in the products as follows:

$$LHV_d = HHV_d \times (1 - M) - H_V \times M \tag{2-1}$$

Where $M$ is the moisture content, $H_V$ is the latent heat of vaporisation of water estimated at 2.447 MJ.kg$^{-1}$ at 25˚C and $HHV_d$ is the gross heating value in MJ.kg$^{-1}$ on dry basis determined as follows:

$$HHV_d = \frac{HHV}{1 - M} \tag{2-2}$$

Where $HHV$ is the measured HHV on wet basis.

### 2.2.3 Batch tests

Triplicate batch testing was used to investigate the biomethane content of each material using an automated methane potential test system (AMPTS) II (Bioprocess Control, Sweden). These assays were performed at mesophilic temperatures (37˚C) using an inoculum to substrate ratio of 2 g $VS_{inoculum}$ per g $VS_{substrate}$. DS and VS were determined before and after the digestion period. The experiment was terminated when the cumulative biomethane production reached a plateau phase (at 60 days). The biomethane production was expressed as biomethane yield, mL $CH_4$.$gVS_{added}^{-1}$, and specific biomethane yield, mL $CH_4$.g $VS_{destroyed}^{-1}$ and adjusted to standard temperature and pressure (STP) as follows:

$$V_{STP} = \left(1 - \frac{P_{vap}}{P_{gas}}\right) \times \frac{P_{gas}}{P_{STP}} \times \frac{T_{STP}}{T_{gas}} \times V_{gas} \qquad \textbf{(2-3)}$$

Where $V_{STP}$ is the volume adjusted to STP, $P_{STP}$ is the standard pressure (101.3 kPa), $T_{gas}$ is the temperature of the measured gas (311 K), $T_{STP}$ is the standard temperature (273 K) and $V_{gas}$ is the measured volume of gas. $P_{gas}$ was calculated as the sum of the partial pressures of methane and carbon dioxide. However, $P_{CO2}$ was neglected in this case as carbon dioxide was removed through the stripping solution. $P_{vap}$ is the water vapour pressure calculated as follows:

$$P_{vap} = 10^{8.1962 - \frac{1,730.63}{T_{gas} - 39.724}} \qquad \textbf{(2-4)}$$

### 2.2.4 Volumes and energy appraisal

Quantities of FOG and sewage sludge were estimated for the whole catchment area. Results from the characterisation and batch testing of FOG were further used for the energetic assessment. The calorific value of methane was assumed at 36 MJ.$m^{-3}$ and the efficiency of combined heat and power engines at 30% (Goss et al., 2017).

#### 2.2.4.1 FOG at source

ArcGIS was used as a support tool for this work to manipulate data with a geographical component. Domestic and commercial properties were respectively extracted from AddressBase® Premium (Ordnance Survey, 2017) and the Food Hygiene Rating Scheme (FHRS) (Food Standards Agency, 2016). A total of 6,543,749 and 68,903 records were obtained for households and FSEs in TWUL catchment. A field survey showed that not all FSEs registered under the FHRS were likely to produce any FOG (Chapters 4 and 5). For each category, a correction factor was applied reflecting the number of establishments likely to produce FOG over the total number of premises (Table 2-1). The correction factor was calculated as the number of premises likely to produce FOG over the total number of establishments for each category. FOG from industrial sources (e.g. food and dairy processing plants) were not included in this assessment as their discharges were assumed to be monitored and controlled under the trade effluent consents by the water utility.

Volumes collectable from households were evaluated at 2.3 kg.household$^{-1}$ per year (Chapter 3). The data for the estimation of FOG generated from FSEs was calculated based on Doherty (2009) and is reported in Table 2-1.

**Table 2-1** Assumptions made on FSEs. Data used to estimate FOG generated from FSEs was based on Doherty (2009). The corrected number of premises were estimated based on the correction factor observed for each business type during the survey work (Chapter 4).

| **Business type** | **FOG collectable (kg.year$^{-1}$)** | **FHRS correction factors** | **Corrected number of premises** |
|---|---|---|---|
| Hotel, bed and breakfast and guest house | 484.5 | 0.83 | 1,615 |
| Hospital, childcare and caring premise | 278.2 | 0.55 | 3,563 |
| Pub, bar and nightclub | 997.3 | 0.50 | 4,840 |
| Restaurant, café and canteen | 498.6 | 0.58 | 23,668 |
| Supermarket and hypermarket | 382.5 | 0.88 | 1,341 |
| School, college and university | 9,152.7 | 0.50 | 5,642 |
| Takeaway and sandwich shop | 2,526.5 | 1.00 | 4,388 |
| Other catering premises | 149.6 | 0.50 | 2,968 |

### 2.2.4.2 FOG in wastewater systems

FOG concentrations were measured monthly at 18 STWs in crude sewage over a period of five years. Briefly, samples were filtered a Whatman$^{TM}$ GF/C grade filter paper. The filter paper was immersed in boiling hexane (40 to 60˚C), using a SOXTHERM® extraction unit, in a pre-weighted glass extraction beaker. Oil and grease were then determined by weight difference and reported in mg.L$^{-1}$. It should be noted that values below the limit of detection of 8.2 $mg.L^{-1}$ were replaced with this value. Oil and grease were measured on average at 57.6 $mg.L^{-1}$ at these STWs (Chapter 6); this average value was used for the other sites. Quantities of FOG were estimated based on dry weather flow, which is the average daily flow received at STWs, and subtracted from undigested lipids originating from human faeces estimated at 4.1 $g.capita^{-1}.day^{-1}$ with a range of 1.9 to 6.4 $g.capita^{-1}.day^{-1}$ (Rose et al., 2015). Volumes collected in SPSs were assumed equal to STWs. Sewer deposits were estimated subtracting volumes of FOG at source (i.e. domestic and FSE) from volumes at STWs.

### 2.2.5 Sewage sludge

Data on sewage sludge generation from anaerobic digestion was obtained from TWUL. Yearly averages of feeding rates in tonnes dry solids per day were used for each anaerobic digestion sites. The average VS content of sewage sludge was assumed at 75%.

## 2.3 Results and discussion

### 2.3.1 Physicochemical characterisation

The six types of FOG waste collected in the catchment had very different physicochemical characteristics. FOG from households and GRUs, semi-solid at room temperature, had a brown-yellowish colour and looked very similar to UCOs (Figure B-1). The sewer deposit sample was solid and harder than the other substrates and contained many contaminants such as wipes and plastic waste. Fat balls from STW were darker than those collected from SPS, but both samples had a softer texture than that of the sewer deposit and contained less contaminants. Finally, floating scum had a yellow-greyish colour, with a less structured form (Figure B-2). Domestic and GRU FOG presented the lowest moisture content of all the materials, with values around 1% and 15% respectively. FOG collected in sewers and fat balls from SPS and STW, had on average lower moisture contents than floating scum 30%, 46%, 47% and 91% respectively (Table 2-2). As expected, moisture content of FOG wastes increased further away from the source point. Similar observations were reported by

Williams et al. (2012), who reported values of 45%, 52% and 70% for pumping station, sewer deposit and STW respectively. Predictably, the lipid content was inversely proportional to the water content, ranging from 85 to 99% DS for STW, SPS, fatberg, GRU and domestic (Table 2-2). Surprisingly, the floating scum, generally believed to be FOG, showed a relatively lower lipid content, and had organic concentrations comparable to that of sewage sludge. As a comparison, lipids in sewage sludge were measured at around 11% DS.

**Table 2-2** Composition in water and organic compounds of different types of FOG wastes available in the catchment.

| **Waste** | **Water (%wt.)** | **Fibres (%DS)** | **Proteins (%DS)** | **Lipids (%DS)** | **Carb. (%DS)** | **Ash (%DS)** |
|---|---|---|---|---|---|---|
| **Domestic** | 1.2±0.1 | 0.1[1] | 0.8±0.2 | 84.5±5.3 | 14.7±5.1 | 0.0 |
| **FSE** | 14.8±11.7 | 0.1[1] | 0.7±0.1 | 101.0±0.4 | 0.0 | 0.0 |
| **Fatberg** | 30.0±2.9 | 0.1[1] | 0.9±0.1 | 93.1±9.2 | 5.0±8.7 | 1.5±0.9 |
| **SPS** | 46.1±2.3 | 3.1±1.2 | 3.8±0.6 | 93.1±4.5 | 0.4±0.7 | 3.5±0.1 |
| **STW** | 47.2±10.9 | 3.3±1.2 | 3.5±0.3 | 94.5±3.3 | 0.0 | 5.0±0.8 |
| **Floating scum** | 91.1±1.5 | 28.3±4.8 | 9.6±1.7 | 13.7±2.4 | 43.7±8.8 | 4.8±3.9 |
| **Sewage sludge** | 90.1±0.03 | 22.9±3.6 | 30.7±1.2 | 11.2±1.3 | 12.4±5.5 | 22.7±0.6 |

[1] Value below the limit of detection; Carb.: carbohydrates.

When examining the availability of FOG wastes, approximatively 79,810 tonnes.year$^{-1}$ could be collectable from FSEs, whereas households would only produce around 14,920 tonnes.year$^{-1}$ (Figure 2-1a). The FOG production rate, calculated from households and FSEs, would be at around 6.4 kg.person$^{-1}$ per year. This result is comparable to data available from previous studies with values ranging from 4 up to 10 kg.person$^{-1}$.year$^{-1}$ (Canakci, 2007).

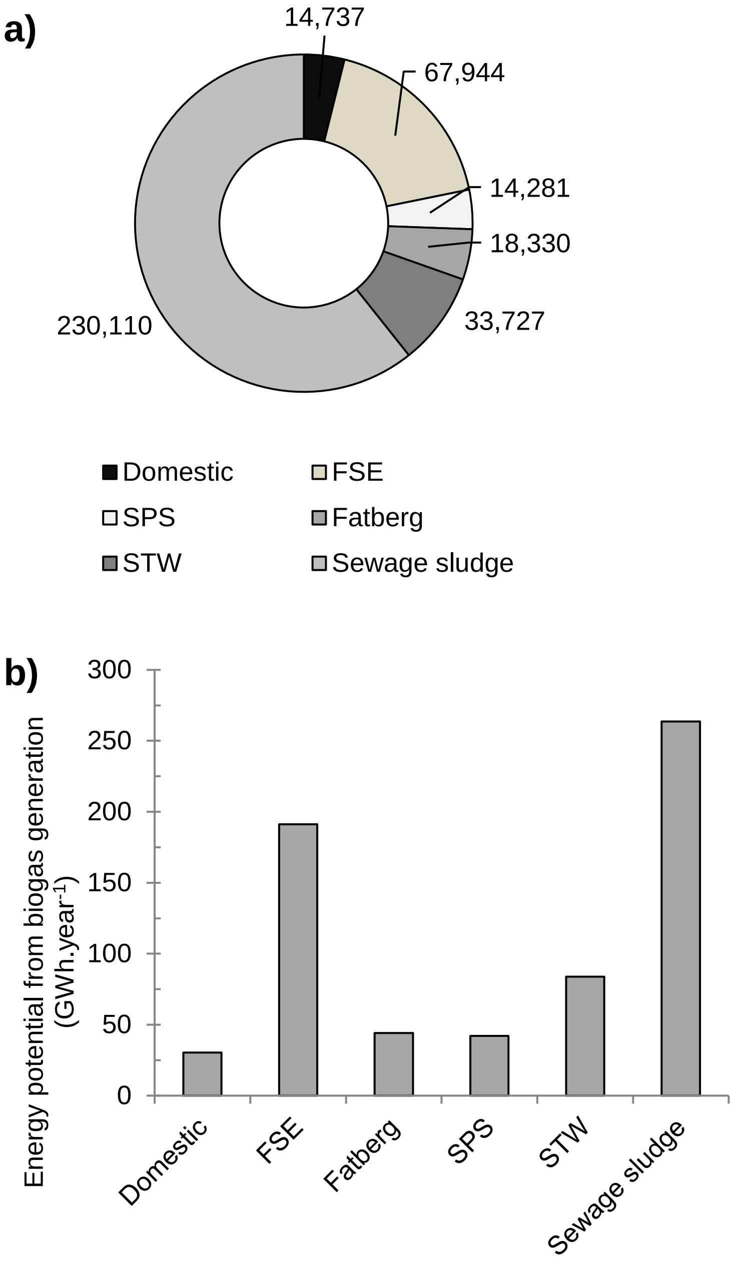


**Figure 2-1** Quantities on a tonnes.year$^{-1}$ dry basis of different types of FOG wastes available in the catchment (a) and their energy potential as biomethane in co-digestion (b).

### 2.3.2 Biogas potential

In order to comprehensively assess the energy recovery potential of all the FOG materials, batch digestion system were used to calculated biomethane yields and biomethane specific yields. All FOG samples produced more biogas than sewage sludge alone (Table 2-3). These values were comparable to methane yields for lipid-rich waste reported by other authors, ranging from 606 to 928 mL CH4.g $VS_{added}^{-1}$ (Davidsson et al., 2008; Luostarinen et al., 2009; Yalcinkaya and Malina Jr., 2015). Sewer deposit, STW fat balls and floating scum displayed a greater standard deviation than the other wastes tested. This was probably due to the preparation of these highly contaminated materials as producing a homogeneous sample was very challenging (Figures B-1 and B-2). The much higher biomethane yields (e.g. biomethane per gram of $VS_{destroyed}$) and therefore bioconversion efficiencies were obtained when digesting FOG compared to sewage sludge (500±31 STP mL CH4.g $VS_{destroyed}^{-1}$) or floating scum (367±105 STP mL CH4.g $VS_{destroyed}^{-1}$), with yields ranging from 695±98 to 908±145 STP mL CH4.g $VS_{destroyed}^{-1}$. The floating scum collected at STW produced less biogas than both FOG and sewage sludge, suggesting a close match to the latter and probably a high content in fibres.

**Table 2-3** Biogas production for FOG and sewage sludge. Theoretical estimates were based on the organic macromolecules composition of the materials.

| Samples | Theoretical biogas production (mL $CH_4$.g $VS^{-1}$) | Biomethane yield (STP mL $CH_4$.g $VS_{added}^{-1}$) | VSd (%) | Biomethane specific yield (STP mL $CH_4$.g $VS_{destroyed}^{-1}$) |
|---|---|---|---|---|
| Domestic | 915±31 | 773±13 | 93±15 | 685±98 |
| GRU | 931±2 | 938±39 | 80±3 | 890±42 |
| SPS | 866±49 | 981±12 | 91±6 | 903±50 |
| Sewer deposit | 963±52 | 801±94 | 64±11 | 908±145 |
| STW | 839±35 | 829±285 | 94±3 | 795±258 |
| Floating scum | 380±6 | 291±101 | 75±8 | 367±105 |
| Sewage sludge | 411±16 | 382±6 | 69±4 | 500±31 |

Analyses on the lipid fraction showed that FOG triglycerides contained long-chain fatty acids (LCFAs) of 14 or more carbons. LCFAs are associated with inhibition of methanogenesis and toxicity to the anaerobic digestion process (Girault et al., 2012; Luostarinen et al., 2009; Noutsopoulos et al., 2013). This inhibition was found to be dependent on concentrations and types of LCFAs (Dasa et al., 2016). Oleic acid (C18:1) was reported as the most predominant LCFA found in GTW with concentrations ranging from 34 to 48% of total fatty acids (TFA) (Canakci, 2007; Suto et al., 2006). Similar observations were made with domestic and GRU FOG where oleic acids were measured at 46±2 and 45±10% of TFA. Vegetable oils have higher content in mono- and polyunsaturated fatty acids compared to animal fats (Gunstone et al., 1986), and are the most commonly used cooking fat in FSEs in the UK (on average about 14 L every 100 meals) (Envirowise, 2008). Accordingly, FOG collected at source shared a relatively comparable fatty acid profile to that of vegetable oils. Despite variations between samples, several authors have reported higher levels of saturation in sewer deposits ranging from 41 to 86% of TFA, with palmitic acid (C16:0) being the most common saturated fatty acid (He et al., 2011; Keener et al., 2008; Nieuwenhuis et al., 2018). Fat balls from SPS presented a slightly lower degree of saturation than sewer deposits, measured at 30±1% of TFA. As a comparison STW fat balls and sewage sludge showed a relatively similar fatty acid profile, with a degree of saturation respectively at 43±1 and 46±1% of TFA. This shift from unsaturated to saturated fatty acids is still unclear (Figure 2-2). Some authors have suggested that micro-organisms might be involved in that transformation (Williams et al., 2012) whilst others have hinted at the contribution of soap products (He et al., 2017).

Fatty acids composition is very important for anaerobic digestion as the different fatty acids are degraded in different way by the microbial communities in the digester and hence have a different impact on the final biogas production. In addition, unsaturated fatty acids must be first converted in saturated fatty acids before being degraded via the β-oxidation pathway (Salama et al., 2019). For example, oleic acids, found predominantly in FOG collected at source, have been reported by several authors to have greater toxic effects on the anaerobic digestion process than saturated fatty acids, such as palmitic acid (Alves et al., 2009; Dasa et al., 2016; Shin et al., 2003). Davidsson et al. (2008) reported slower digestion time of stearic acid compared to oleic acid.

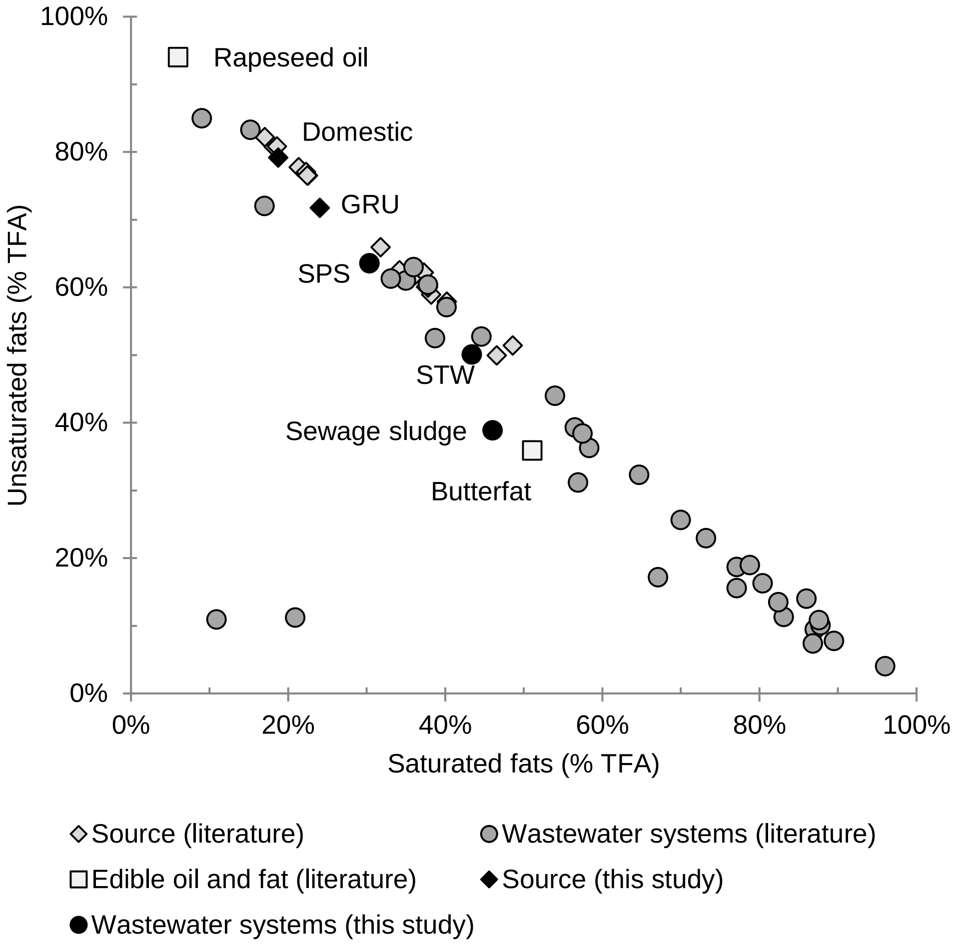


**Figure 2-2** Unsaturated fatty acids reported against saturated fats in FOG wastes as % of total fatty acids. Edible oil and fat are represented with ■ and FOG wastes are categorised as follows: source (◆) and wastewater systems (●).

These results confirm that FOG wastes are desirable substrates for anaerobic digestion even when collected from the networks. However, to avoid detrimental impacts, further care is needed to optimise the feeding regime of FOG materials, not only in terms of quantity but also in terms of source and composition.

### 2.3.3 Energy recovery potential

Higher organic matter and lipids concentration translated into higher energy content which was measured as the calorific content of the different materials using a bomb calorimeter (Figure 2-3). FOG collected at source, domestic and GRU, had high calorific values of 36±4 and 33±4 MJ.kg$^{-1}$ on dry basis respectively. Both values were in the range of those previously reported for GTWs

(Al-Shudeifat and Donaldson, 2010) and UCOs at 35 and 39 MJ.kg$^{-1}$ respectively (Khalisanni et al., 2008). The fatberg sample was measured at 27 MJ.kg$^{-1}$ DS whilst SPS and STW had lower values measured at around 13 MJ.kg$^{-1}$. Floating scum (19 MJ.kg$^{-1}$ DS) and sewage sludge (18 MJ.kg$^{-1}$ DS) showed similar values, indicating a reduction in calorific value as the location extended away from the source point.

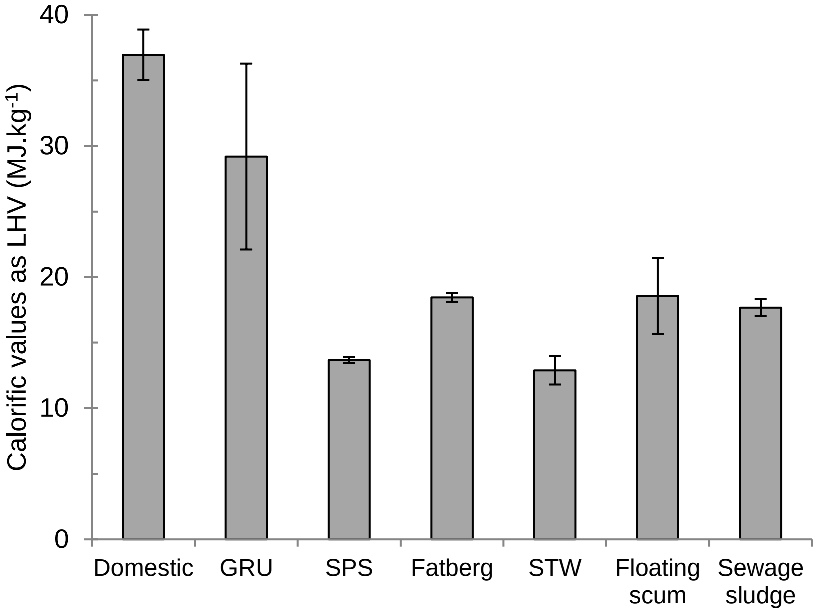


**Figure 2-3** Calorific values, as LHV, of FOG wastes in the sewerage catchment and sewage sludge.

Lipids and water concentration showed a linear inverse correlation for all the samples analysed in this study and those reported in literature (Figure 2-4). Interestingly, oil concentrations in FOG deposits reported by Williams, et al. (2012) were much lower than those measured by this study and Keener et al. (2008) in the US respectively 8.8% and 40.5% on average. This suggests that waste collected from the network is likely to be highly variable in terms of quality and contamination as it gets in contact with sewage and other waste materials in the sewers. Critically, the increased moisture content reduced the lipids fraction by mass indicating that not only does FOG collected from the network require more effort, but this negative is compounded through a reduction in its resultant energy value. The total energy available (i.e. calorific value measurement) plotted against the energy available from the conversion of biogas showed conversion yields ranging from 20 to 80% for FOG and averaging 31% for sewage sludge (Figure 2-5). Not all the energy contained in FOG is convertible to biomethane through anaerobic digestion. Particularly, FOG collected at source demonstrated lower energy conversion yields than other wastes collected further downstream. Facilitating the hydrolysis step, which is the rate limiting step, through pre-treatments (e.g. enzymatic) could help improving the efficiency of the digestion of FOG.

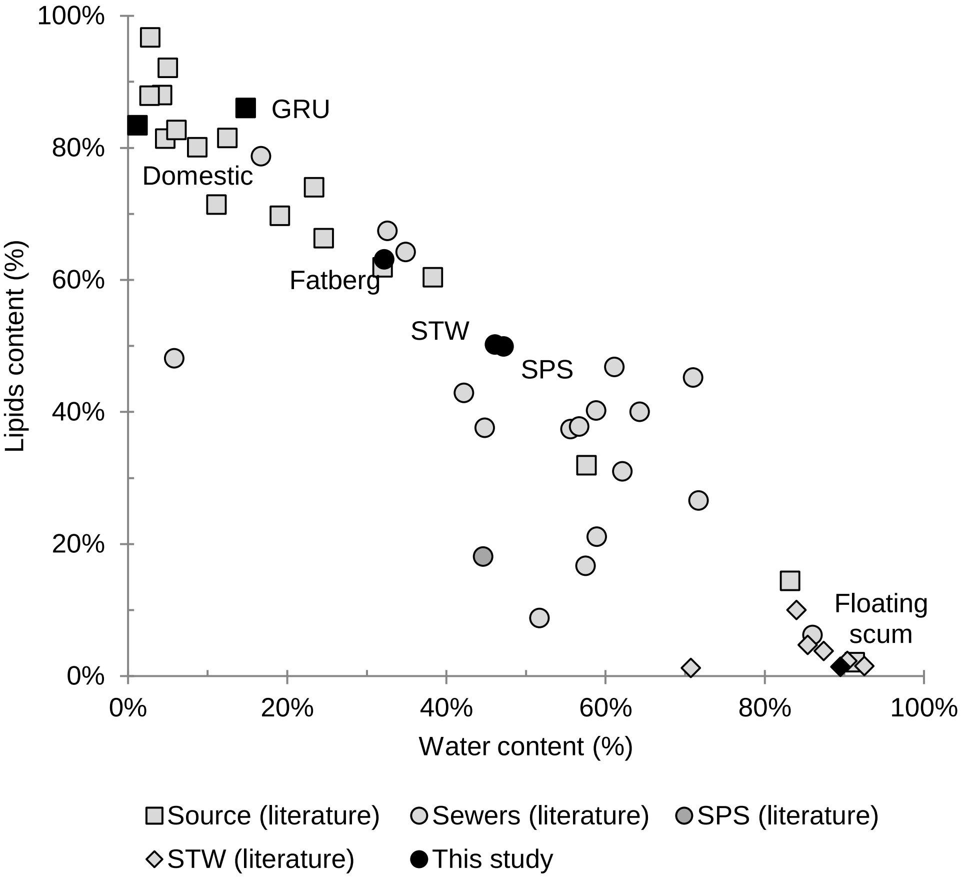


**Figure 2-4** Lipids and water content of FOG wastes from this study and literature (reported as % wet weight). FOG wastes are categorised as follows: source (■) wastewater systems (●) and STWs (◆).

This initial characterisation indicated that materials collected at source with high lipid content, such as domestic and GRU, could be easily used as biodiesel feedstock. Whereas other wastes, such as SPS, sewer and STW, with higher water content, would require an initial dewatering step. The water in the feedstock reacts with the catalyst during the transesterification process leading to a more laborious and expensive process (Sanford et al., 2009). These materials could be better suited for energy recovery through anaerobic digestion. Biogas derived energy from sludge is currently generating 264 GWh.year$^{-1}$. Biogas from sewer and STW could add an additional 128 GWh.year$^{-1}$. Whereas FOG from households and FSEs, estimated at 30 and 191 GWh.year$^{-1}$ of biogas (Table 2-4), could be converted into approximately 59,340 m$^3$ of biodiesel (at 80% conversion and density of 0.9).

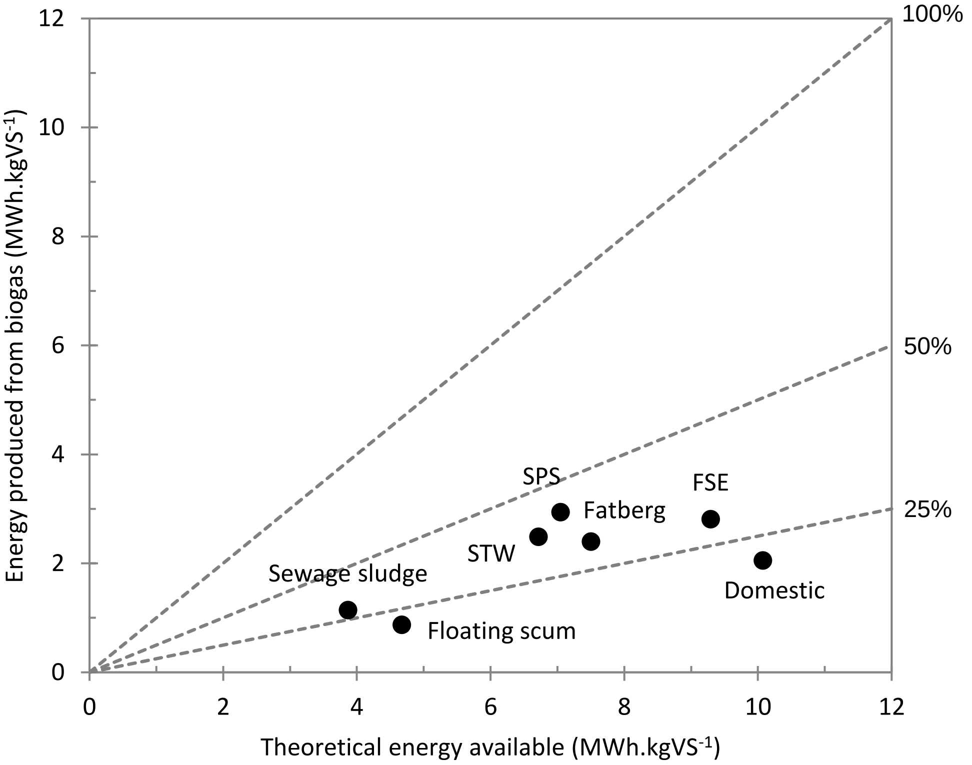


**Figure 2-5** Calorific values of FOG and sewage sludge plotted against biomethane produced for: household FOG (domestic); FOG from FSE’s grease removal unit (GRU); FOG/fat balls from pumping station (SPS) and from sewage treatment works (STW); FOG from sewer deposit (fatberg); FOG from floating scum at the entrance of the STW (floating scum) and sewage sludge.

**Table 2-4** Energy potential from FOG in the Thames Water catchment

| | Domestic | FSE | Fatberg | STW | Sewage sludge |
|---|---|---|---|---|---|
| **Material potential (tonnes.year$^{-1}$)** | 14,920 | 79,809 | 27,449 | 67,281 | 306,800[1] |
| **Energy potential (GWh.year$^{-1}$)** | 153 | 647 | 141 | 241 | 1,506 |
| **Energy produced from biogas (GWh.year$^{-1}$)** | 30 | 191 | 44 | 84 | 264 |

[1] Reported as ton DS per year

One of the main obstacles to energy generation from some of the FOG wastes studied is collection. Cleaning of sewers and SPSs is either planned or reactive and involves combined vacuum and jetting machines. FOG collected from these tankers would need to be further processed as these systems tend to break them down and mix them with sewage. Whilst equipment seems to be commercially available for FOG collection in SPSs, their efficiency still needs to be demonstrated. In contrast, preliminary treatments can be found at STWs to remove FOG from municipal wastewater; the use of these wastes as co-substrates for anaerobic digestion has been reported by several authors (Girault et al., 2012; Harris et al., 2017; Long et al., 2012; Luostarinen et al., 2009; Silvestre et al., 2011). However, it requires the installation of additional assets contributing to larger operational costs. Another alternative at STWs would be to enhance primary sedimentation tanks in order to increase FOG removal alongside sewage sludge. Further research is needed to assess the performance of such technologies and the economic viability of collecting FOG from FSEs as a robust logistic management would be required to tailor a sustainable disposal route.

## 2.4 Conclusions

The characterisation of selected FOG wastes focused on three main aspects: physicochemical composition, organic macromolecules concentrations and LCFA profiles. The main difference was found in the water content: FOG collected from networks (SPS and sewers) and STW had higher moisture content than FOG collected at source (domestic and FSEs). Predictably, FOG was found to be desirable substrate for anaerobic co-digestion as their high organic matter and lipids content resulted in high methane potential (773-981 STP mL $CH_4$.g $VS_{added}^{-1}$).

The assessment of volumes of FOG collectable indicated FSEs to be the main source with around 67,926 tonnes.$year^{-1}$ (on dry basis) of material relatively easy to collect and potentially available for energy recovery (191 GWh.$year^{-1}$). The anaerobic digestion of FOG wastes, collected either at source or in the networks, could be almost equivalent to the current energy generated from sewage sludge at TWUL sites. In other words, anaerobic co-digestion could help generating around a third of Thames Water's overall electricity consumption. Although FOG from wastewater networks or STWs still have high values for energy recovery, the practicality and feasibility of collecting these wastes could counterbalance the benefits from biogas generation. This further suggested that collection of FOG before it reaches the sewers is highly desirable. Still, volumes and methods of collection should be analysed in order to assess the economic feasibility of developing sustainable schemes.

## 2.5 Acknowledgments

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# 3 Evaluating the potential of domestic fats, oils and greases (FOG) for energy recovery

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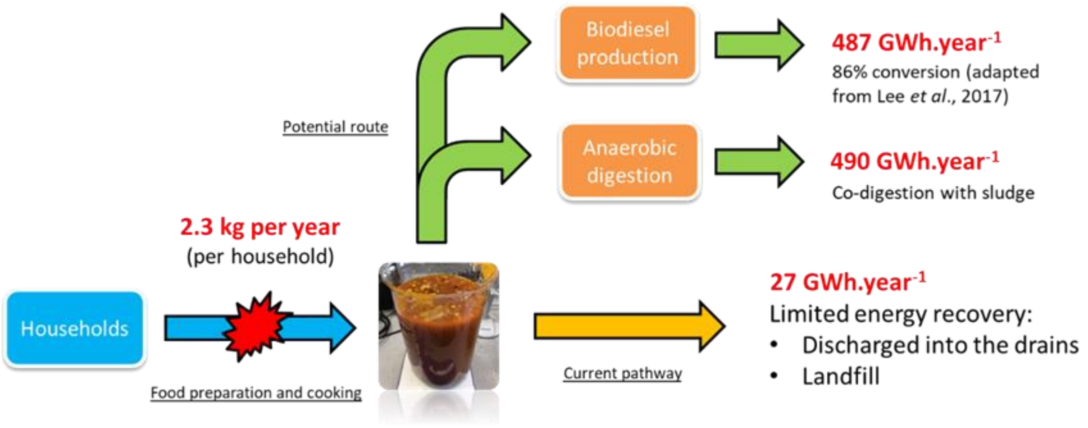
## Graphical abstract



## Abstract

The management and monitoring of fats, oils and greases (FOG) discharges in the sewerage systems have been mostly limited to commercial and industrial sources. Very little is known on emissions at domestic level and the potential of this source for energy recovery. To answer these questions, FOG wastes were collected monthly, for a year, from 31 households in the Thames Water catchment. Around 2.3 kg·year$^{-1}$ of FOG were collected per household (equivalent to 0.8 kg·year$^{-1}$ per capita). These figures were relatively low compared to what was previously reported in other studies but still translated in an annual potential for collection of over 62,380 tonnes of FOG in the UK. Amongst the participating households, disposal in the general waste bin (65% of respondents) and recycling with food waste (19% of respondents) were the most common ways to manage FOG emissions. Household FOG waste showed similar characteristics to used cooking oil collected from food service establishments suggesting its potential as feedstock for biodiesel production. Biomethane yield of household FOG was measured at 875 mL $CH_4$·g $VS_{added}^{-1}$, twice as much that of sewage sludge, measured in this study at 376 mL $CH_4$·g $VS_{added}^{-1}$, making it a desirable substrate for anaerobic digestion. It was thus estimated that energy recovery from household FOG through anaerobic co-digestion or biodiesel production could generate about 490 GWh·year$^{-1}$ in the UK. Whereas landfill or incineration would prove to be less attractive options with potentials estimated at 27 and 126 GWh·year$^{-1}$ respectively. Yet, as the complete capture of FOG is unrealistic, further consideration is required regarding collection modes to maximise recycling rates.

**Keywords**: Anaerobic digestion; biodiesel; fatberg; waste management; household waste

## 3.1 Introduction

Over the last decade, the accumulation of fats, oils and greases (FOG) in sewerage systems has gained awareness across the different stakeholders of the water industry but also with the general public (Engelhaupt, 2017; Moss, 2018). Food outlets, which are believed to be one of the main contributors towards FOG emissions, are accordingly under deeper scrutiny from water authorities. In contrast, domestic customers are often a lower priority for interventions generally achieved through customer awareness and education campaigns (Georges et al., 2017; Wallace et al., 2017). Yet, it is likely that the impact of households might not be negligible especially in densely populated areas such as London. Whilst a few piloted customer campaigns have tried to drive more sustainable behaviours on the domestic end, very little information has been published and often limited to their assumed positive outcomes on sewerage systems (Anglian Water, 2014; Foden et al., 2017; Olleco, 2015; Yorkshire Water, 2015).

As FOG is an energy-rich waste, diverting it from discharges into the drains had the potential to go beyond protecting wastewater assets. To date, research has largely focused on FOG collected from food service establishments (FSEs), demonstrating its potential to enhance biogas generation when used a co-substrate in anaerobic digestion with sewage sludge (Kabouris et al., 2009a; Long et al., 2012) or the organic fraction of municipal solid waste (Grosser et al., 2017; Kumar et al., 2018) and to be converted into biodiesel (Lee et al., 2017). To date, research has largely focused on FOG collected from food outlets. However, the levels of water and free fatty acids (FFAs) in some sources of FOG wastes are known to negatively affect their energy recovery. For instance, water reacts with the catalyst used during biodiesel conversion leading to a more laborious and expensive process (Sanford et al., 2009). High level of acidity in the oil (e.g. presence of FFAs) leads to the formation of soaps during the transesterification process (Saraf and Thomas, 2007), reducing the reaction's yields and increasing the viscosity of the biodiesel mixture (Atadashi et al., 2012). Critically, it is generally admitted that significant problems may occur in the transesterification process when the FFA content is above 3% (Dorado et al., 2002). To address this shortcoming, the most commonly employed technique is an acid esterification with methanol and sulphuric acid (Van Gerpen, 2005). Further to this, long chain fatty acids (LCFAs), which are the most prominent component of FOG wastes, showed toxic effects of LCFA on acetoclastic methanogens in anaerobic digesters (Alves et al., 2009; Palatsi et al., 2010). This can result in their accumulation, causing sludge flotation, digester foaming and blockages of pipes and pumps (Alves et al., 2009). Therefore, understanding the physicochemical properties of household FOG is recommended prior to utilisation.

Ultimately, there is a need for both quantitative and qualitative data from domestic sources to be published for to support the development of a more sustainable waste management strategy. To address this knowledge gap, a one-year trial was developed to collect FOG from 31 households. The collected wastes were first characterised chemically, and then evaluated for their potential as co-substrates for anaerobic digestion with sewage sludge. Using experimental data in conjunction with published literature, a high-level assessment of energy potentials was provided for the UK and put into perspectives with existing regulatory frameworks.

## 3.2 Material and methods

### 3.2.1 Collection

Around 150 TWUL employees were initially emailed and 31 volunteered to take part in the trial. Reusable sealable containers were provided to volunteers and collection was scheduled monthly over a year. Containers were pre-weighted and masses were recorded, monthly, for each volunteer. Each volunteer corresponded to one household. As participation varied over the course of the trial, results were adjusted to reflect volumes based on participation (i.e. total number of months participated) and average volumes collected were reported in $kg.month^{-1}$.

Monthly, the content of each container collected was blended in a 5 L glass beaker. The samples were then heated to 35˚C in order to melt solid fats and, finally, sieved to remove large particulates of food waste. FOG was then stored in a cold room at 4˚C for further analyses.

An initial survey was conducted to determine demographics along with participants' current FOG disposal practices. Six months into the trial, another survey was carried out to investigate people's experience in relation to FOG collection. Both surveys were emailed to participants and then collected either as digital or hard copies (Appendix D). This survey was only conducted to provide insights on FOG sources in households and it is agreed that it might not reflect practices for the wider UK population or elsewhere. FOG disposal routes, gathered from the survey of participants, were compared to information provided by local authorities located within TWUL catchment on their respective websites (accessed in August 2018).

Results from this trial were compared to those gathered during a similar study conducted by TWUL in 2011 (McKinney, 2012). In brief, around 220 households in a residential estate were engaged and domestic FOG was collected from the participants' doorstep monthly over a year. Unfortunately, at this time, no information was gathered on the quality of the FOG collected. Households are not singular identities, and many factors affect food choices (Committee on Examination of the Adequacy of Food Resources and SNAP Allotments, 2013), and in turn FOG generation. Critically, it becomes important to gather additional data on production rates.

The engagement from this study was evaluated at 21%: 31 out of the 150 Thames Water employees emailed expressed interest to take part in the trial. This was relatively similar to the door-to-door collection study where 59 out of the 220 properties targeted (27%) took part at least once in the trial (McKinney, 2012).

### 3.2.2 Physicochemical characterisation

Dry solids (DS) and volatile solids (VS) were determined according to standard methods (APHA, 2005). The major organic constituents were determined by laboratory analyses: fibres by successive digestion with acid and alkali under controlled conditions (Sciantec Analytical, 2018f), proteins as total Kjeldahl nitrogen (Sciantec Analytical, 2018b) and lipids through Weibul acid hydrolysis (Sciantec Analytical, 2018c). Carbohydrates were estimated as the remaining fraction. Theoretical methane yields were calculated from these organic constituents considering that carbohydrates, proteins and lipids respectively yield 415, 496 and 1,014 mL $CH_4.gVS^{-1}$ at standard conditions of temperature and pressure (STP) (Angelidaki and Sanders, 2004).

Methylated fatty acids profiles were obtained by gas-liquid chromatography using a free fatty acid (FFA) phase column of dimensions 25m x 0.20mm ID and detection by flame ionisation detector. Fats and oils were trans-esterified to fatty acid methyl esters (FAME) by heating under reflux for two hours with a mixture of methanol and sulfuric acid in toluene. The resulting methyl esters were extracted using a small volume of n-hexane. The n-hexane solution was dried using anhydrous sodium sulphate and then transferred to a chromatography vial (Sciantec Analytical, 2018d).

Peroxide, saponification and acid values were respectively determined in accordance with methods AOCS Cd 8-53, EN ISO 6293 and EN 14104. The ester value was calculated by subtracting the acid value from the saponification value. The percentage of FFAs, in terms of oleic acid, was calculated from the acid value as:

$$FFA\ (\%) = \frac{Acid\ value}{56.1} \times 28.2 \tag{3-1}$$

Gross calorific values were determined experimentally using a calorimeter (Parr model 6100) equipped with a 1108CL oxygen bomb. Samples were freeze dried beforehand (Sciantec Analytical, 2018e). The lower heating values (LHV) were estimated from the measurement of calorific values by subtracting the heat of vaporisation of water in the products as follows:

$$LHV_d = HHV_d \times (1 - M) - H_V \times\ M \tag{3-2}$$

Where $M$ is the moisture content, $Hv$ is the latent heat of vaporisation of water estimated at 2.447 MJ.kg$^{-1}$ at 25˚C and $HHV_d$ is the gross heating value in MJ.kg$^{-1}$ on dry basis determined as follows:

$$HHV_d = \frac{HHV}{1 - M} \tag{3-3}$$

Where $HHV$ is the measured HHV on wet basis.

### 3.2.3 Anaerobic batch testing

Digested sludge, serving as inoculum, was sampled from a full-scale anaerobic digester treating municipal sewage. This plant was using the Cambi thermal hydrolysis process to pre-treat sewage sludge prior to anaerobic digestion. Sewage sludge samples were obtained from the same site after the thermo-hydrolysis treatment step (Table 3-1).

**Table 3-1** Inoculum and substrate characteristics.

| **Parameters** | **Inoculum** | **Sludge** | **FOG** |
|---|---|---|---|
| DS (%) | 4.8±0.1 | 9.7±1.6 | 81.5±12.9 |
| VS (% DS) | 61.7±1.0 | 76.7±0.6 | 99.4±0.7 |
| Lipids (% DS) | 6.7±0.5 | 12.8±2.4 | 94.3±6.6 |

Triplicate batch testing was conducted using automatic methane potential test systems (AMPTS) II at mesophilic temperatures (39˚C). A ratio of 2:1 $VS_{inoculum}$:$VS_{substrate}$ was used for this trial (Nazaitulshila et al., 2015). Batch testing was conducted in 1 L glass bottles continuously stirred with a dedicated stirrer and each bottle was connected to a $CO_2$ stripping solution with a pH indicator to show solution saturation. Reactors only containing the inoculum were operated to take into account any endogenous biomethane production. Combinations of FOG and sludge were digested at different substrate ratios with identical feed concentrations of 8.1 g VS (Table 3-2). At the end of each experiment, DS and VS were measured to evaluate the VS destruction. Organic macromolecules were analysed from the digested samples as described in section 3.2.2.

**Table 3-2** Initial amount of inoculum, FOG and sludge (expressed as g VS) added to the reactors.

| **FOG concentrations (%)** | **0** | **10** | **20** | **30** | **40** | **50** | **60** | **70** | **80** | **90** | **100** |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Inoculum (g VS) | 16.2 | 16.2 | 16.2 | 16.2 | 16.2 | 16.2 | 16.2 | 16.2 | 16.2 | 16.2 | 16.2 |
| FOG (g VS) | 0.0 | 0.8 | 1.6 | 2.4 | 3.2 | 4.0 | 4.9 | 5.7 | 6.5 | 7.3 | 8.1 |
| Sludge (g VS) | 8.1 | 7.3 | 6.5 | 5.7 | 4.9 | 4.0 | 3.2 | 2.4 | 1.6 | 0.8 | 0.0 |

Methane production rates were adjusted to STP as follows:

$$V_{STP} = \left(1 - \frac{P_{vap}}{P_{gas}}\right) \times \frac{P_{gas}}{P_{STP}} \times \frac{T_{STP}}{T_{gas}} \times V_{gas} \qquad \textbf{(3-4)}$$

Where $V_{STP}$ is the volume adjusted to STP, $P_{STP}$ is the standard pressure (101.3 kPa), $T_{gas}$ is the temperature of the measured gas (311 K), $T_{STP}$ is the standard temperature (273 K) and $V_{gas}$ is the measured volume of gas. $P_{gas}$ was calculated as the sum of the partial pressures of methane and carbon dioxide; $P_{CO2}$ was neglected in this case as carbon dioxide was removed through the stripping solution. $P_{vap}$ is the water vapour pressure calculated as follows:

$$P_{vap} = 10^{8.1962 - \frac{1{,}730.63}{T_{gas} - 39.724}} \tag{3-5}$$

### 3.2.4 Energy potential

Four energy recovery routes were considered: (1) landfilling, (2) incineration, (3) conversion to biodiesel (3) and anaerobic digestion (4). In landfills, methane is produced as a by-product of the degradation of organic wastes with reported production rates of 43 $m^3$ per ton (Themelis and Ulloa, 2007). The calorific value of biomethane was 36 $MJ.m^{-3}$ whilst generation efficiency of electricity was assumed at 30% from combined heat and power plants (Goss et al., 2017). Energy generation from incineration was calculated from the lower heating value of domestic FOG assuming a 20% conversion efficiency (CIWEM, n.d.). Experimental methane generation and biodiesel conversion yields were used to calculate energy potentials from anaerobic digestion and biodiesel production. The equivalent of 1 $m^3$ of biodiesel was 0.78 ton of oil equivalent (toe) further corresponding to an energetic value of 11.6 $MWh.toe^{-1}$ (Eurostat, 2018).

## 3.3 Results and discussion

### 3.3.1 Domestic survey and FOG production rates

The survey of domestic FOG generation revealed that all the respondents predominately used vegetable oil and in particular olive oil (73% of the respondents). In addition, 77% also used animal fat, predominately in the form of butter. The collected oil was from either oil residues from pans and plates or fats from cooked meats associated to 82% and 55% of respondents with an additional source coming from used food jars. Comparison to the previous survey revealed a shift in cooking practice as the previous surveyed identified the main cooking practices as deep fat frying (48%), shallow frying (2%), bhajee frying (2%) and wok frying (5%). In both cases vegetable oils were identified as the main FOG source which is consistent with practice in FSEs (Envirowise, 2008).

Production rates, from the 31 households monitored, ranged from 0.01 to 0.53 kg.month$^{-1}$ with an average value of 0.19 kg.month$^{-1}$ per household (Figure 3-1). This is lower than observed during the previous survey where a wider range of values were recorded between 0.01 up to 6.88 kg.month$^{-1}$ per household. In addition, rates higher than 1.0 kg.month$^{-1}$ were measured for 11 households whereas all households in this trial produced less than 0.60 kg.month$^{-1}$. Other reported studies are congruent with the current one indicating an overall reduction in FOG generation per household. For instance, a recent UK survey estimated FOG generation rates of 0.22 kg.month$^{-1}$ per month (Quested et al., 2013) although this was reassessed to be within a range of 0.05 to 0.17 kg.month$^{-1}$ per household (Gelder and Grist, 2015). In Canada, the Capital Regional District in British Columbia estimated FOG from domestic sources at 0.47 kg.month$^{-1}$ per household (Blanc and Arthur, 2013).

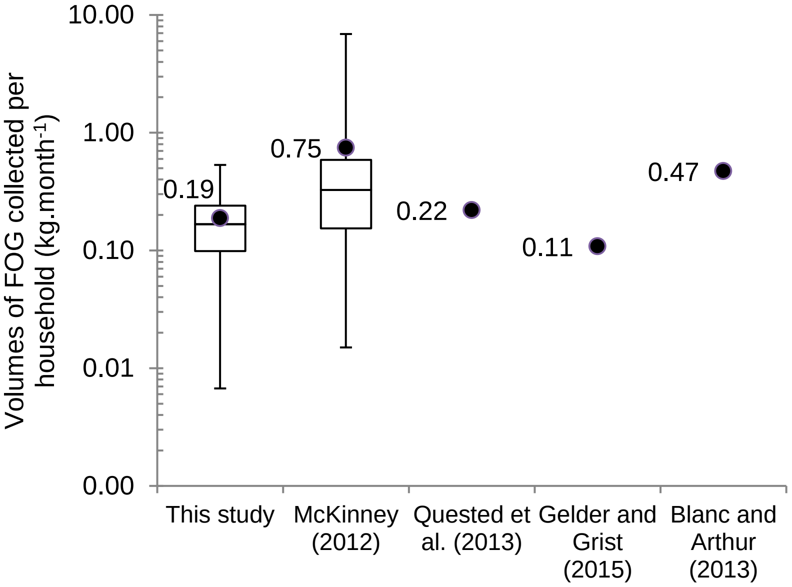


**Figure 3-1** Volumes of FOG collected from this trial compared to estimates available in literature. The line in the middle of each box represents the median, the upper half of the box represents the third quartile and the lower one the 2nd quartile. The error bars represent the minimum and maximum. Black markers represent the averages.

In estimating FOG production rates from households, diversity within and between households needs to be appreciated. Households are not singular identities and food preparation, and in turn FOG generation, is strongly impacted by a variety of factors including number of occupants as well as social and cultural factors (Committee on Examination of the Adequacy of Food Resources and SNAP Allotments, 2013). One notable difference between both studies was household sizes measured at 2.7 and 4.4 occupants respectively for this study and McKinney (2012). However, volumes normalised based on occupancy in this study were still found lower, at 0.07 kg.month$^{-1}$ per capita, compared to McKinney (2012), at 0.17 kg.month$^{-1}$, suggesting other factors influencing FOG production.

Current disposal routes amongst respondents were further investigated. Disposing of the FOG in the general waste bin was identified as the most common route, representing 65% of the respondents (Figure 3-2a). A further 19% recycled the FOG with the food waste and 3% into fat traps with 13% stating that they did not have a way to dispose of FOG (i.e. discharges into the drains). UK water companies in all provide advice on FOG management in order to limit disposal down the sewer and generally encourage putting into the bin (Severn Trent Connect, 2019; Thames Water Utilities, 2016). Further, they recommend customers follow the advice of their local council. However, a survey of 102 local authorities revealed that 25% did not provide any guidance on their website. Where available the predominant FOG collection routes advised by the local councils were collection point at their household waste recycling centre (HWRC) (56%), food waste collection (15%) and into the waste bin (9%) (Figure 3-2b). Interestingly, none of the participants in this study and only a few respondents to the survey undertaken in 2011 were disposing of their FOG this way. Foden et al. (2017) suggested that the unsuccessfulness of this approach was likely due to its poor fit with the rhythm of households' everyday life. In relation to collection in food waste, 72% of the surveyed councils had kerbside food waste collection scheme but only 21% accepted FOG in caddies, mainly in the form of solid fats and in small volumes.

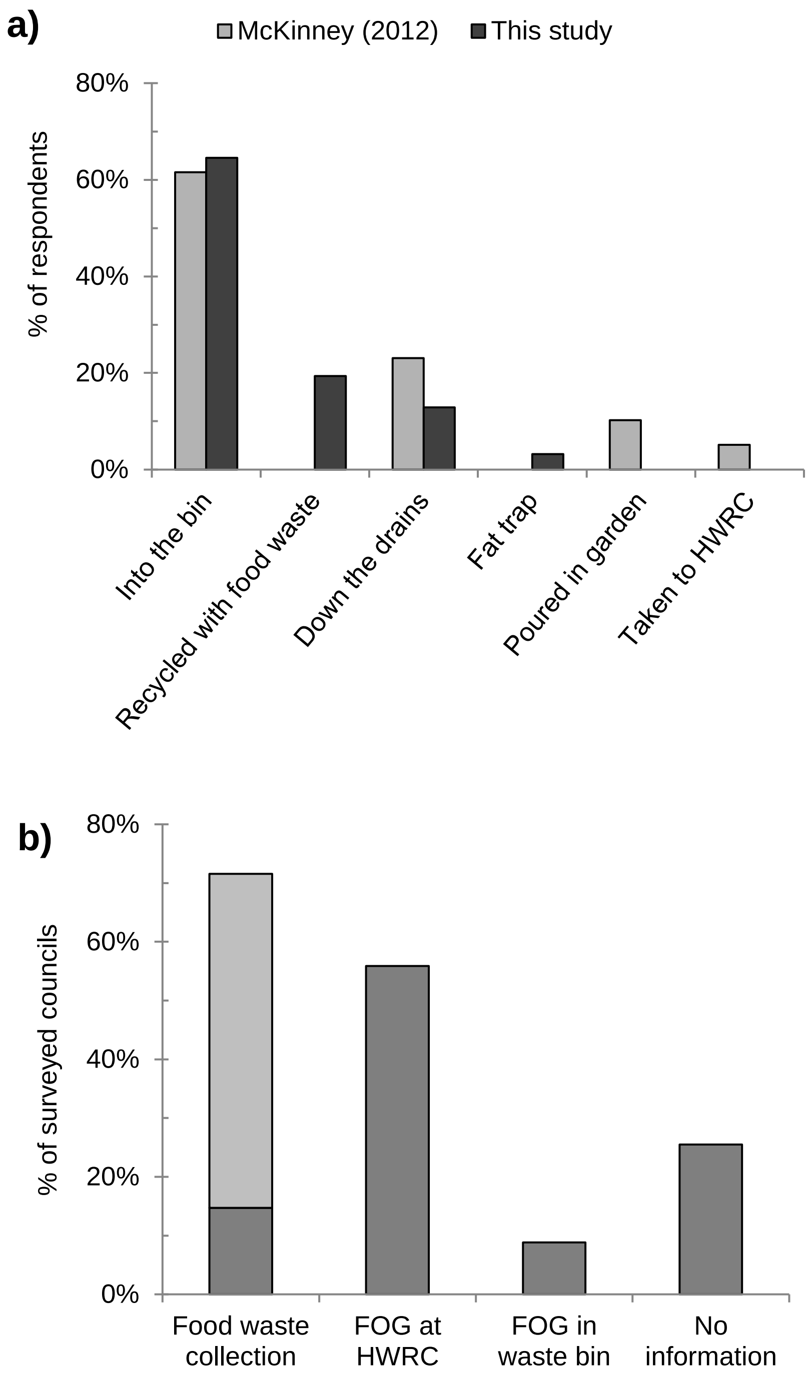


**Figure 3-2** Disposal routes as reported by survey respondents (a) and suggested by local authorities (b). Information from local authorities was obtained from the websites of 102 councils located within TWUL catchment.

### 3.3.2 Physicochemical characterisation

The collected FOG had physical properties that were observed to be relatively different between households, ranging from yellow to light brown colours and either liquid or semi-solid states at room temperature (Figure C-1).

The water content of the blended domestic FOG was on average 4.2±2.3% (Table 3-3), being slightly higher than that reported for UCOs generated from the food industry, ranging between 0.04-1.4% with an average of 0.2% (Cheah et al., 2016; Hailei and Hui, 2014; Sanford et al., 2009; Sanli et al., 2011; Supple et al., 2002). By contrast, FOG collected from grease separators in food outlets were associated with much higher and variable moisture content, depending upon the type of separator installed, ranging between 0.1-91.3% (Canakci, 2007; Karnasuta et al., 2007; Long et al., 2012; Miot et al., 2013, chapter 5).

**Table 3-3** Physicochemical parameters of FOG collected at source. Average values are presented alongside minimum and maximum. Values generated for domestic FOG were obtained from this study whilst those for UCOs and FOG from FSEs were found in literature.

| **FOG waste** | **LHV ($MJ.kg^{-1}$)** | **Water content (%)** | **FFA (%)** | **Peroxide value (meq $H_2O_2.kg^{-1}$)** | **Ester (%)** |
|---|---|---|---|---|---|
| UCO | 39.3<br>(37.2 – 40.3) | 0.2<br>(0.04 – 1.4) | 1.4<br>(0.1 – 8.9) | 40.8<br>(0.5 – 200.4) | 99.0<br>(90.1 – 99.9) |
| FSE FOG | 35.4<br>(24.5 – 41.6) | 18.6<br>(0.1 – 91.3) | 34.3<br>(0.7 – 97.8) | 6.7<br>(0.2 – 52.1) | |
| Domestic FOG | 38.2<br>(36.3 – 39.3) | 4.2<br>(1.5 – 8.3) | 2.7<br>(2.3 – 3.1) | 23.9<br>(12.7 – 31.1) | 96.1<br>(92.9 – 97.7) |

Nearly 100% of the solids in domestic FOG were volatile. Lipids and carbohydrates accounted for most of the organics measured with concentrations respectively found at 94.3±6.6% DS and 5.7±6.1% DS (Table 3-1). This further translated into high LHV measured on average at 38.2±1.4 $MJ.kg^{-1}$ on dry basis (Table 3-3). In comparison reported values for UCOs are slightly higher LHV, typically ranging between 37.2 and 40.3 $MJ.kg^{-1}$ (Ortner et al., 2016; Sanli et al., 2011; Supple et al., 2002) whilst FOG from grease separator was between 24.5

and 41.6 MJ.kg$^{-1}$ (Chapter 5). Ultimately, with relatively low water content and lipid-rich composition, domestic FOG represents a valuable energy source which has the potential to be converted into biogas or biodiesel.

It is important to understand the physicochemical properties of these wastes as parameters including water and FFA can hinder the viability of the process. The collected FOG had FFA content similar to UCOs, with FFA levels measured at 2.7±0.3%. UCOs typically contain between 0.1-9.0% FFAs and are considered a good biodiesel feedstock (Berrios et al., 2010; Cheah et al., 2016; Sanford et al., 2009; Sanli, Canakci and Alptekin, 2011). By contrast, FFA concentrations are higher in FOG collected from grease separators, ranging between 0.7-97.8% with a median value of 34.3% (Canakci, 2007; Karnasuta et al., 2007; Kobayashi, Kuramochi and Xu, 2016).

### 3.3.3 Anaerobic batch testing

The collected and blended FOG wastes were used as feedstock, alone or co-digested with sludge, in batch reactors, to assess their potential for biomethane generation. The biomethane yield of FOG was measured at 875±108 STP mL $CH_4$.g $VS_{added}^{-1}$, twice as much that of sewage sludge, measured at 376±32 STP mL $CH_4$.g $VS_{added}^{-1}$. Biomethane yields for domestic FOG were found in relatively good agreement with theoretical estimates calculated at 974±44 mL $CH_4$.g $VS^{-1}$ (Table 3-4). Yet, these experimental yields were found lower than those measured for FSE FOG at 938±39 STP mL $CH_4$.g $VS_{added}^{-1}$ (Chapter 2) and 993 STP mL $CH_4$.g $VS_{added}^{-1}$ (Kabouris et al., 2009a) but relatively comparable to those reported in Chapter 5 at 872±148 STP mL $CH_4$.g $VS_{added}^{-1}$. Lipids degradation rates were also found to be lower for domestic FOG compared to FSE FOG, respectively at 87±0.3% and 94±8%. Critically, the degradation of lipids from domestic FOG produced less methane at 840±61 STP mL $CH_4$.g $lipids_{destroyed}^{-1}$.

**Table 3-4** Results from the batch testing of sewage sludge and domestic FOG.

| **Parameter** | **Sewage sludge** | **FOG** |
|---|---|---|
| Experimental methane yield (STP mL.g $VS_{added}^{-1}$) | 376±32 | 875±108 |
| Theoretical methane yield (STP mL.g $VS^{-1}$) | | 974±44 |
| VS destruction (%) | 57±7 | 87±11 |
| Lipids destruction (%) | 35±4 | 87±0.3 |
| Experimental methane yield (STP mL.g $VS_{destroyed}^{-1}$) | 645±141 | 942±36 |

The results from reactors digesting mixtures of FOG and sewage sludge in different concentrations showed the methane potential was increased with increasing amount of domestic FOG (Figure 3-3a). Similar results were obtained by Davidsson et al. (2008) co-digesting grease trap waste collected from FSEs. Reactors only digesting sludge exhibited the lowest lipids degradation measured at 35±4% (Figure 3-3b). As a benchmark, lipids degradation in full-scale anaerobic systems generally vary from 20 to 70% (Liu, 2018). As more FOG over sludge were added to the reactors, the lipids degradation rates increased suggesting a good degradation of lipids contained in domestic FOG. The maximum degradation rate was reached with FOG only and was measured at 87±0.3% suggesting not all the lipids were degradable in these conditions further posited to be caused by LCFAs known to inhibit acetoclastic methanogens and, in turn, biogas generation (Alves et al., 2009).

The five most common LCFAs measured in domestic FOG were: oleic (C18:1), linoleic (C18:2), palmitic (C16:0), stearic (C18:0) and linolenic acids (C18:3) with respective concentrations of 41.4±10.3%, 31.5±9.1%, 12.2±1.7%, 4.8±1.0% and 3.8±3.2% of total fatty acids (Table 3-5). Data on the toxicity of LCFAs, reported as the concentration causing a 50% relative activity loss of the specific methane production, has been published in literature by several authors (Alves et al., 2001; Lalman and Bagley, 2000; Pereira et al., 2005; Prinst et al., 1972; Shin et al., 2003). Using these values, inhibitory loadings were calculated for the main LCFAs measured in domestic FOG (Table 3-5) thus identifying oleic and linoleic acids with inhibitory concentrations to the anaerobic digestion process with loadings as low as 0.1 g VS. Consequently, without a proper feeding strategy, the addition of FOG to anaerobic digesters could become an operational risk if the accumulation of LCFAs is not prevented.

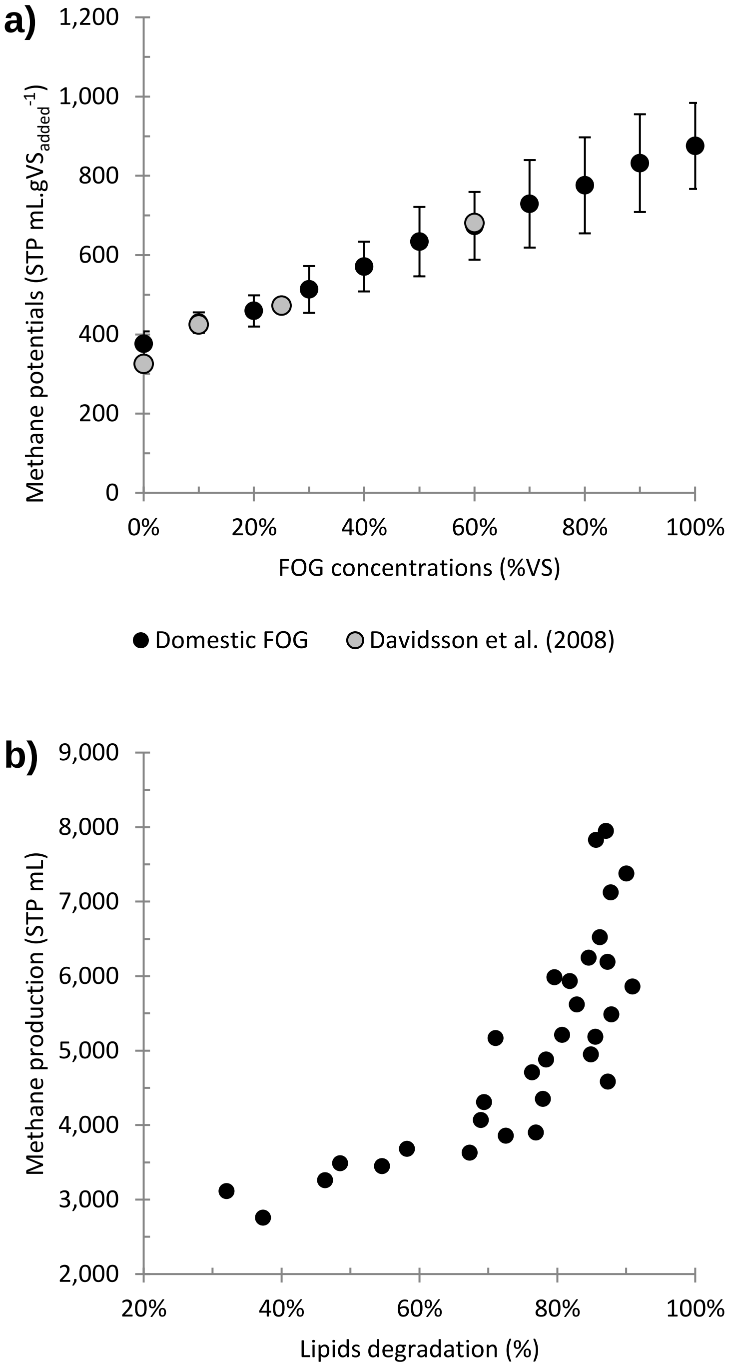


**Figure 3-3** Cumulative biomethane production reported against FOG concentrations (expressed as % VS added) (a) and lipids removal rates (b).

**Table 3-5** Five most common LCFAs measured in domestic FOG and their associated $IC_{50}$. The inhibitory loadings were calculated from the characterisation of FOG.

| LCFA | Concentration in FOG (% total fatty acid) | $IC_{50}$ ($mg.L^{-1}$) | Inhibitory loading (g VS) |
|---|---|---|---|
| **C16:0** | 12.2±1.7 | 1,100 (Pereira et al., 2005) | 9.3 |
| **C18:0** | 4.8±1.0 | 1,500 (Shin et al., 2003) | 32.4 |
| **C18:1** | 41.4±10.3 | 50 (Alves et al., 2001) | 0.1 |
| **C18:2** | 31.5±9.1 | 30 (Lalman and Bagley, 2000) | 0.1 |
| **C18:3** | 3.8±3.2 | 500 (Prinst, Van Nevel and Demeyer, 1972) | 13.4 |

### 3.3.4 Perspectives for the recovery of FOG

With 27.6M households in the UK, it is estimated that there is the potential to collect 62,380 tonnes of FOG annually. The London region only would account for 23% of this volume, equating to around 14,240 tonnes of FOG per year. In comparison, for the same catchment, it was previously estimated that 79,810 tonnes of FOG were produced annually from FSEs (Chapter 2). The comparison of the energy potential that the domestic FOG could yield reveals similar levels when processed as a co-substrate in anaerobic digestion or biodiesel production (assuming conversion yields at 86% (Lee et al., 2017)), at yields of 490 $GWh\cdot year^{-1}$ and 487 $GWh\cdot year^{-1}$ respectively. In other words, this would represent 22% of the estimated 2,220 $GWh\cdot year^{-1}$ generated in the UK from sewage sludge (Mills, 2015). These potential yields greatly exceed the equivalent levels achievable from landfill or incineration at 27 and 126 $GWh\cdot year^{-1}$ respectively. In assuming a conversion efficiency of 30% to generate electricity from biogas (Goss et al., 2017), using biogas generated from anaerobic digestion or biodiesel produced from household FOG in combined heat and power engines would power 35,230 houses (considering an average domestic energy consumption of 4.2 $MWh\cdot year^{-1}$ (UK Department of Energy & Climate Change, 2014)). In comparison, using FOG in waste-to-energy plants would generate enough power to supply 30,340 houses.

A common challenge, irrespective of processing preference, is the collection of the material and hence what proportion of the total estimate is practical. This depends on collection mode be it door-to-door or centralised collection schemes. Maximum collection rates have been hypothesised to occur from kerbside collection which also presents a better fit with household routines (Seyring et al., 2015). However, the logistical, financial and environmental implications of such a service (Foden et al., 2017) mean that co-collection with food wastes appears a more practical option. Currently, only a small percentage of local councils providing a kerbside food waste collection were accepting FOG due to difficulties in handling FOG in food waste caddies (Figure 3-2b). However, assuming that food waste is produced at a rate of 22 kg·month$^{-1}$·household$^{-1}$ (Quested and Parry, 2016), FOG would typically represent less than 1% of this volume and so should not cause any difficulties. With 61% of UK local authorities collecting food waste from households and estimated participation rates of 45% (WRAP, 2016a), around 17,120 tonnes of FOG could be recovered though co-collection (equating to 27% of the total volume generated nationwide). The alternative is *bring schemes* where the FOG is collected in local drop of points (Seyring et al., 2015) with illustration of such approaches in some US municipalities including co-development with retailers to improve the fit of these methods with household's routine (City of Dallas, 2019). Assessing participation rates for *bring schemes* is a difficult exercise, nevertheless a study published on food waste collection from flats using *bring schemes* estimated participation rates of 14% (WRAP, 2016b). Ultimately, assuming similar rates, around 8,730 tonnes of FOG would be collectable annually from *bring schemes*.

The collection approach adopted will impact the potential downstream processing routes. Co-collection with source-segregated food waste directs preference towards anaerobic digestion. In contrast, segregated FOG collection enables high yield routes to be used. Such collection could be processed in either food waste or municipal sewage digesters. In the case of the latter, current regulations in some countries, such as the UK, means that inclusion of FOG into the digestor changes the regulatory regime such that the co-digestate produced is still a waste under the revised Waste Framework Directive requiring potential expensive permitting for its disposal to land or treatment (Iacovidou et al., 2012). As such this favours the use of collected food in purpose food waste digesters. In contrast, no such barriers exist for inclusion of collected FOG for biodiesel conversion with full scale facilities already operating within the UK (UK Department for Transport, 2019). This is supported by existing regulatory drivers encouraging the production of biofuels such as the Renewable Transport Fuel Obligation. The challenges then become ones of source quality and the financial impacts of the collection system.

## 3.4 Conclusions

This study showed that on average 2.3 kg of FOG per year could be collected from every household (0.19 $kg.month^{-1}$). In the UK, the amount of household FOG potentially collectable would represent 1% of the total food waste arising nationwide from households, equating to 62,830 tonnes annually.

The physicochemical characterisation of household FOG revealed water and FFA contents of 4.2±2.3% and 2.7±0.3% respectively, suggesting additional pre-treatment might be required for biodiesel production. Whilst, these wastes also demonstrated high methane potential, measured at 875 mL $CH_4.g\ VS_{added}^{-1}$, high concentrations of potentially inhibitory LCFAs, such as oleic and linoleic acids, might require further attention to determine the process safe boundaries.

Recovering energy from FOG through biodiesel conversion or anaerobic co-digestion was estimated with the potential to generate 487 and 490 $GWh.year^{-1}$ respectively, in the UK. Co-collection with food waste was suggested as one of the potential options to maximise penetration rates. However, this will require understanding stakeholders' drivers and potential barriers to implementing sustainable schemes.

## 3.5 Acknowledgments

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# 4 Towards a risk register for improved management of fats, oils and greases (FOG) discharges from food outlets

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## Abstract

Whilst food service establishments (FSEs) are a significant source of fats, oils and greases (FOG) discharges into sewer networks, the understanding of FOG pathways in commercial kitchens is relatively poor. Previous studies have too often been limited to reporting the paucity of FOG mitigation techniques within commercial catering and food businesses without trying to understand how and where FOG generation occurs and what measures are undertaken to mitigate its impact on sewers. In this contribution we extend understanding of how FOG is perceived and managed by those working within FSEs. A questionnaire survey of FSEs was used to expose awareness of and experiences with FOG as well as characterise two important behaviours; the nature and frequency of kitchen appliance cleaning regimes, and waste management practices (n = 107). Findings demonstrate that the awareness of issues caused by FOG in sewerage systems is independent of job role or position held and that a majority of respondents (74%) were acquainted with the potential impacts of poor FOG management. Application of a risk register approach revealed a relatively low risk of emissions from waste frying oils and exposed a number of behaviours which can serve to reduce FOG emission potential including pre-rinsing of plates and cleaning of fryers and extraction hoods. Critically, 69% of FSEs had no means to manage their FOG emissions, thereby risking the accumulation of FOG in sewer lines. Findings lead us to conclude that sampled FSEs were generally unaware of the contribution of the various sources of FOG, limiting their responses to the recycling of waste frying oils. Growing concern about the impacts of poor FOG management in FSEs is driving a need for improved communication and education to improve perception of the problem. The risk register developed in this paper could be used to suggest efforts to reduce and mitigate FOG emissions from FSEs.

**Keywords:** Behaviours; sewer deposits; food service establishments

## 4.1 Introduction

Over the last decade, uncontrolled discharges of fats, oils and greases (FOG) from food service establishments (FSEs) have attracted increased attention from both water infrastructure operators concerned about the obstruction of sewer flows and the general public as high profile sewer blockages appear in media headlines (Engelhaupt, 2017; Moss, 2018). Once allowed to solidify and/or deposit in sewer lines, such discharges tend to form large assemblages (often called fatbergs) thereby reducing a sewer's effective capacity and leading, in some cases, to sewer flooding (He et al., 2017). Uncontrolled discharges of FOG will inevitably put sewerage systems under increasing pressure. Changing dining habits which see people eating out more frequently (Paddock et al., 2017) and projected population growth (Office for National Statistics, 2017a), are both driving significant increases in the number of FSEs. As a result, water companies are having to deploy additional resources to manage FOG problems with annual spend in the UK, for example, being between £15M and £50M (Williams et al., 2012).

Infrastructure operators are becoming increasingly interventionist in their attempts to reduce the downstream impacts of FOG. Waste minimisation can be achieved at source through the promotion of good kitchen management practices in combination with on-site remediation techniques. Over recent years, in an attempt to reduce the number of sewer blockages, UK water service providers have taken a more aggressive approach towards FSEs under Section 111 of the Water Industry Act 1991 (UK Parliament, 1991), leading in some cases to prosecution (Brockett, 2016; Hackett, 2018) and penalties. With multiple potential FOG streams present in a commercial kitchen, the need to treat effluents before their release into the drains becomes critical if sewer blockages are to be minimised. This is achieved either through biological and/or physical means (Wallace et al., 2017). Biological remediation aims at degrading FOG from kitchen effluents or from agglomerated deposits in drainage systems using either enzymes or microorganisms; whilst physical separation involves the capture of grease before it reaches the sewers. In the UK, the Building Regulations require any commercial kitchens serving hot food to be fitted with a grease separator complying with the appropriate British Standard or any “other effective means of grease removal” (HM Government, 2002).

However, the understanding of FOG generation and disposal pathways in kitchens is still relatively poor and too often limited to reporting the paucity of FOG mitigation techniques within commercial catering and food preparation establishments (ECAS, 2016; Thames Water Utilities, 2018) without trying to understand how and where FOG generation occurs and what measures are undertaken to mitigate its impact on sewers. Below, we directly address this knowledge gap by extending the understanding of how FOG is perceived and managed by those working within FSEs. A questionnaire survey (n = 107) of FSE operators was used to expose awareness of and experiences with FOG as well as characterise two important behaviours: the nature and frequency of kitchen appliance cleaning regimes, and waste management practices. Using information gathered during this survey, a risk register was developed identifying activities contributing to FOG emissions in order to prioritise future efforts to reduce and mitigate FOG discharges.

## 4.2 Methods

A list of FSEs was obtained from the UK Food Standards Agency (Food Standards Agency, 2016) and used to identify and access commercial kitchens in two small towns. A total of 107 FSEs agreed to participate in the study. The sample was comprised of: 51% restaurants (full meal), 14% cafés, 14% institutional food services (e.g. schools and nursing homes), 13% pubs and 7% fast food outlets (i.e. takeaways). Around 37% of the sampled FSEs were part of a chain whilst 53% were private businesses. Schools, accounting for 9% of the sample, fell into a third category in terms of ownership as the kitchen premises belonged to the school but food preparation was carried out by a third party.

A semi-structured questionnaire-based survey tool was developed to gather information on how FOG is perceived by FSE operators and the contribution of cleaning regimes to FOG-related problems. The questionnaire was divided into five sections: (1) characteristics of the FSE, (2) kitchen equipment and cleaning regime, (3) food waste and used cooking oils (UCOs) disposal regimes, (4) means of FOG prevention and (5) knowledge of FOG problematics. The semi-structured researcher-administered questionnaires were conducted with FSE employees (one from each establishment), each lasting between 10 and 15 minutes. Participant selection relied on access to the most knowledgeable individual present at the establishment at the time of access. Those individuals questioned at each establishment came from a number of job roles and functions: owners (25%), facility or restaurant managers (39%) and kitchen staff (36%).

Collected data was recorded in Microsoft Excel and frequency analysis used to report the relative significance of respondent beliefs, understandings, and behaviours. Results were reported using quantified terms. Cross tabulation was used to analyse combinations of parameters. To understand the potential contribution of kitchen equipment to FOG discharges, interviewees were asked about the method and frequency of cleaning of their equipment. Each appliance, based on respondents' inputs, was evaluated for its likelihood to contribute to the FOG problem. Thus, any activity involving the discharge of grease rich-waters into the drains was considered as a high risk.

## 4.3 Results

Overall, 74% of survey respondents were acquainted with the consequences of FOG discharges in sewerage systems (i.e. sewer blockages). Establishment owners were found to be slightly more likely to be aware (85% of the group) than managers (71%) or kitchen staff (68%). A common theme in conversations with respondents was their reference to cooking oils when asked about their knowledge about FOG. In commercial kitchens, these oils are mostly used for deep fat frying activities. Amongst the surveyed premises, purchased cooking oil volumes ranged between 1 L per week and 200 L per week, with a median value of 30 L per week. Encouragingly, 82% of the surveyed establishments were recycling their UCOs whilst 3% of premises were disposing of small volumes into the general waste bin and 7% did not use any cooking oil. The remaining 8% of respondents, whilst using small volumes of cooking oil, were not recycling them potentially allowing UCOs to reach drainage systems. Volumes of UCOs generated by the establishments ranged from zero up to 200 L per week, with a median value of 20 L per week. With significant volumes of UCOs being generated, it is understandable why they are identified as the most prominent source of FOG by FSE operators. In addition, it is worth noting that UCOs have a well demonstrated economic incentive to recycle the product through the mechanism of UCO collectors offering a rebate or discount on the supply of fresh oil.

A well-recognised contributing behaviour to FOG discharges is dishwashing which, in many establishments, is linked to the practicalities of food serving (Garza et al., 2005). Typically, FSEs serve food either in washable or disposable dishes with the latter often associated with the fast food industry. Amongst the respondents to this survey, 68% were using reusable plates whilst 7% used disposable material exclusively and 24% used a combination of both. Where crockery cleaning was undertaken, FSEs either hand washed (22% of the respondents) or relied on automated dishwashing equipment (76%). For 50% of the respondents, washing up was conducted using both a pre-rinse arm and dishwasher (Table 4-1). 24% of the respondents did not mention using neither pot scrubbers nor pre-rinse arms whilst using a dishwasher, therefore possibly relying on basic hand-washing dishes or loading directly into the dishwasher. Interestingly, Gurd (2018) reported significant differences in the effluent composition of pre-rinse sinks and dishwashers. Whilst both effluents had similar concentrations in terms of emulsified FOG, more free FOG was observed in sink samples such as the emulsified fraction represented 42±16% of total FOG compared to 94±9% in dishwasher effluents. Ultimately, the different pot washing methods used by FSE operators will have a direct impact on the type of effluent discharged and on the effectiveness of remediation techniques.

**Table 4-1** How is washing up being conducted by FSE operators.

| | Use of pre-rinse arms or pot scrubbers | | |
|---|---|---|---|
| **Use of dishwasher** | Yes | No | Unknown |
| Yes | 54 (50%) | 26 (24%) | 1 (1%) |
| No | 6 (6%) | 18 (17%) | |
| Unknown | | | 2 (2%) |

Although dishwashing is one of the largest contributors of FOG emissions from commercial kitchens, appliances also contribute to the release of FOG into drainage networks. Amongst the surveyed establishments, the use of extraction hoods (94% of establishments), conventional ovens (79%), and fryers (76%) predominated with grills (50%) and combination ovens (34%) also relatively common. Based on the information provided by respondents, around 86% of establishments with combination ovens had cleaning regimes possibly contributing to FOG discharges as most of them had a steam cleaning cycle whose effluent is typically discharged straight to the drains. A second commercial appliance with significant FOG discharge potential is extraction hoods, possessed by 70% of surveyed establishments. These ventilation systems are designed to extract heat, FOG and other vapour emissions generated within the kitchen. Filters are fitted to prevent FOG from entering the ventilation system. Respondents routinely cleaned these filters in kitchen sinks or outside of their premises, thus discharging the accumulated FOG directly into drainage systems. A similar proportion of establishments (70%) operating deep-fat fryers also discharged potentially harmful effluents to the drains as a result of appliance cleaning. In most cases, wasted frying oils were drained, and then fryers were filled with water and/or cleaning products. It was common for respondents to discharge these effluents directly into the drains. Furthermore, fryer baskets were commonly cleaned using dishwasher appliances. Cleaning practices for conventional ovens and grills were far less risky with only 25% and 36% of establishments respectively adopting behaviours which resulted in FOG discharges to drain. Good practices for grills included using aluminium foil to prevent grease build-up or wiping surfaces with dry towels to remove any debris. Cleaning of conventional ovens was typically achieved either by dry wiping internal surfaces and/or using detergents with water, further disposed into the drains. These results clearly indicate that there are several grease contamination points in kitchens which require improved management in order to avoid FOG accumulation in sewers.

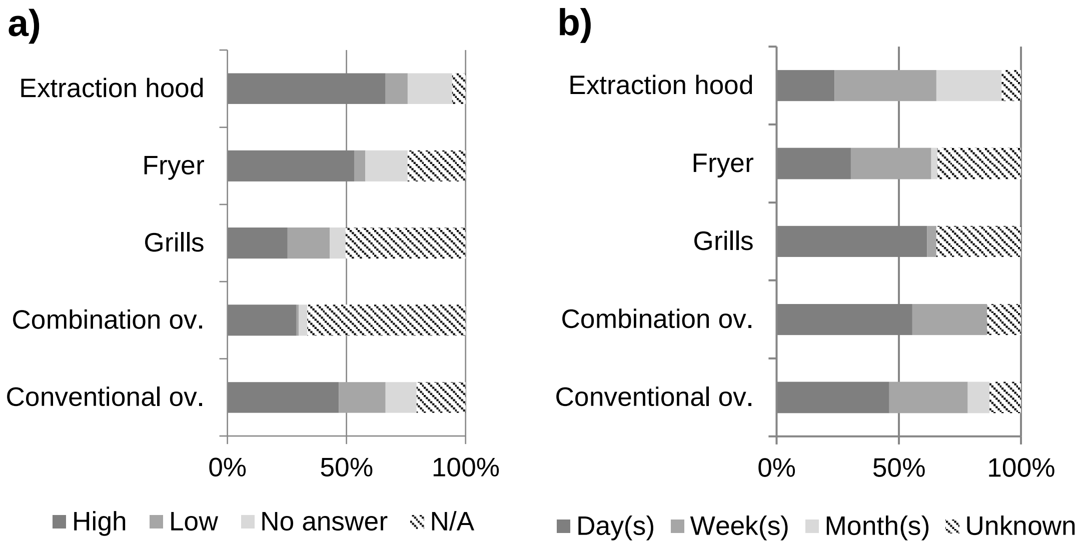

**Figure 4-1** Evaluated likeliness of kitchen appliances' cleaning to contribute to FOG emissions (a) and their associated cleaning frequencies (b).

Despite the requirement for commercial kitchens to be fitted with an approved grease separator or other effective means of grease removal, a large number of the surveyed establishments (69%) did not possess any type of FOG remediation system to treat their effluents and only 23% were using physical separation. In comparison, field experience has reported similar figures further estimating that only 10% to 20% of FSEs have a grease separator (ECAS, 2016; Thames Water Utilities, 2018). Three main types of grease separation devices were recorded through the survey: grease interceptors which are normally located underground in the sewage collection system (2% of all respondents), and smaller indoor devices which can be either solely gravity-based (11% of all respondents) or hydro-mechanical (6% of all respondents) (Figure 4-2). It is worth mentioning that two of the premises had a grease collection system which was, in reality, a wet well, downstream from the kitchen allowing FOG accumulation and whose efficiency was further questioned. Another notable finding was that three respondents were using a combination of physical and biological remediation techniques to reduce their FOG discharges, and one respondent was using both grease separators and a GRU

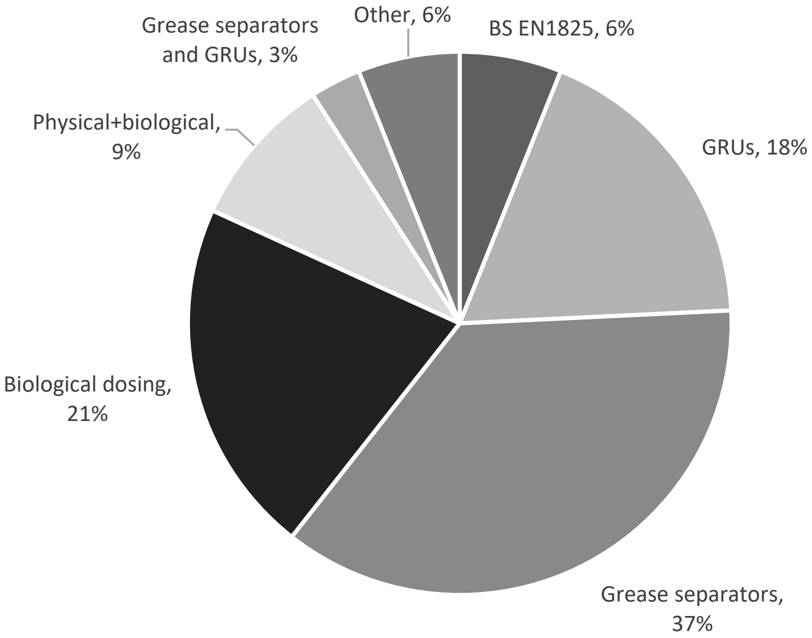

**Figure 4-2** Types of FOG remediation technique in FSEs. In total, 33 out of 107 FSEs (31%) had a mean to deal with their FOG discharges. “Other” refers to the presence of a simple well downstream the kitchen potentially allowing FOG retention.

To ensure grease separators achieve efficient FOG removals, adequate and frequent maintenance is required (Ducoste et al., 2008). Maintenance is largely dependent upon the unit type (Table 4-2). Typically, large grease interceptors require emptying out on a three-months basis (Wallace et al., 2017) whilst pump out intervals of three weeks to three months are recommended for smaller grease separators (Gurd, 2018). In comparison, hydro-mechanical units require daily maintenance (e.g. emptying oil collection cassette, cleaning of wiper blades) with deep cleaning planned every three to four months. Based on interviewees’ responses, 24% of the grease separators installed were not maintained as regularly as recommended in existing standards. In other words, only 15% of the

FSEs surveyed were potentially efficiently physically managing their effluents. In addition, there are reasons to assume that some of the systems installed were undersized. In one instance, the premises owner had bought a separator from the internet with no consideration to flow rates. In another case, the grease separator was already fitted when the premise was repurposed. In their study, Gallimore et al. (2011) demonstrated that a doubling of the flow rate to grease separators could reduce their efficiency by up to 96% depending on the type of unit.

**Table 4-2** Maintenance frequency of grease separators. Recommended cleaning frequencies are based upon existing standards (Table A 1).

| Unit type | Recommended | Daily | 1 to 3 weeks | 1 to 3 months | More than 3 months | Unknown |
|---|---|---|---|---|---|---|
| **Grease interceptor** | Every 3 months | | | 2 | | 1 |
| **Passive grease separator** | Three weeks to two months | | 6 | 5 | 3 | 3 |
| **GRU** | Daily | 5 | 1 | 1 | | |

Local authorities conduct regular inspections of FSEs through their environmental health responsibilities and there are recorded cases of Environmental Health Officers (EHOs) recommending the removal of grease separators as a potential health hazard (Drinkwater et al., 2017; Grenz and Patel, 2007). Only 27% of respondents with grease management processes reported that their systems had been inspected by an EHO.

By comparison to physical remediation, 7% of surveyed FSEs used biological additives as a remediation technique. Interestingly, 43% of these respondents experienced the effect of FOG on their drains (Figure 4-2). Whilst this data does not permit hard and fast conclusions to be drawn on the efficacy of biological additives it is important to understand its wider context. Numerous products are commercially available claiming to degrade FOG but years of experience within water utilities, with inconclusive trials, have led to scepticism regarding their efficacy (Mattsson et al., 2014; Mosholi and Cloete, 2018; Shaffer and Steinbach, 2007). In some cases, their use has even been banned by water utilities (Seiler, 2016).

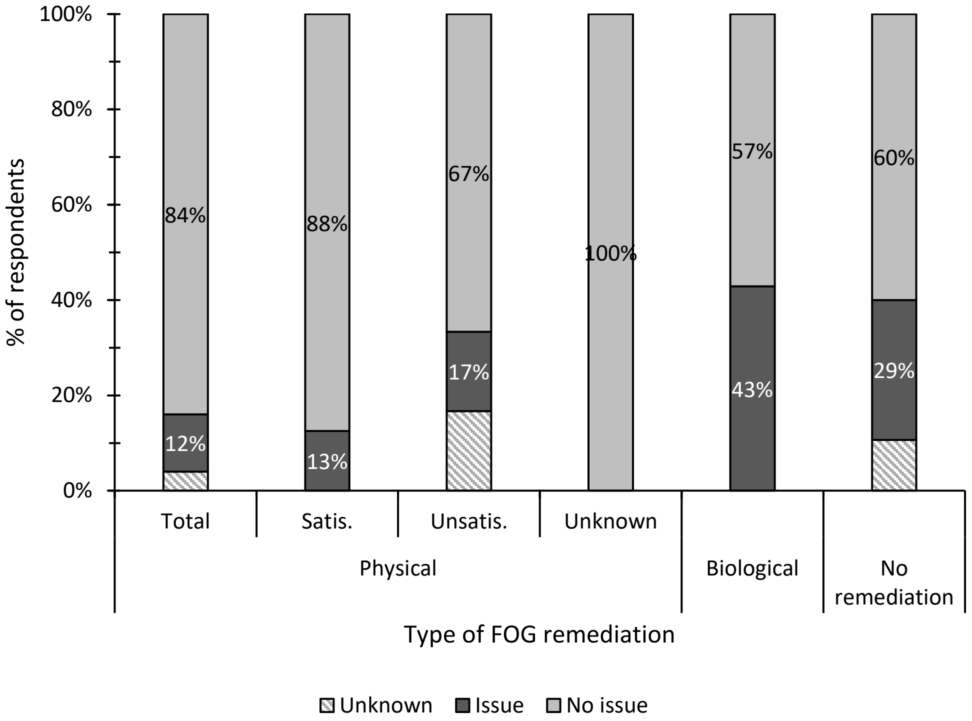

**Figure 4-3** Respondents' experience with FOG-related problems in relation to remediation techniques. FSEs relying on both physical and biological remediations are grouped as "Physical".

Finally, respondents were asked about the reasons for the absence of FOG management where this was evident through earlier questioning. A total of 21% of respondents did not know about the existence of remediation techniques. In comparison, 37% of respondents believe they did not need any grease management. During the interviews, a common misconception from FSE operators was that FOG only exists in the form of UCOs and that their recycling is sufficient to avoid accumulation of grease in sewer lines. These results clearly suggest the need for more efforts to be conducted in terms of educating FSE operators.

Surprisingly, around 65% of the FSE operators had never experienced the effect of FOG in their sewer lines. UK sewerage companies are responsible for private sewers and lateral drains whilst FSEs are only liable for sections between their property and the lateral drain. Keener et al. (2008) reported that FOG deposits tend to form between 50 to 200 metres downstream of FSEs. Consequently, sewer operators accrue most of the problems from uncontrolled discharges of FOG with FSEs being spatially removed from the problems they create. Critically, it is possible due to the relatively low number of FOG-related incidents in FSEs, FOG is not perceived as an issue for them and do not justify investment in grease management.

## 4.4 Towards a risk register

Drawing from information gathered through the survey reported above together with data from previous studies, a risk register is proposed to classify different kitchen operations, equipment, and behaviours in terms of their potential to cause FOG deposition in mains sewers (Table 4-3).

**Table 4-3** Evaluated risk of activities with the potential of contributing to FOG discharges.

| Activity | Risk | Comment(s) |
|---|---|---|
| **Combination ovens** | Low | Whilst combination ovens discharge their effluents directly into the drains, in most cases, these appliances were used as steamers. |
| **Grills** | Medium | Perceived as a lower risk than other sources considering cleaning practices. |
| **Conventional ovens** | Medium | Perceived as a lower risk than other sources considering cleaning practices. |
| **Pre-rinse sinks** | High | Posited to be one of the main sources of FOG in light of cleaning practices. |
| **Dishwasher** | High | FOG measurements demonstrated high concentrations of chemically stable oil emulsions in dishwasher effluents posited to be more difficult to manage. |
| **Extraction hood** | High | Cleaning operations contributing to the release of oil-rich water into the drains through dishwasher or kitchen sink. |
| **Fryer** | High | Cleaning operations contributing to the release of oil-rich water either directly into the drains or through dishwasher or kitchen sink. |

Combi-ovens were assigned with the lowest impact score as in most cases, they were used as steamer rather than for cooking high-fat foods, therefore limiting the amount of grease generated and discharged into the drains. By contrast, cleaning of exhaust hoods, in particular their filters, could contribute to large volumes of FOG entering the drainage system. Based on the efficiency of a prototype for treating grease filter washwater developed by Ghaly et al. (2007), FOG concentrations in these washwaters are estimated at 9 g.L$^{-1}$, being 10 times higher than that of kitchen sinks. Whilst no data was captured on FOG emissions from fryers, it is safe to assume their impact to be of similar significance to that of exhaust hoods. In light of cleaning frequencies captured in Figure 4-1b, this would translate into periodical high discharges of FOG into the drains from exhaust hoods and fryers. Similarly, for conventional ovens and grills, estimated at a medium impact, sinks were highlighted as one of the main disposal routes for detergent-rich washwaters. Ultimately, there is reasonable evidence suggesting that kitchen sinks and dishwashers should be a priority for interventions. Of particular interest, higher FOG concentrations were measured from sink effluents than from dishwasher effluents, respectively at 879±583 mg.L$^{-1}$ and 313±92 mg.L$^{-1}$. Furthermore, both sink and dishwasher effluents display different physicochemical properties suggesting they would require to be managed using distinct techniques. To illustrate, dishwashers produce chemically-stable oil-water emulsions with droplet sizes smaller than 20 μm (Chan, 2010) but conventional gravity-based separators are only believed to remove efficiently oil droplets greater than 30-50 μm (Ryan, 1986). This suggests that grease separators might not be well suited for dishwasher effluents. By contrast, Gurd et al. (2019) suggested that biological additives were more likely to achieve removal of these FOG droplets.

Common practice within the industry recommends a FOG management system on each contamination point. However, research has shown that the efficacy of FOG management is source dependent. From a FSE point of view, managing several FOG control systems could become a financial burden impacting their business profitability and, in turn, a major obstacle to implementation. In proposing trade-off solutions, further research will be needed to quantify and characterise the different FOG fluxes. To illustrate, data published by Gurd et al. (2019) puts into perspective with daily water usage for sinks and dishwashers, estimated at 1,703 L and 1,624 L (Gleick et al., 2003) suggests that 1.5 kg and 0.5 kg of FOG would be allowed daily from one FSE into the drains (from sink and dishwasher respectively). Whilst a case-by-case approach is recommended over a one-size-fits-all approach, it is possible that targeting in priority kitchen sinks would offer the highest benefits in terms of sewer relief.

## 4.5 Conclusions

This study exposed a number of behaviours possibly contributing to discharges of FOG into sewerage systems. Whilst waste frying oils were identified by FSEs operators as one of the main sources of FOG, other pathways were often unacknowledged. Very few FSEs were potentially mitigating the impact of their effluents on the sewers. On recommending biological additives as a sustainable remediation technique, there is a great need for the development of standardised methodologies for their testing and application as, currently, it is often limited to manufacturers' claims.

There is a significant effort required to sensitise FSE operators to the broader context of FOG management. It is critical for the industry to reach a consensus on FOG management in order to provide a structured framework to FSEs. In addition, there is a broad need for a joint approach between stakeholders (i.e. local authorities and water utilities) to ensure compliance is maintained in FSEs over time as it could degrade due to high staff turnovers.

## 4.6 Acknowledgments

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# 5 Separation of fats, oils and greases (FOG) from kitchen effluents: Waste characterisation, production rates and potential for anaerobic co-digestion with thermo-hydrolysed sewage sludge

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## Abstract

Fats, oils and greases (FOG) were collected from grease removal units (GRUs) from 14 food service establishments (FSEs). FOG was generated at rates ranging from 0.9 to 18.0 g per meal served. The oil phase from the collected materials demonstrated relatively low water content ranging from 1.9% to 32.3%. Lipids were the most prominent organic macromolecules found at 54.0-99.0% in terms of dry solids. Expectedly, this translated into high ultimate biomethane potentials measured at 871.6±147.6 STP mL per g volatile solids (VS) added. The addition of FOG to thermo-hydrolysed (TH) and non-hydrolysed sewage sludge at 7.5% VS increased biomethane yields by 10% and 8% respectively. The use of TH sludge did not lead to a reduction in the utilisation of FOG in anaerobic digesters.

**Keywords**: Sewer deposit; fatberg; food service establishments (FSEs); biomethane

## 5.1 Introduction

Fats, oils and greases (FOG) are ubiquitous to wastewater originating from households, food service establishments (FSEs) and industries. Minimising FOG emissions is critical to avoid the formation of fatbergs in sewer systems contributing to high operational costs for water utilities estimated between £15M and £50M per annum in the UK (Williams et al., 2012). Discharges from FSEs are generally managed using grease separators which are designed to retain FOG from wastewater based on the difference of density between oil and water.

Grease separators create a new waste stream, and using collected wastes, also referred to as grease trap waste (GTW), as co-substrates in anaerobic digestion with sewage sludge has been widely studied to recover energy from FOG as reported by Long et al. (2012). This process, consisting in the joint treatment of sludge and organic material (i.e. FOG), produces methane-rich biogas which can be used for electricity generation increasing the overall sustainability of sewage treatment works (STWs). The addition of FOG, at concentrations ranging between 5% to 48% in terms of volatile solids (VS) fed, has been reported to increase methane generation rates between 9% and 66% (Davidsson et al., 2008; Kabouris et al., 2009a). However, high concentrations of long-chain fatty acids (LCFAs), which are a prominent component of FOG, are known to hamper biogas production by affecting substrate transport and utilisation and inhibiting methanogens (Cockrell, 2007; Sober et al., 2010). Therefore, it becomes critical to document the characteristics of the wastes to be used. In order to ensure the efficient bioconversion of LCFAs and overcome their potential inhibition, different strategies have been proposed: (1) saponification of lipids to enhance solubilisation (Battimelli et al., 2010), (2) enzymatic pre-treatments (Bouchy et al., 2012; Cirne et al., 2007) and (3) the application of pulse-feeding procedures (Palatsi et al., 2009; Ziels et al., 2017).

However, despite these reported benefits, studies focusing on anaerobic co-digestion have not taken into consideration recent advancements in sludge treatment such as thermal hydrolysis (TH) aiming to solubilise organic matter to make substrates more available to microorganisms by applying high pressure and high temperature. Critically, it is hypothesised that microbial communities would be less inclined to digest FOG in presence of other substrates made easier to utilise and, in turn, making the process inefficient to recover energy from FOG. Yet, the joint TH of FOG and sludge is not recommended as it has been shown to be unsuccessful at promoting the degradation of LCFAs into simpler substrates (Charuwat et al., 2018), and in the case of fat-rich slaughterhouse waste actually promoted the formation of inhibitory compounds to the digestion process (Cuetos et al., 2010).

Reduction of FOG discharges from FSEs is achieved in two types of physical separators that principally differ in how the removed FOG is stored. In traditional separators the captured FOG remains within the tank and is periodically removed with a proportion of the water to generate GTW. By contrast, grease removal units (GRUs) transfer the captured FOG into a separate tank expected to be more FOG-rich water-light compared to GTW. To illustrate, GTW has been reported to contain up to 95% water (Miot et al., 2013; Robbins et al., 2011). To date, there is no reported data on the quantity and character of GRU wastes and it is often assumed to mirror that of used cooking oils (UCOs) (Wallace et al., 2017).

The overall objective of this investigation was to extend knowledge on the characteristics of FOG collected from FSEs and assess their value as substrates for anaerobic co-digestion in advanced systems that utilise TH pre-treatment of the sewage sludge. To achieve this, FOG wastes collected from 14 FSEs using GRUs were characterised in terms of their physicochemical properties and biomethane potentials. These wastes were further used as feedstock for anaerobic co-digestion that encompassed the TH treatment of sewage sludge. To the authors’ knowledge this paper represents the first study ascertaining volumes of FOG produced from FSEs based on actual data using GRUs (Table 1-1) and using FOG as co-substrate with TH sludge.

## 5.2 Material and methods

### 5.2.1 Collection systems

A total of 14 FSEs were part of this study; their kitchens were surveyed by the GRU manufacturers who recommended installation on each grease contamination point (e.g. combination ovens, pre-rinse sinks). Three GRU models, complying with the ASME A112.14.4 standard (ASME, 2001), were tested, and a total of 24 units were installed (Table D-1). The three models relied on an oleophilic and hydrophobic skimming system. Two of the units possessed a self-regulated electric heating element to melt animal fats whilst the third one relied on hot kitchen effluents to ensure FOG were not solidifying.

The FSEs covered: 2 pubs, 1 café, 2 schools, 1 nursing home and 8 full-meal restaurants. FSEs were provided with 10 L containers to dispose of the waste removed from the GRUs. When full, containers were collected, weighted and their content was transferred into a 20 L glass aspirator where it was allowed to settle for 20 minutes to separate both oil and aqueous phases by gravity.

## 5.2.2 Physicochemical characterisation

### 5.2.2.1 Aqueous phase

Aqueous phase total chemical oxygen demand (tCOD) was measured using Hach Lange test kits (LCK 014). In addition, FOG concentrations were determined as hexane extractable material (HEM). For HEM analyses, samples, collected in 1 L glass bottles, were filtered using a Whatman$^{TM}$ GF/C grade filter paper. The filter paper was then immersed in boiling hexane (40 to 60˚C), using a SOXTHERM® unit, in a pre-weighted glass extraction beaker. Oil and grease were determined by weight difference and further reported in mg.L$^{-1}$.

### 5.2.2.2 Oil phase

Dry solids (DS) and VS were measured according to APHA methods (APHA, 2005). A chemical characterisation of the main organic fractions including lipids, fibres and proteins was performed on each material. Fibres were measured as the organic matter remaining after samples were de-fatted and digested successively with acid and alkali under controlled conditions (Horwitz, 2003). Crude proteins were determined as total Kjeldahl nitrogen defined as N x 6.25 (Sciantec Analytical, 2018b). Lipids were measured using a modified Wiebul acid hydrolysis method (Sciantec Analytical, 2018c). Carbohydrates were estimated subtracting the contribution of lipids, fibres and proteins.

Fatty acids profiles were obtained by gas-liquid chromatography using a free fatty acid phase column of dimensions 25m x 0.20mm ID and detection by flame ionisation detector. Fats and oils were trans-esterified to fatty acid methyl esters by heating under reflux for two hours with a mixture of methanol and sulfuric acid in toluene. The resulting methyl esters were extracted using a small volume of n-hexane. The n-hexane solution was dried using anhydrous sodium sulphate and then transferred to a chromatography vial (Sciantec Analytical, 2018d).

Gross calorific values were determined experimentally using a calorimeter (Parr model 6100) equipped with a 1108CL oxygen bomb. Samples were freeze dried beforehand (Sciantec Analytical, 2018e). The lower heating values (LHV) were estimated from the measurement of calorific values by subtracting the heat of vaporisation of water in the products as follows:

$$LHV_d = HHV_d \times (1 - M) - H_V \times M \qquad \textbf{(5-1)}$$

Where $M$ is the moisture content, $H_V$ is the latent heat of vaporisation of water estimated at 2.447 MJ.kg$^{-1}$ at 25˚C and $HHV_d$ is the gross heating value in MJ.kg$^{-1}$ on dry basis determined as follows:

$$HHV_d = \frac{HHV}{1 - M} \qquad \textbf{(5-2)}$$

Where $HHV$ is the measured HHV on wet basis.

## 5.2.3 Anaerobic co-digestion

### 5.2.3.1 Inoculum and substrates

Two main sets of experiments were conducted using either TH or conventional (i.e. non-hydrolysed) sludge. In both cases, fresh sludge samples were collected weekly to fortnightly and stored at 4˚C. Digested sludge, serving as inoculum for batch testing and to seed semi-continuous reactors with TH sludge, was sampled from a full-scale anaerobic digester treating municipal sewage. This plant utilises Cambi thermal hydrolysis as a pre-treatment prior to anaerobic digestion. TH sludge samples were taken from the same site after this pre-treatment step. For the digestion runs using non-hydrolysed sewage sludge, samples were obtained from a different STW prior to anaerobic digestion. Semi-continuous reactors were initially seeded with digestate from the same STW.

**Table 5-1** Inoculum and substrate characteristics.

| Parameters | TH inoculum | TH sludge | Conv. sludge | FOG |
|---|---|---|---|---|
| DS (%) | 5.3±2.2 | 9.4±1.0 | 6.0±1.1 | 80.3±14.0 |
| VS (% DS) | 60.4±2.6 | 77.7±1.6 | 76.4±3.6 | 100.0±0.0 |
| Lipids (% DS) | 6.7±0.5 | 12.8±2.4 | | 88.0±11.2[a] |
| pH | | 5.7±0.1 | 6.4±0.2 | |
| VFA (mg.L$^{-1}$) | | 3,124.1±906.0 | 5,134.7±2,141.0 | |
| Alkalinity (mg.L$^{-1}$) | | 2,279.1±317.5 | 4,181.5±1,807.0 | |

[a] Average used from the characterisation of oil phases.

### 5.2.3.2 Batch experiments

Triplicates batch testing were performed to investigate the ultimate biomethane production rates of each material using an AMPTS II system (Bioprocess Control, Sweden). The assays were performed at mesophilic temperatures (39˚C) using an inoculum to substrate ratio of 2 g VS inoculum per g VS substrate. The experiments were terminated when the cumulative biomethane production reached a plateau phase after around 100 days. Biogas produced was passed through a $CO_2$ stripping solution, and methane production rates were adjusted to standard conditions of temperature and pressure (STP) as follows:

$$V_{STP} = \left(1 - \frac{P_{vap}}{P_{gas}}\right) \times \frac{P_{gas}}{P_{STP}} \times \frac{T_{STP}}{T_{gas}} \times V_{gas} \qquad \textbf{(5-3)}$$

Where $V_{STP}$ is the volume adjusted to STP, $P_{STP}$ is the standard pressure (101.3 kPa), $T_{gas}$ is the temperature of the measured gas (311 K), $T_{STP}$ is the standard temperature (273 K) and $V_{gas}$ is the measured volume of gas. $P_{gas}$ was calculated as the sum of the partial pressures of methane and carbon dioxide. $P_{CO2}$ was neglected in the case of the batch testing as carbon dioxide was removed through the stripping solution. $P_{vap}$ is the water vapour pressure calculated as follows:

$$P_{vap} = 10^{8.1962 - \frac{1,730.63}{T_{gas} - 39.724}} \qquad \textbf{(5-4)}$$

DS, VS and lipids were determined before and after the digestion period. Control reactors, without substrate, were used to determine the endogenous methane production rate of the inoculum which was subtracted from the test reactors.

### 5.2.3.3 Semi-continuous experiments

To mimic full-scale operations and investigate the degradation of FOG, two sets of experiments were conducted: using thermo-hydrolysed (R1-R3) and non-hydrolysed (R4 and R5) sewage sludge. Continuously stirred reactors were operated at mesophilic temperatures (around 39˚C) in semi-continuous conditions with daily manual feeding. Equal volumes of material were withdrawn and fed to maintain a constant working volume of 8 L. Temperature was controlled by immerging the reactors in a water bath heated by an electrical resistance. Each reactor was connected to water bottles and biogas volumes were recorded daily as the total volume of water displaced from the bottle. The biogas content was measured daily using a gas analyser (Geotech GEM2000 PLUS). Volumes were adjusted to STP as described in section 5.2.3.2.

The organic loading rate (OLR) was fixed at 4.5 kg VS.m$^{-3}$.d$^{-1}$ for the control digester (R1). During six retention times, FOG was introduced into R2 and R3 every 48 hours at a concentration of 7.5% VS of the control OLR. After this period, FOG was added daily at a concentration of 7.5% VS (of R1 load).

DS and VS were measured daily, whilst pH, volatile fatty acids (VFA) and alkalinity were measured at least bi-weekly in the digestate to evaluate process stability. After three hydraulic retention times (HRTs), lipids in the digestate were analysed weekly or bi-weekly. Briefly, samples were dried and then hydrolysed with hydrochloric acid followed by ether extraction (Sciantec Analytical, 2018c). Capillary suction time (CST) was used to measure digestate dewatering behaviour. It was determined using a Triton Electronics 304B and Whatman 17 CHR filter paper.

Another digestion run was performed using two reactors operated under similar conditions than those with thermo-hydrolysed sludge. These reactors were fed with non-hydrolysed sewage sludge. R4 served as a control and was fed daily only with sludge whilst FOG was added daily to R5 at a concentration of 7.5% VS (of the control load). This experiment was conducted to identify possible differences in biomethane generation from FOG and conventional sludge. Biogas volumes were recorded as described for reactors R1-R3. DS, VS, pH, VFA and alkalinity were also measured weekly to bi-weekly.

**Table 5-2** Reactors characteristics.

| **Parameter** | **R1 Control** | **R2 Run 1** | **R2 Run 2** | **R3 Run 1** | **R3 Run 2** | **R4 Control** | **R5** |
|---|---|---|---|---|---|---|---|
| **FOG feed (% VS)** | 0 | 7.5 | 7.5 | 7.5 | 7.5 | 0 | 7.5 |
| **OLR ($kg\ VS.m^{-3}.d^{-1}$)** | 4.5 | 4.6 | 4.9 | 4.6 | 4.9 | 2.5 | 2.6 |
| **Volume (L)** | 8.0 | 8.0 | 8.0 | 8.0 | 8.0 | 8.0 | 8.0 |
| **Temperature (˚C)** | 39 | 39 | 39 | 39 | 39 | 39 | 39 |
| **HRT (days)** | ~16 | ~16 | ~16 | ~16 | ~16 | ~17 | ~17 |

## 5.3 Results and discussion

### 5.3.1 FOG production rates

On average, FOG accounted for 40% of the GRU wastes collected with variations reported from 12-99% between FSEs. Due to their nature, oleophilic materials used to mechanically remove FOG will inevitably capture anything with an oil-like nature (e.g. FOG, soaps). Quantities of FOG produced ranged from 0.9 up to 18.0 g per meal served with a median value of 3.9 g per meal (Figure 5-1a). Interestingly, the lowest FOG production rate was observed for one of the schools (FSE 10). Further investigation indicated that food was served in disposable dishes for students consequently contributing to smaller volumes generated from washing up activities. During this study, no FOG was collected from FSE 12 further attributed to the relatively lower number of covers served. Similarly, sites 7 and 3 used no or very limited amounts of cooking oil leading to production rates of 1.9 and 2.6 g of FOG per cover respectively. Whilst FSEs 1 and 8 were both classified as pubs, their respective FOG production rates were 3.1 and 18.0

g.cover$^{-1}$. This difference was further posited to be related to the type of food being prepared, with FSE 1 being a gastropub and FSE 8 a more conventional pub mainly serving fried food. It is worth mentioning that over the course of the trial, units specifically connected to combination ovens did not collect any waste as they were often only used as steamers.

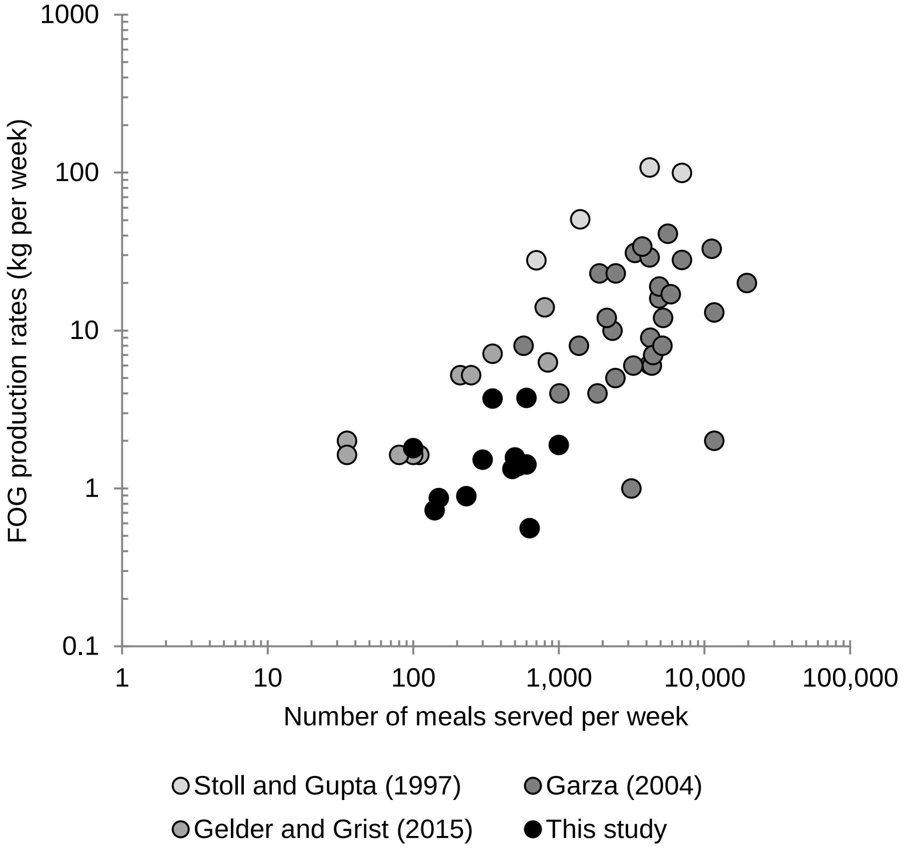

**Figure 5-1** FOG production rates reported against the number of covers served.

The current measured dataset enables validation of previous estimates based on either theoretical (Gelder and Grist, 2015) or FOG concentrations measured in effluents from five restaurants in Bangkok (Stoll and Gupta, 1997) and from grease interceptors at 28 FSEs in the US (Garza Armando, 2004). The former indicated rates between 7.5 and 46.6 g of FOG per meal and the later 0.2 up to 14.6 g per meal based on an assumption that 77% of the FOG is captured from grease separators (Gallimore et al., 2011) (Figure 5-1). Production rates calculated from FOG measured in effluents from Thai restaurants were found between 1.5 and 39.8 g of FOG per meal. The current dataset is predominantly at the lower end with a rate less than 6 g of FOG per meal observed at 10 of the 13 sites. This potentially indicated FOG levels are lower than previously estimated which, in part, reflect changes in food practice and cuisine preference as well as general kitchen practice.

**Table 5-3** Characteristics of FOG wastes collected from GRUs installed at 14 FSEs.

| **FSE** | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **12** | **13** | **14** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Type** | Pub | Café | School | Rest. Indian | Rest. Italian | Rest. Hotel | Rest. Italian | Pub | Rest. Museum | School | Rest. Lebanese | Rest. British | Nursing home | Rest. Indian |
| **Weekly covers** | 500 | 480 | 525 | 300 | 600 | 350 | 1,000 | 100 | 600 | 360 | 150 | 100 | 231 | 140 |
| **UCO (L.week$^{-1}$)** | 25 | 20 | 1 |  | 100 | 50 | 0 | 20 |  | 10 | 9 | 3 |  | 30 |
| **FOG (%)** | 63 | 30 | 52 | 99 | 69 | 47 | 24 | 33 | 46 | 12 | 79 |  | 13 | 92 |
| **FOG (L.d$^{-1}$)** | 0.25±0.14 | 0.21±0.04 | 0.22±0.11 | 0.24±0.17 | 0.64±0.35 | 0.55±0.30 | 0.29±0.10 | 0.29±0.20 | 0.22±0.17 | 0.09±0.05 | 0.14±0.08 |  | 0.14±0.017 | 0.11±0.08 |
| **FOG (g.cover$^{-1}$)** | 3.1 | 2.8 | 2.6 | 5.1 | 6.2 | 10.6 | 1.9 | 18.0 | 2.4 | 0.9 | 5.8 |  | 3.9 | 5.2 |
| **Water (%wt)** | 19.1±13.2 | 4.7±3.8 | 4.8±3.7 | 5.0±2.0 | 2.7±2.4 | 4.3±4.5 | 24.6±13.1 | 25.9±12.6 | 8.5±6.2 | 32.3 | 26.3±20.7 |  | 11.1±16.5 | 12.5±8.5 |
| **VS (%wt)** | 80.7±13.4 | 95.6±3.6 | 94.2±4.5 | 95.0±2.0 | 97.4±2.4 | 96.5±3.2 | 75.1±13.3 | 79.1±7.4 | 91.5±6.0 | 67.7 | 74.2±20.9 |  | 89.0±16.3 | 87.5±8.5 |
| **Lipids (%VS)** | 86.4±9.8 | 85.4±14.7 | 88.2±9.4 | 99.0±1.8 | 90.4±6.5 | 93.9±1.3 | 88.3±12.4 | 97.9±3.6 | 90.9±0.5 | 54.0 | 81.2±7.0 |  | 80.4±0.3 | 96.6±0.0 |
| **Carbohydrates (%VS)** | 13.6±9.6 | 14.5±17.7 | 11.8±8.8 | 1.0±1.8 | 9.6±6.5 | 6.0±4.2 | 11.7±12.8 | 2.1±3.6 | 9.1±1.6 | 45.4±1.8 | 17.4±6.4 |  | 19.5±18.9 | 3.3±4.7 |
| **$\text{LHV}_d$ (MJ.kg$^{-1}$)** | 35±9 | 33±6 | 34±4 | 38±1 | 38±3 | 40±1 | 31±7 | 33±4 | 37±4 | 24 |  |  | 35±1 | 37±2 |
| **HEM (mg.L$^{-1}$)** | 43±16 | 104±18 | 2,496 | nd | 593±569 | 389±395 | 561±124 | 1,229±735 | 347±176 | 72 | nd |  | 545±131 | nd |
| **tCOD (g.L$^{-1}$)** | 6.6±0.2 | 3.3±0.2 | 9.1 | nd | 4.0±1.2 | 6.3±1.4 | 3.7±0.4 | 7.8±2.7 | 9.6±1.8 | 1.5 | nd |  | 5.8±1.3 | nd |

nd: not determined

## 5.3.2 Physicochemical characterisation

### 5.3.2.1 Aqueous phase

The aqueous phase from the GRU wastes was identified to be highly variable in terms of tCOD and HEM with respective ranges of 1.5-9.57 $g.L^{-1}$ and 43.0-2,496.0 $mg.L^{-1}$. In comparison, Suto et al. (2006) reported FOG concentrations in the aqueous phase of GTW to range from 130 to 93,000 $mg.L^{-1}$ with an average value of 15,416 $mg.L^{-1}$. Similarly, tCOD from GTW aqueous phase was reported at much higher concentrations 15.9-21.2 $g.L^{-1}$ (Lopez et al., 2014) and 1.3 to 566.0 $g.L^{-1}$ (Suto et al., 2006). The difference reflects the impact of regular removal of the captured FOG has GRUs compared to interceptors reducing the potential for the material to be extracted in the aqueous phase.

### 5.3.2.2 Oil phase

The water content measured in the oil phase from the collected material varied across the FSEs, from 2.7% to 32.3% (Table 5-3). The median value across all FSEs was measured at 11.1%. These values were significantly lower than those reported in literature for GTW ranging from 27.0% up to 96.8% (Davidsson et al., 2008; Evans et al., 2012; Kabouris et al., 2009a; Martínez et al., 2012; Miot et al., 2013; Moyce and Murray, 2010; Wan et al., 2011). The median value from published figures was 84.4% and reflects that the contents are extracted from the interceptor diluting the captured FOG. This indicates a potential benefit for GRUs instead of interceptors when considering collection and utilisation of FOG from FSEs.

All the solids contained in FOG samples were volatile. Lipids were the most prominent macromolecules in FOG with ranges from 54.0-99.0% DS. Across all FSEs, the median was measured at 88.3% DS (Table 5-3). By contrast, reported lipids concentrations in GTW were highly variable, from 12.8-91.0% DS with a median value of 31.0% DS (Evans et al., 2012; Kabouris et al., 2009a; Martínez et al., 2012; Miot et al., 2013; Moyce and Murray, 2010; Wriege-Bechtold et al., 2010).

Consequently, with relatively low water content and high lipids concentrations, GRU-FOG was found with high $LHV_d$ ranging 24-40 $MJ.kg^{-1}$ (with a median of 37 $MJ.kg^{-1}$). Some of the collected wastes had $LHV_d$ close to that of UCOs, ranging between 37-40 $MJ.kg^{-1}$ (Ortner et al., 2016; Sanli et al., 2011; Supple et al., 2002). Critically, these wastes represent a valuable source of energy as lipids are capable of yielding more methane during anaerobic digestion (Labatut, 2012).

In using FOG as co-substrate, attention should be paid to LCFAs which are known to have inhibitory effects on the process. The most common LCFAs found in all the FOG samples were oleic acid (C18:1), ranging from 27.4 to 58.2% of total fatty acids, 11.2-44.9% for linoleic acid (C18:2) and 9.8-21.5% for palmitic acid (C16:0) (Figure 5-2a). Other LCFAs were found at lower concentrations such as linolenic acid (C18:3), ranging from 0.7-6.1% of total fatty acids and stearic acid (C18:0) from 3.4-6.5%. These concentrations were found within the same range that those reported by other authors in literature (Canakci, 2007; Karnasuta et al., 2007; Suto et al., 2006; Tran et al., 2016) (Figure 5-2a).

Data on LCFAs toxicity, reported as the concentration causing a 50% relative activity loss of the specific methane production, has been published in literature by several authors (Alves et al., 2001; Lalman and Bagley, 2000; Pereira et al., 2005; Prinst et al., 1972; Shin et al., 2003). Based on their reported $IC_{50}$, inhibitory loadings were calculated for the main LCFAs for each FOG sample (Figure 5-2b). According to literature, oleic (C18:1) and linoleic acids (C18:2) are the main inhibitory LCFAs found in FOG (Alves et al., 2001; Lalman and Bagley, 2000). Their respective inhibitory loadings were found at 0.9 and 1.2 g $VS_{added}$, corresponding to FOG loading rate of 3.3% and 2.5% in terms of VS in the semi-continuous reactors. Thus, the initial amount of FOG pulse-fed every 48 hours to semi-continuous reactors was selected at 7.5% VS. In order to ensure the efficient bioconversion of LCFAs and overcome their potential inhibition, different strategies have been proposed: (1) saponification of lipids to enhance solubilisation (Battimelli et al., 2010), (2) enzymatic pre-treatments (Bouchy et al., 2012; Cirne et al., 2007) and (3) the application of pulse-feeding procedures

(Palatsi et al., 2009; Ziels et al., 2017). In this study, in a first run, pulse-feeding was applied to allow biomass acclimation in anaerobic co-digesters.

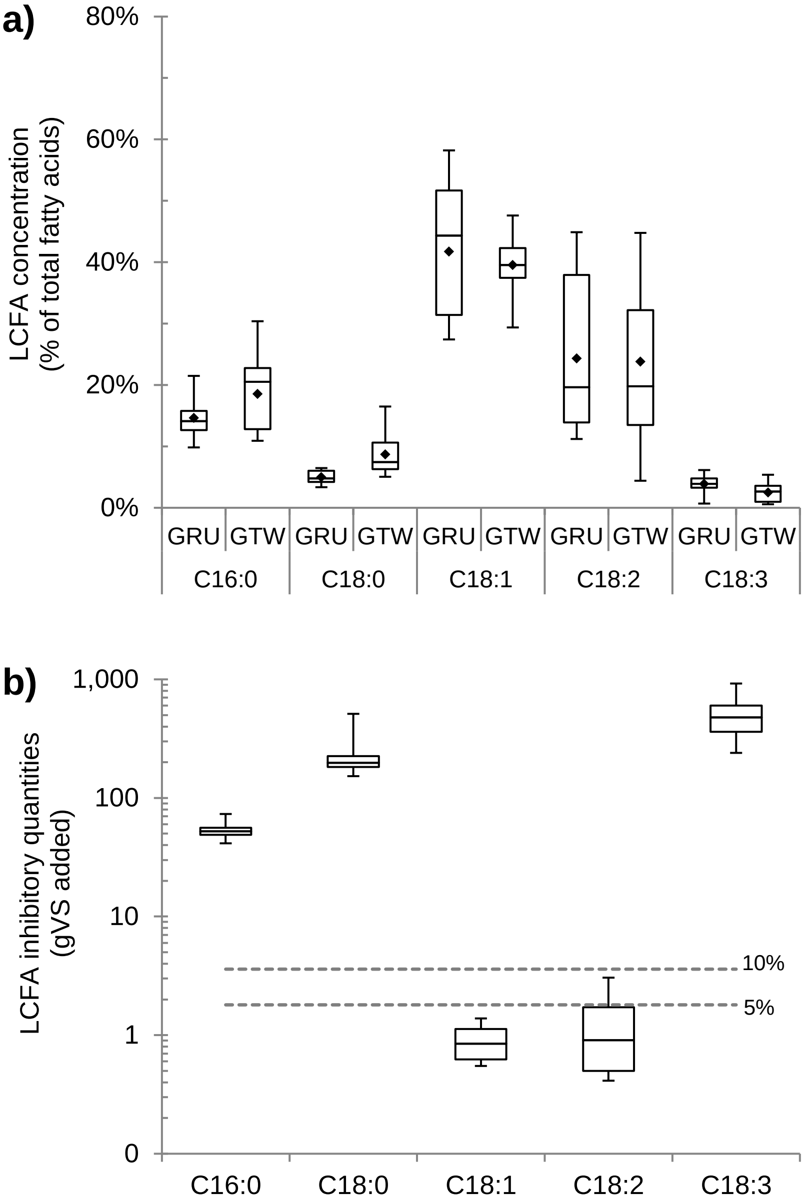

**Figure 5-2** LCFA concentrations measured in GRU FOG and from literature (a) and their associated inhibitory concentrations (b). Inhibitory concentrations were based on values published in literature, and from the characterisation of FOG in this study.

### 5.3.3 Anaerobic batch testing

Batch testing was conducted to assess the methane potential of GRU-FOG and compare it to that of sewage sludge. Whilst sewage sludge yielded 383±61

STP mL $CH_4$.g $VS_{added}^{-1}$, FOG generated 872±148 STP mL $CH_4$.g $VS_{added}^{-1}$ (Table 5-4). These experimental values were supported by the theoretical methane potentials, calculated from the macromolecules concentrations based on Buswell's equation (Angelidaki and Sanders, 2004) from 735.9 up to 1,007.9 STP mL $CH_4$.g $VS_{added}^{-1}$. The median value of the sample was found at 944.2 STP mL $CH_4$.g $VS^{-1}$. Methane yields obtained for the FOG samples were in the range of that reported by Kabouris et al. (2009) at 993 STP mL $CH_4$.g $VS_{added}^{-1}$ in their study on polymer-dewatered GTW. The VS destruction was calculated at 57±16 and 89±5% respectively for the sludge and FOG samples. Lipids degradation was calculated at 74±7% for sewage sludge and 94±8% for FOG. This equated to a methane yield of 944±69 STP mL $CH_4$.g $lipid_{destroyed}^{-1}$.

**Table 5-4** Results from the batch testing of sewage sludge and GRU-FOG.

| **Parameter** | **TH sludge** | **FOG** |
|---|---|---|
| Experimental methane yield (STP mL.g $VS_{added}^{-1}$) | 383±61 | 872±148 |
| VS destruction (%) | 57±16 | 89±5 |
| Lipids destruction (%) | 74±7 | 94±8 |
| Experimental methane yield (STP mL.g $VS_{destroyed}^{-1}$) | 639±47 | 971±152 |
| Theoretical methane yield (STP mL.g $VS_{added}^{-1}$) |  | 944±69 |

Assuming that each waste has a cumulative effect to the overall VS destruction, the biomethane yields for pulse-feeding FOG at 7.5% and daily feeding at 7.5% were projected respectively at 400 and 415 STP mL $CH_4$.g $VS_{added}^{-1}$. In terms of VS destruction, these feeding regimes could generate 657 and 672 STP mL $CH_4$.g $VS_{destroyed}^{-1}$ respectively.

## 5.3.4 Anaerobic co-digestion in semi-continuous conditions

### 5.3.4.1 Co-digestion with thermo-hydrolysed sewage sludge

During the first run, VS destruction was 58.5±2.2% for the control digester, and 52.4±3.7% and 56.5±1.8% for reactors fed with 7.5% VS of FOG (respectively R2 and R3). The methane yields were higher for both co-digesters, measured at

552.7±36.2 and 544.3±28.6 STP mL.g $VS_{destroyed}^{-1}$ which corresponded to a 16-17% difference from the ultimate methane yields. In comparison, methane yields from the control digester were lower, measured at 502.0±28.4 STP mL.g $VS_{destroyed}^{-1}$.

During the second run, co-digesters were fed at 7.5% in terms of VS whilst the control was still operated under the same conditions. Methane yields were measured at 331.6±28.4 and 324.4±28.6 STP mL.g $VS_{added}^{-1}$ for R2 and R3 respectively corresponding to an increase of 13% and 11% to the control. The VS destruction for both co-digesters was found at 54.1±4.6% and 52.4±2.4%. Higher methane yields, normalised in terms of VS destroyed, were obtained for both co-digesters respectively at 612.7±51.1 and 619.6±46.9 STP mL.g $VS_{destroyed}^{-1}$ for R2 and R3. Methane yields were calculated at 912.1±134.7 and 834.5±147.8 STP mL $CH_{4.}$ g VS $FOG_{added}^{-1}$. These results were congruent with the range for the ultimate methane yields measured in batch reactor at 951.7±85.4 STP mL $CH_{4.}$ g VS $FOG_{added}^{-1}$, as well as specific biomethane yields obtained from lipids ranging between 903.9-1,101.2 STP mL $CH_4$.g$VS_{added}^{-1}$ (Labatut, 2012).

Lipids degradation in the control reactor was measured at 73.6±2.6% which was relatively close to degradation rates obtained from batch testing at 74±7% (Table 5-4). As a comparison, full-scale anaerobic digesters treating TH sludge were reported with lipids degradation rates ranging from 64% up to 71% (Liu, 2018). Methane production rates from FOG were calculated at 878.3±329.2 and 798.4±184.3 mL $CH_{4.}$ g FOG $lipids_{added}^{-1}$ respectively for R2 and R3 with daily feeding of FOG. FOG degradation rates were calculated at 83.5±5.8% and 77.1±5.2% respectively for R2 and R3 during the second digestion run. FOG production rates were therefore calculated at 1,102.8±212.0 and 1,043.5±274.5 STP mL $CH_{4.}$ g FOG $lipids_{destroyed}^{-1}$ respectively for R2 and R3.

Whilst both co-digesters were operated under similar conditions, the CSTs were found statistically different during the two runs for R2, being measured at 288.7±107.2 and 179.0±35.3 min.g $DS^{-1}$ (One-way ANOVA, $\alpha = 0.05$, *p*-value = 0.01, Table 5-5). CSTs measured for R3, at 318.3±63.4 and

270.5±48.7 min.g DS$^{-1}$, were found to be statistically different at a confidence level of 90% (One-way ANOVA, α = 0.1, *p*-value = 0.09, Table 5-5). With higher CSTs measured, it was posited that the digestate dewaterability was impacted by the addition of FOG especially during pulse feeding regimes, possibly due to the accumulation of undigested lipids. Another explanation is that the release of extracellular polymeric substances (EPS), stimulated by the addition of FOG (Xu et al., 2015; Yang et al., 2016), led to a poorer dewatering ability (Sheng et al., 2010). However, it is worth mentioning that CSTs, measured at 194.2±39.9 and 156.5±33.4 min.g DS$^{-1}$ for R1 respectively during run 1 and 2, were also found significantly different between both runs (One-way ANOVA, α = 0.05, *p*-value = 0.04, Table 5-5). Further research is recommended to fully capture the impact of adding FOG on digestate dewaterability.

**Table 5-5** Matrix **r**esults from one-way ANOVA performed on the CSTs. Cells highlighted in green and orange respectively represent statistically different results at a confidence interval of 95% and 90%.

| | | **R1** | | **R2** | | **R3** | |
|---|---|---|---|---|---|---|---|
| | ***p*-value** | **Run 1** | **Run 2** | **Run 1** | **Run 2** | **Run 1** | **Run 2** |
| **R1** | **Run 1** | | 0.04 | 0.007 | | $5.10^{-6}$ | |
| | **Run 2** | 0.04 | | | 0.21 | | $8.10^{-5}$ |
| **R2** | **Run 1** | 0.007 | | | 0.01 | 0.41 | |
| | **Run 2** | | 0.21 | 0.01 | | | $7.10^{-4}$ |
| **R3** | **Run 1** | $5.10^{-6}$ | | 0.41 | | | 0.09 |
| | **Run 2** | | $8.10^{-5}$ | | $7.10^{-4}$ | 0.09 | |

**Table 5-6** Effluent characteristics and reactor performance.

| **Parameter** | **R1 – Run 1** | **R1 – Run 2** | **R2 – Run 1** | **R2 – Run 2** | **R3 – Run 1** | **R3 – Run 2** | **R4** | **R5** |
|---|---|---|---|---|---|---|---|---|
| **DS (%)** | 5.2±0.3 | 5.0±0.5 | 5.7±0.4 | 5.2±0.5 | 5.4±0.3 | 5.6±0.8 | 3.6±0.6 | 3.6±0.5 |
| **VS (% DS)** | 60.8±1.7 | 60.6±3.8 | 61.7±3.6 | 60.7±1.7 | 63.4±1.9 | 62.0±2.7 | 60.8±2.7 | 63.0±2.1 |
| **pH** | 7.8±0.1 | 7.6±0.1 | 7.7±0.1 | 7.1±1.7 | 7.7±0.1 | 7.5±0.1 | 7.6±0.1 | 7.6±0.1 |
| **VFA (mg.L$^{-1}$)** | 309.8±64.1 | 237.7±71.5 | 264.9±95.6 | 224.0±66.2 | 209.4±44.5 | 217.4±56.8 | 276.6±143.6 | 360.7±187.8 |
| **Alkalinity (mg.L$^{-1}$)** | 7,401.7±360.1 | 6,551.9±491.3 | 7,104.6±488.3 | 6,228.7±538.3 | 7,331.2±543.9 | 6,230.4±571.1 | 5,274.7±1,055.7 | 5,295.8±1,047.4 |
| **Lipids (% DS)** | 5.8±0.7 | 6.2±1.3 | 6.5±0.9 | 7.1±1.7 | 7.0±1.0 | 8.0±2.0 | | |
| **CST (min.g DS$^{-1}$)** | 194.2±39.9 | 156.5±33.4 | 288.7±107.2 | 179±35.3 | 318.3±63.4 | 270.5±47.7 | | |
| **VSd (%)** | 58.5±2.2 | 52.4±3.7 | 55.4±3.5 | 54.1±1.6 | 56.5±1.8 | 52.4±2.4 | 51.2±4.9 | 51.8±6.0 |
| **Total lipids deg. (%)** | 75.2±1.3 | 71.4±2.4 | 75.6±2.1 | 74.9±1.1 | 74.2±2.2 | 73.1±1.1 | | |
| **FOG deg. (%)** | | | 77.5±7.3 | 83.5±5.8 | 68.9±8.5 | 77.1±5.2 | | |
| **Biogas (L STP.g VS$_{added}$$^{-1}$)** | 484.9±27.5 | 463.7±45.4 | 488.5±29.4 | 513.4±46.0 | 498.3±25.0 | 502.7±48.4 | 429.3±89.8 | 494.5±73.7 |
| **$CH_4$ (L STP.g VS$_{added}$$^{-1}$)** | 293.3±14.6 | 289.2±27.1 | 305.3±17.3 | 331.6±28.4 | 307.6±15.5 | 324.4±28.6 | 274.6±59.8 | 311.4±40.1 |
| **$CH_4$ (L STP.g VS$_{destroyed}$$^{-1}$)** | 502.0±28.4 | 552.6±41.3 | 552.7±36.2 | 612.7±51.1 | 544.3±28.6 | 619.6±46.9 | 548.7±151.0 | 596.3±100.6 |
| **$CH_4$ (L STP.g VS FOG$_{added}$$^{-1}$)** | | | 639.6±263.6 | 912.1±134.7 | 705.4±332.4 | 834.5±147.8 | | 843.0±484.7 |
| **$CH_4$ (L STP.g FOG lipids$_{added}$$^{-1}$)** | | | 671.6±241.5 | 878.3±329.2 | 705.4±332.4 | 798.4±184.3 | | |
| **$CH_4$ (L STP.g FOG lipids$_{destroyed}$$^{-1}$)** | | | 912.1±134.7 | 1,102.8±212.0 | 1,048.1±533.1 | 1,043.5±274.5 | | |

#### 5.3.4.2 Co-digestion with non-hydrolysed sewage sludge

A digestion run was conducted to evaluate the performance of anaerobic co-digestion of GRU-FOG and non-hydrolysed sewage sludge. Two reactors were operated under the same conditions. The control (R4) was fed only with sludge whilst the co-digester (R5) was fed daily with FOG at 5% VS. Methane yields of 274.6±59.8 mL.g $VS_{added}^{-1}$ were obtained for R4 which were comparable to those of R1 measured at 291.6±20.4 mL.g $VS_{added}^{-1}$ during the two runs.

The daily addition of FOG to an anaerobic digester treating sewage sludge led to an increase of 11% in biogas production at 311.4±40.1 STP mL.g $VS_{added}^{-1}$ (Table 5-6). Biomethane yields were increased by 10% and 6% in terms of $VS_{added}$ and $VS_{destroyed}$ compared to the control digester. The addition of FOG generated 843.4±484.7 STP mL.g VS $FOG_{added}^{-1}$ being relatively comparable to average yields obtained with R2 and R3 using TH sludge at 875.2±143.1 STP mL.g VS $FOG_{added}^{-1}$.

In this study, the highest FOG loading rate achieved without digester failure was 7.5% in terms of VS. The addition of FOG to TH sludge increased the methane yields to 328.0±28.1 STP mL $CH_4$.g $VS_{added}^{-1}$ corresponding to a 10% compared to the control. Similarly, FOG added to non-hydrolysed sludge at 7.5% in terms of VS increased biomethane yields by 8% compared to that of the control. However, using TH sludge allows operation at higher OLR, at 4.9 kg $VS.m^{-3}.d^{-1}$, than non-hydrolysed sludge, at 2.6 kg $VS.m^{-3}.d^{-1}$. Consequently, this enables the treatment of larger volumes of FOG through anaerobic digestion and increase in biogas generation. Importantly, the use of TH on the sewage sludge did not lead to a reduction in the utilisation of the added FOG.

Experiments conducted on GTW demonstrated similar trends with increased methane yields ranging from 24-38% with the addition of 10-18% VS (Grosser and Neczaj, 2016), up to 66% with the addition of 48% VS (Kabouris et al., 2009a), 28 and 40% when adding 25% and 46% VS (Yalcinkaya and Malina, 2015a) and 24% with the addition of 52% of GTW in terms of VS (Ziels et al., 2016). Compared next to each other, these studies show a plateau phase is reached with the addition of 20% of FOG in terms of VS corresponding to an increase of 30% in methane generation (Figure 5-3). Relatively good FOG degradation rates were obtained in this study with TH sludge at loadings of 7.5% VS. It is speculated that the more FOG added to the system the lower the degradation rates. Further research is suggested to identify optimum degradation rates, and in turn operating safe boundaries.

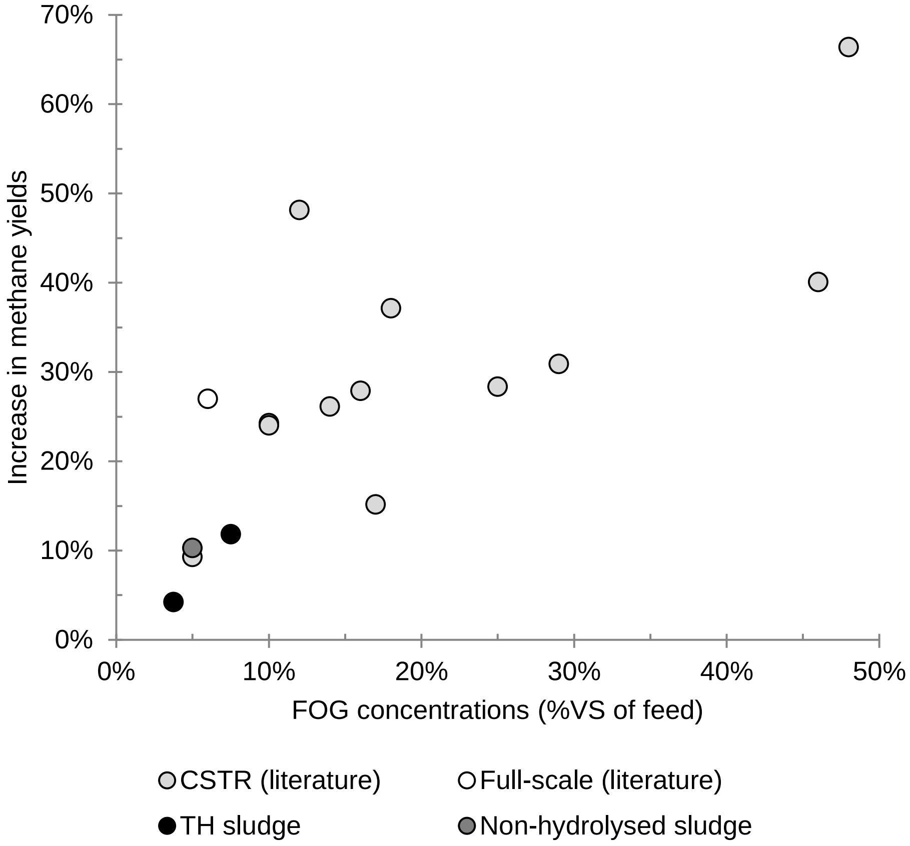

**Figure 5-3** Increase in methane generation from FOG co-digestion as reported in this study and in literature. Increased in methane yields were calculated based on their respective control.

## 5.4 Conclusions

Anaerobic co-digestion with TH sludge did not lead to a reduction in the utilisation of added FOG. However, using TH sludge allowed operation at higher OLR compared to non-hydrolysed sludge enabling the treatment of larger volumes of FOG and increase biogas generation. Whilst this study clearly demonstrated the value of GRU-FOG as a resource, there is a broad need for more research to be conducted to determine the safe operational boundaries with an emphasis on dewatering ability of co-digestate at large scale.

## 5.5 Acknowledgments

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# 6 Assessing the potential of enhanced primary clarification to manage fats, oils and greases (FOG) at wastewater treatment works

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## Abstract

Daily, sewage treatment works (STWs) receive large volumes of fats, oils and greases (FOG), by-products of food preparation. To increase FOG removal at STW, conventional primary sedimentation tanks (PSTs) can be enhanced using chemical coagulant or through dissolved air flotation (DAF) techniques. This work aimed to assess the potential benefits of enhanced primary treatment for FOG removal through an energy and costs analysis. To achieve this, a five-year sampling programme was conducted monthly at 15 STWs measuring FOG concentrations in crude and settled sewage (i.e. after primary treatment). In addition, two DAF pilot systems were trialled for four months and their performance, in terms of FOG removal, was assessed and compared to that of a control primary clarifier. Across the 18 STWs, influent FOG concentrations were found at 57.2±11.5 mg.L$^{-1}$. Chemical coagulants dosed prior to PSTs increased FOG removal rates on average to 71% whilst traditional sedimentation only achieved 52% removal. Effluent FOG concentrations were found between 14.1-21.9 mg.L$^{-1}$ and 18.2-35.8 mg.L$^{-1}$ respectively. By contrast, DAF achieved FOG effluent concentrations on average at 10.3±3.7 mg.L$^{-1}$ corresponding to 74% removal from a relatively low influent concentration of 40.3±29.8 mg.L$^{-1}$. Thus, enhanced primary treatments have the potential to reduce organic load to secondary treatment and increase energy generation through anaerobic digestion. The overall net energy balance was estimated at 2,269 MWh.year$^{-1}$ for the DAF compared to 3,445 MWh.year$^{-1}$ for the chemically-enhanced PST making it a less financially attractive alternative. Yet, in the case where the works require upgrading to accommodate flow or load increases, DAF appeared as a sensible option over sedimentation offering significantly lower capital costs and footprint. In relation to FOG management, upgrading all STWs is not realistic and will require understanding where the benefits would be the highest.

**Keywords**: Dissolved air flotation (DAF); sewage treatment works (STWs); hexane extractable material (HEM); primary sedimentation tanks (PSTs)

## 6.1 Introduction

Daily, large volumes of fats, oils and greases (FOG), by-products of food preparation, are believed to reach sewage treatment works (STWs). FOG not only causes pipe blockages within the sewers but also disrupts settlement and clarification processes at STWs hindering treatment efficiency (Wallace et al., 2017). In addition, FOG exerts an extra load of organic matter onto the secondary aerobic treatment stage thereby increasing the overall aeration demand. Whilst the long-chain fatty acids, which make up the majority of the FOG, can be consumed under both aerobic and anoxic conditions, kinetic studies showed that these fatty acids were degraded at a much slower rate than sugars and other substrates (Chipasa and Mędrzycka, 2006; Novak and Kraus, 1973). Consequently, FOG can accumulate within the reactors potentially enhancing the risk of foaming through stimulating the growth of filamentous microorganisms (Lefebvre et al., 1998). To avoid their detrimental impacts on downstream processes, FOG can be separated from the wastewater at the front end through a preliminary treatment step. In addition, the collected FOG is a rich energy source which can be valorised through anaerobic digestion with reported increases in methane yield of up to 138% with the addition of 23% of FOG on volatile solids (VS) basis to sewage sludge (Silvestre et al., 2011).However, separation of FOG through a preliminary stage is not always a viable option for the treatment of municipal wastewater as inclusion requires the installation of additional assets (Pastore et al., 2015). .

In relatively recent years, enhanced primary treatment has been introduced through the use of coagulant dosing prior to sedimentation to increase solids and/or phosphorus removal. In addition, the use of dissolved air flotation (DAF) is being considered as an alternative process. DAF is commonly used in drinking water and industrial waste treatment and works by injecting air saturated pressurised water into the tank. This results in the formation of a large mass of small bubbles (40-60 μm) which combine with the solids reducing their density and causing them to float to the surface where they are removed (Edzwald, 2010). The technology is particularly effective against low density solids and hence it is posited offer real potential for FOG removal at STW. To illustrate, in FOG-rich industrial wastewaters, removal levels of 89 and 98% have been reported from slaughterhouse effluents (Al-Mutairi et al., 2008; Karpati and Szabo, 1984; Travers and Lovett, 1985), 60% from dairy wastewaters (Monroy et al., 1995) and up to 97% in effluents from meat-manufacturing plants (El-Awady, 1999). In comparison, only a few studies have reported FOG removal efficiencies using DAF in urban wastewaters, ranging from 28% up to 72% (Kuo and Goh, 1992; Levy et al., 1972). The paucity of reported municipal cases reflects the combination of increased maintenance/operational complexity and energy demand associated with bubble generation. However, since then, technologies have become available with more optimised recycle systems and new methods of forming microbubbles (Crossley and Valade, 2006).

This work aimed to assess the potential benefit of enhanced primary treatment on FOG removal and establish the energy and operating cost basis for its potential inclusion. To achieve this, an extensive sampling programme was conducted at 15 STWs over a five-year period and FOG concentrations were measured in crude and settled sewage (i.e. after primary treatment). The sites were predominantly traditional sedimentation tanks with four sites upgraded to include chemical dosing. In addition, two DAF pilot systems were trialled for four months and their performance, in terms of FOG removal, was assessed and compared to that of a control primary clarifier. Originally intended to be installed on one of the sites monitored, the DAF plants were trialled at a different STW due to site restraints. The results from both the extensive monitoring and the DAF plants were utilised in an economic analysis to assess the potential of advanced techniques as an alternative to conventional primary sedimentation tanks (PSTs) for FOG removal.

## 6.2 Material and methods

### 6.2.1 Process operation

Spot samples of crude sewage were taken monthly at 15 STWs owned by Thames Water Utilities Ltd. (TWUL) (sites 2 to 16) as part of a routine sampling. Settled sewage samples were also collected after PSTs (Table 6-1). Removal rates were calculated from averaged concentrations and presented with their associated propagation of uncertainties. This sampling programme was conducted over a period of five years (2013 to 2018). For each site, the surface overflow rate (SOR) was calculated as follows:

$$SOR = \frac{Average\ daily\ flow}{A} \tag{6-1}$$

Where: the average daily flow received at STWs is the dry weather flow (DWF) multiplied by a factor 1.2 and expressed in $m^3.d^{-1}$ (as commonly employed in the industry) and $A$ is the surface area of primary clarifiers in $m^2$.

Desludging from all primary treatments was achieved based on cycles controlled by timer. Ferric sulphate was dosed at concentrations around 30 $mg.L^{-1}$ (based on TWUL asset standards) upstream of the primary treatment for phosphorous removal at sites 1 to 5.

**Table 6-1** Key parameters and concentrations of organics for the sampled STWs. Concentrations are expressed as averages with their associated standard deviation. Sites 2 to 16 were monitored over a period of five years; site 1 was monitored for four months during the DAF trials. PST type is defined either as conventional (Conv.) or chemically-enhanced (CE).

| | | | | | | | **Concentrations in crude sewage (mg.L$^{-1}$)** | | | | **Concentrations in effluent from primary treatment (mg.L$^{-1}$)** | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **STW** | PE | DWF (m$^3$.d$^{-1}$) | Solids loading (kg.d$^{-1}$) | SOR (m.h$^{-1}$) | PST type | Per cap. HEM (g.d$^{-1}$) | HEM | tCOD | BOD$_5$ | SS | HEM | tCOD | BOD$_5$ | SS |
| **1** | 20,090 | 3,760 | 364 | 0.33 | CE | 7.5 | 40±30 | 452±247 | 154±84 | 290±133 | 14±7 | 169±49 | 59±18 | 91±37 |
| **2** | 123,820 | 21,970 | 2,415 | 0.69 | CE | 9.3 | 53±27 | 574±221 | 228±82 | 330±135 | 12±10 | 222±96 | 85±40 | 100±155 |
| **3** | 130,580 | 24,570 | 1,910 | 0.43 | CE | 11.9 | 63±51 | 724±428 | 274±233 | 466±589 | 16±13 | 232±87 | 82±39 | 135±145 |
| **4** | 156,840 | 25,590 | 1,193 | 0.39 | CE | 10.3 | 63±36 | 658±253 | 247±83 | 420±258 | 18±15 | 252±85 | 92±40 | 78±30 |
| **5** | 927,830 | 205,740 | 7,726 | 0.94 | CE | 13.4 | 60±46 | 578±169 | 212±57 | 375±129 | 22±14 | 251±61 | 111±62 | 100±49 |
| **6** | 89,160 | 15,240 | 1,153 | 0.39 | Conv. | 10.3 | 60±29 | 670±222 | 265±82 | 303±94 | 32±18 | 411±122 | 159±66 | 116±40 |
| **7** | 411,980 | 100,180 | 5,221 | 0.71 | Conv. | 14.4 | 59±43 | 633±240 | 218±80 | 417±199 | 25±15 | 319±113 | 127±44 | 165±177 |
| **8** | 145,410 | 30,020 | 4,277 | 0.74 | Conv. | 11.9 | 58±37 | 670±246 | 203±72 | 427±224 | 36±24 | 413±237 | 150±62 | 209±203 |
| **9** | 180,230 | 53,600 | 4,161 | 0.88 | Conv. | 11.4 | 38±37 | 340±126 | 130±44 | 233±340 | 19±14 | 208±76 | 79±25 | 95±116 |
| **10** | 888,100 | 192,200 | 6,119 | 1.02 | Conv. | 10.6 | 49±23 | 507±139 | 209±55 | 255±69 | 23±13 | 250±73 | 98±29 | 76±21 |

| | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **11** | 166,770 | 28,910 | 1,695 | 0.68 | Conv. | 13.4 | 77±50 | 663±377 | 262±97 | 352±242 | 31±18 | 409±154 | 162±48 | 179±129 |
| **12** | 221,660 | 52,910 | 3,921 | 0.82 | Conv. | 18.0 | 75±42 | 409±154 | 287±88 | 439±192 | 34±26 | 398±184 | 153±77 | 209±125 |
| **13** | 425,890 | 88,960 | 4,052 | 0.71 | Conv. | 9.6 | 46±28 | 678±186 | 206±64 | 364±128 | 19±12 | 328±69 | 117±33 | 104±32 |
| **14** | 406,400 | 73,520 | 3,283 | 0.36 | Conv. | 11.5 | 64±31 | 774±313 | 253±84 | 536±293 | 23±21 | nd | nd | nd |
| **15** | 227,040 | 42,640 | 1,313 | 0.14 | Conv. | 10.6 | 56±24 | 640±257 | 251±79 | 370±291 | 35±21 | nd | nd | nd |
| **16** | 121,150 | 28,390 | 1,614 | 0.67 | Conv. | 9.1 | 39±23 | 481±186 | 170±76 | 284±180 | 26±21 | 258±94 | 92±33 | 103±54 |

nd: not determined.

To investigate the performance of flotation techniques to remove FOG, two flotation pilot-scale systems were trialled at a municipal STW with a population equivalent (PE) of 20,090 (site 1). Unlike sites 2 to 16, STW 1 did not have any access restriction and was therefore an ideal candidate for the DAF trial. The primary treatment of wastewater was achieved via three parallel chemically-enhanced (CE) PSTs; ferric sulphate was dosed at around 25 mg.L$^{-1}$ for phosphorus removal into the PST distribution chamber. One of the PSTs was used as a control for this study. To reduce the amount of coagulant dosed into the pilot plants, a baffle was installed near the feeding point in the distribution chamber (Figure 6-1). As a consequence, dosing in the control-PST was reduced. Auto-desludging pumps were run for 5 minutes every 3 hours.

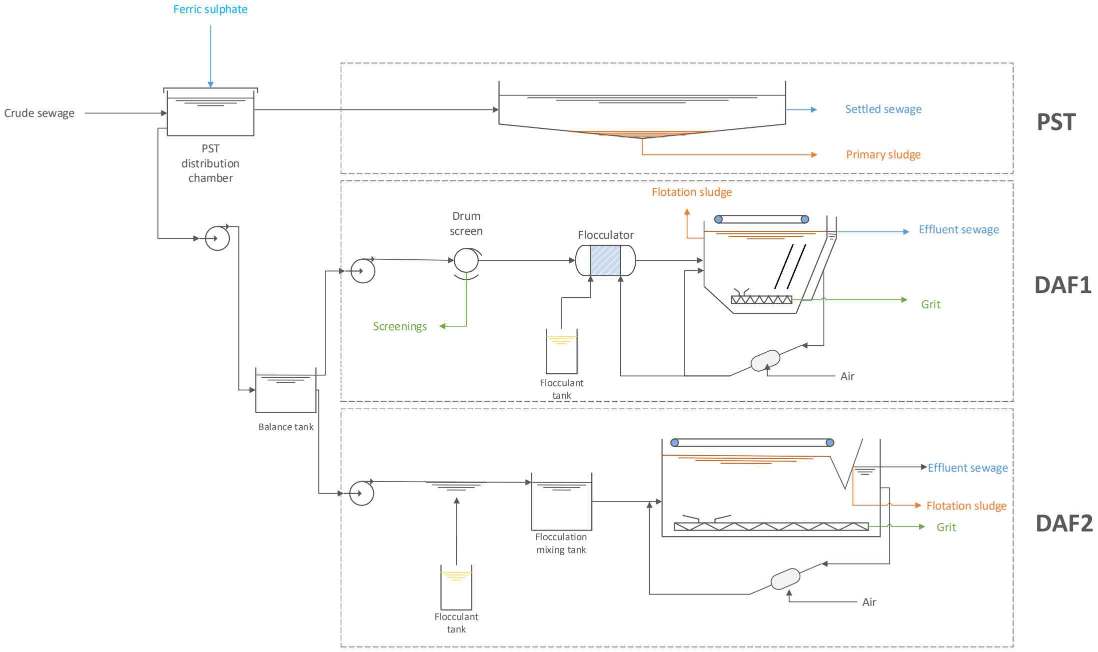

**Figure 6-1** Schematic of the pilot-scale trial.

Crude sewage was pumped from the control-PST feeding point into a balance tank from where sewage was fed to both flotation units (Figure 6-1). The main differences between the two pilot-scale systems were the operation of DAF2 at a lower water pressure of 3.5 bar, and DAF1 being fitted with lamella plates increasing its effective surface area from 0.7 m$^2$ to 2.9 m$^2$ (Table 6-2). DAF effluents were discharged into the drain and recirculated through the treatment works.

For coagulation/flocculation, two different polyacrylamide-based polyelectrolyte aids were used. Flocculant A, recommended by one of the pilot plant manufacturers, is characterised as a cationic medium charge, high molecular weight polymer. In comparison, flocculant B was characterised as high anionic charge, very high molecular weight polymer. Dilute water solutions were made from dry polymers in 200 L round tanks. These solutions were renewed daily once used up. The dosage of polymer added was 1.5 mg of active substance per L of sewage. For DAF1, dilute solutions were dosed into the sewage using a peristaltic pump. Coagulation/flocculation was achieved in a tubular contact zone prior to the flotation unit. In the case of DAF2, dilute solutions of polymer were dosed into a coagulation/flocculation tank (450 L) equipped with a mixer whose rotation speed could be adjusted.

**Table 6-2** Control-PST and DAF operating parameters.

| **Parameter** | **PST** | **DAF1** | **DAF2** |
|---|---|---|---|
| **Flow treated ($m^3.d^{-1}$)** | 1,221 | 120 | 192 |
| **Influent solids load ($kg.d^{-1}$)** | 354 | 35 | 56 |
| **Screens** | N/A | 2 mm | N/A |
| **Recirculated water pressure (bar)** | N/A | 6 | 3.5 |
| **Effective surface area ($m^2$)** | 170 | 2.9 | 4.8 |
| **Recycling ratio (% of inlet flow rate)** | N/A | 25% | 25% |
| **Bubble size (µm)** | N/A | 20 to 40 | 10 to 70[1] |
| **Surface overflow rate ($m.h^{-1}$)** | 0.3 | 1.7 | 1.7 |
| **Energy consumption ($kWh.m^{-3}$)** |  | 0.06 | 0.07 |
| **Air to solids ratio** |  | 0.08 | 0.04 |

[1] with 90% being between 20 and 50 µm according to the manufacturer

### 6.2.2 Analytical methods

Sewage samples were analysed for total chemical oxygen demand (tCOD), biochemical oxygen demand ($BOD_5$) and suspended solids (SS) according to APHA methods (APHA, 2005). Total P was measured through inductively coupled plasma mass spectrometry (ICP-MS) using a Thermo Scientific$^{TM}$ iCAP$^{TM}$ 5200 DV. The determination of FOG in these samples was achieved by filtration, solvent extraction and gravimetry (HM Stationery Office, 1987). Briefly, wastewater samples were collected in 1 L glass bottles and filtered using a Whatman® GF/C grade filter paper. The filter paper was immersed in boiling hexane (around 50˚C), using a SOXTHERM® extraction unit, in a pre-weighted glass extraction beaker. After the solvent reduction program had run, the solvent was evaporated from beakers before being reweighed. Oil and grease concentrations were determined by weight difference. For the clarity of this paper, these results were referred to as hexane extractable material (HEM). The reporting limit of detection for this analysis was 8.2 mg.L$^{-1}$. These analyses were performed by UKAS 17025 accredited TWUL laboratories.

During the DAF pilot-scale experiments, sewage sludge samples were regularly taken from the control-PST during auto-desludging cycles and the flotation plants. Dry solids (DS) and VS were analysed according to APHA methods (APHA, 2005). The lipids content of sewage sludge was measured using a modified Wiebul acid hydrolysis method (Sciantec Analytical, 2018c).

To allow comparison between processes, HEM concentrations were normalised based on sludge produced ($Q_S$) as follows:

$$X_{HEM} = \frac{(HEM_{in} \times Q_i) - (HEM_{out} \times Q_{out})}{Q_S} \qquad \textbf{(6-2)}$$

Where $HEM_{in}$ and $HEM_{out}$ are HEM concentrations measured in influent and effluent (g.m$^{-3}$), $Q_i$ is the inlet flow (m$^3$.d$^{-1}$), and $Q_{out}$ is the outlet flow (m$^3$.d$^{-1}$). The sludge production was calculated as follows:

$$Q_S = \frac{(SS_{in} - SS_{out}) \times Q_i}{\%DS} \qquad \textbf{(6-3)}$$

Lipids concentrations measured in sludge were normalised based on *Qs*:

$$X_{sludge} = \frac{(SS_{in} - SS_{out}) \times Q_i \times X_{lipids}}{Q_S} \tag{6-4}$$

Where $SS_{in}$ and $SS_{out}$ are SS concentrations measured in influent and effluent (g.m$^{-3}$), and $X_{lipids}$ is the lipids concentrations in sludge (as %DS).

### 6.2.3 Economic evaluation

A case study was used to investigate the economic viability of retrofitting conventional clarifiers with DAF technologies at a hypothetical STW serving a PE of 500,000. Wastewater flow was assumed at 0.2 m$^3$.PE$^{-1}$ per day (Henze and Comeau, 2008). Incoming BOD$_5$, SS and FOG loads, as well their removal rates from primary clarifiers and DAF, were estimated based on average values collected for sites 2 to 16 (Table 6-5). The CE-PST (low dose) scenario assumed lower chemical dose at 10 mg.L$^{-1}$ would be needed only for FOG removal compared to the CE-PST (high dose) using around 30 mg.L$^{-1}$ for phosphorous removal. The DAF scenario was based on removal rates obtained with DAF2-FlocB at 67%, 75% and 74% respectively for BOD$_5$, SS and HEM (i.e. removal rate achieved with lower HEM influent concentrations). The DAF – cost neutral scenario was developed assuming BOD$_5$, SS and HEM concentrations of 51 mg.L$^{-1}$, 74 mg.L$^{-1}$ and 10 mg.L$^{-1}$, obtained for DAF2-FlocB, would be achieved equating to removal of 77%, 81% and 82% based on average influent concentrations obtained from sites 2 to 16.

The base year of this economic evaluation was 2018. Some cost data was collected in EUR and converted at the rate EUR:GBP of 0.80 (2008) and EUR:GBP of 0.88 (2018). Cost indices were used to adjust for the difference in capital costs over time, using the Chemical Engineering Plant Cost Index, and upon location based on European Construction Costs (2019). The relationship used for cost indices was as follows:

$$Cost\ A = Cost\ B \times \frac{Index\ A}{Index\ B} \tag{6-5}$$

Index values for equipment costs in 2008 and 2018 were 575.4 and 603.1. Location factors used for the UK, Denmark and Germany were respectively 100, 145.4 and 96.6 (European Construction Costs, 2019).

Capital expenditure (CapEx) for DAF was based on costs provided by the manufacturer of DAF2 at £1.76M for a plant treating 1,250 $m^3.h^{-1}$ of sewage, and £0.10M for the associated dosing plant. CapEx for PST was adapted from COWI A/S (2010) and estimated at £53 per PE. Capital costs were annualised over their lifetime (*n*) at an interest rate (*i*) of 2.8% (Ofwat, 2017). DAF and PST were assumed with lifetimes of 50 years whilst that of dosing plant was 10 years. The annualised cost of capital (ACC) was calculated as follows:

$$ACC = CapEx \times \frac{i\,(1+i)^n}{i\,(1+i)^n - 1} \tag{6-6}$$

Operational expenditures (OpEx) from STWs were based on (1) primary treatment (chemical costs and energy demand), (2) aeration (energy demand) and (3) sludge conditioning (chemical cost for thickening and dewatering and cake transportation cost). Energy generation from anaerobic digestion was calculated based on the sludge output from primary treatments and any additional FOG removed (Tables 6-3 and 6-4).

**Table 6-3** List of assumptions used for the economic analysis (primary and secondary treatments).

| **Parameter** | **Value** | **Reference** |
|---|---|---|
| **1 – Primary treatment** | | |
| Energy consumption of PST | 0.62 $Wh.m^{-3}.d^{-1}$ | Newell (2012) |
| Energy consumption of CE-PST | 1.05 $Wh.m^{-3}.d^{-1}$ | Newell (2012) |
| Energy consumption of DAF | 70 $Wh.m^{-3}$ | DAF2 manufacturer |
| **Coagulant dose for CE-PST** | 17.3 $g.m^{-3}$ | adapted from TWUL asset standards |
| Cost of ferric sulphate | £344.4 per ton | Kemcore (2019) |
| **2 – Secondary treatment** | | |
| BOD of FOG | 1.8 kg $BOD.kg\ FOG^{-1}$ | adapted from Groenewold et al. (1982) |
| Secondary sludge production | 0.8 kg $SS.kg\ BOD^{-1}$ | TWUL internal data |
| $O_2$ demand for $BOD_5$ removal | 0.9 kg $O_2.kg\ BOD_5^{-1}$ | TWUL internal data |
| $O_2$ demand for endogenous respiration | 0.04 kg $O_2.kg\ MLSS^{-1}$ | TWUL internal data |
| Food to microorganisms ratio | 0.2 | TWUL internal data |
| Power requirement for aeration | 1.5 $kWh.kg\ O_2^{-1}$ | TWUL internal data |

**Table 6-4** List of assumptions used for the economic analysis (sludge conditioning and anaerobic digestion).

| Parameter | Value | Reference |
|---|---|---|
| **3 – Sludge conditioning** | | |
| Polymer dose for thickening/dewatering | 10 kg per ton DS | SNF Floerger (n.d.) |
| Thickening solids capture | 95% | Andreoli et al. (2007) |
| **4 – Anaerobic digestion** | | |
| Primary sludge destruction | 55% | Barber (2014) |
| Primary sludge biogas yield | 0.98 $m^3.kgVS\ destroyed^{-1}$ | Barber (2014) |
| Secondary sludge destruction | 30% | Barber (2014) |
| Secondary sludge biogas yield | 0.79 $m^3.kgVS\ destroyed^{-1}$ | Barber (2014) |
| COD of FOG | 2.8 g $COD.g\ lipids^{-1}$ | Labatut et al. (2011) |
| COD destruction of FOG | 44% | Labatut et al. (2011) |
| Biomethane yield | 0.35 $m^3.kg\ COD^{-1}$ | Angelidaki and Sanders (2004) |
| Calorific value of methane | 36 $MJ.m^{-3}$ | |
| Calorific value of biogas | 18 $MJ.m^{-3}$ | |
| Electrical conversion efficiency | 30% | Goss et al. (2017) |
| Transportation costs | £8.5 per $m^3$ | ADAS UK Ltd (2013) |

## 6.3 Results and discussion

### 6.3.1 Occurrence of FOG in crude sewage

Across the 18 STWs monitored, influent HEM concentrations ranged from 38±37 $mg.L^{-1}$ (site 9) up to 77±50 $mg.L^{-1}$ (site 11) with a median measured at 59 $mg.L^{-1}$ (Table 6-1). Great variations between spot samples were observed ranging from the minimum detection limit (8.2 $mg.L^{-1}$) up to 340 $mg.L^{-1}$. Sites 9, 15 and 16, reported the lowest HEM concentrations and also displayed the lowest median

$BOD_5$ concentrations respectively measured at 130±44 $mg.L^{-1}$, 206±64 $mg.L^{-1}$ and 170±76 $mg.L^{-1}$ (Table 6-1). Values across the sites were consistent with previous reported FOG levels which vary between 10 and 100 $mg.L^{-1}$ (Dehghani et al., 2014; Gelder and Grist, 2015; Pujol and Lienard, 1989; Quéméneur and Marty, 1994; Raunkjær et al., 1994; Wiltsee, 1998).

The reported concentrations equate to per capita contribution of HEM from 9.1 $g.d^{-1}$ up to 18.0 $g.d^{-1}$ with a median measured at 11.0 $g.d^{-1}$ (Table 6-1). In the UK, FOG production rates at source (i.e. from households and food outlets) have been estimated around 17 $g.capita^{-1}.d^{-1}$ (Chapter 2). To allow comparison, reported concentrations at STW require to be adjusted from the contribution of soaps, at 1.5 $g.capita^{-1}.day^{-1}$ (Ram et al., 2018), and lipids from faeces, at 4.1 $g.capita^{-1}.day^{-1}$ (Rose et al., 2015), as the use of a hexane extraction step within the procedure means that additional material will be included. Adjusting the measured data accordingly indicates that the actual contribution of FOG is within the range 3.5-12.4 $g.capita^{-1}.d^{-1}$ and a median value of 5.4 $g.capita^{-1}.d^{-1}$. It is posited that the difference reflects accumulation within the sewer network, potentially accounting for 69% of the FOG that enters the network. A stronger relationship was observed between influent HEM and $BOD_5$ concentrations compared to tCOD and SS concentrations. To illustrate, correlations of 0.79, 0.87 and 0.65 were determined for tCOD, $BOD_5$ and SS respectively (Figure 6-2). The close relationship existing between the $BOD_5$ and HEM could therefore indicate a good degradation by aerobic biological organisms over a specific period.

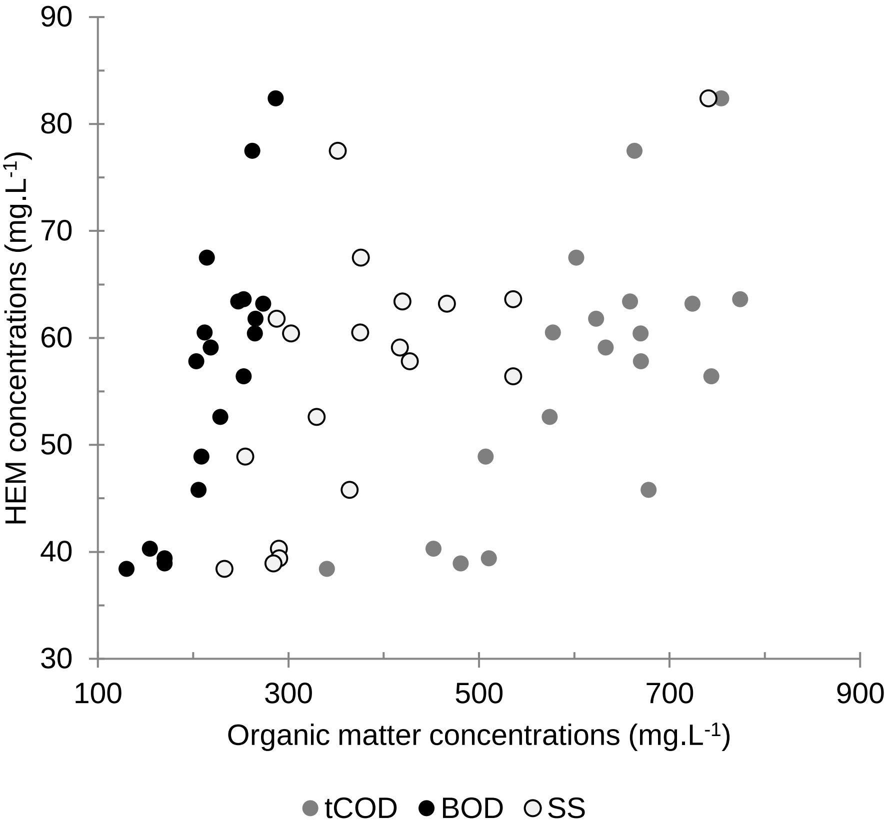

**Figure 6-2** HEM concentrations reported against tCOD, $BOD_5$ and SS concentrations measured in crude sewage for each STW.

### 6.3.2 Treatment performance

The lowest effluent HEM concentrations were measured for site 13 at 19±12 $mg.L^{-1}$ whilst the highest were reported for site 8 at 36±24 $mg.L^{-1}$ (Table 6-1). Corresponding removal efficiencies across the traditional sedimentation processes ranged between 33±4 and 64±8% (median 55%). In contrast, enhancing primary treatment through chemical dosing increased FOG removal to between 64±9% up to 76±9% with a median of 73%. This equated to effluent FOG concentrations of between 12±10 and 22±14 $mg.L^{-1}$. A one-way ANOVA resulted in a $F_{value}$ of 15.5 and a $F_{crit}$ of 4.7 at a confidence level of 95% (*p*-value of 0.002) such that there was a significant difference between HEM removal rates from conventional and CE-PSTs. The current results reflect a higher overall removal than observed during previous studies that reported FOG removal rates from conventional PSTs at US STWs which were between 45% (Loehr and de

Navarra Jr., 1969; Murcott, 1992) and 47% (Gehm, 1942). With respect to chemical dosing, previous trials reported removal of 59% (Kuo and Goh, 1992) and 71% (Murcott, 1992). SOR were substantially higher in trials conducted by Kuo and Goh (1992) at between 1.5 $m.h^{-1}$ to 3.0 $m.h^{-1}$. However, FOG removal did not correlate with SOR (Figure 6-3), a finding supported by Loehr and de Navarra Jr. (1969). It is posited that removal reflects more the efficacy of dosing in relation to dosed amount and mixing conditions, observations that are commonly reported with respect to coagulation of drinking water (Fearing et al., 2004) and tertiary treatment of sewage (Murujew et al., 2020).

Similarly, tCOD, $BOD_5$ and SS removal efficiencies were found significantly higher with CE-PSTs. To illustrate, removal efficiencies across conventional PSTs ranged between 38-52% for tCOD (median 46%), 26-53% for $BOD_5$ (median 42%), and 49-72% for SS (median 62%). By contrast, CE-PSTs achieved tCOD removals between 56-68% (median 62%), 48-70% for $BOD_5$ (median 63%), and 70-81% for SS (median 72%). A stronger relationship was observed between HEM and both tCOD and $BOD_5$ removal rates, with correlation of 0.79 and 0.77, than with SS determined at 0.55 (Figure 6-4).

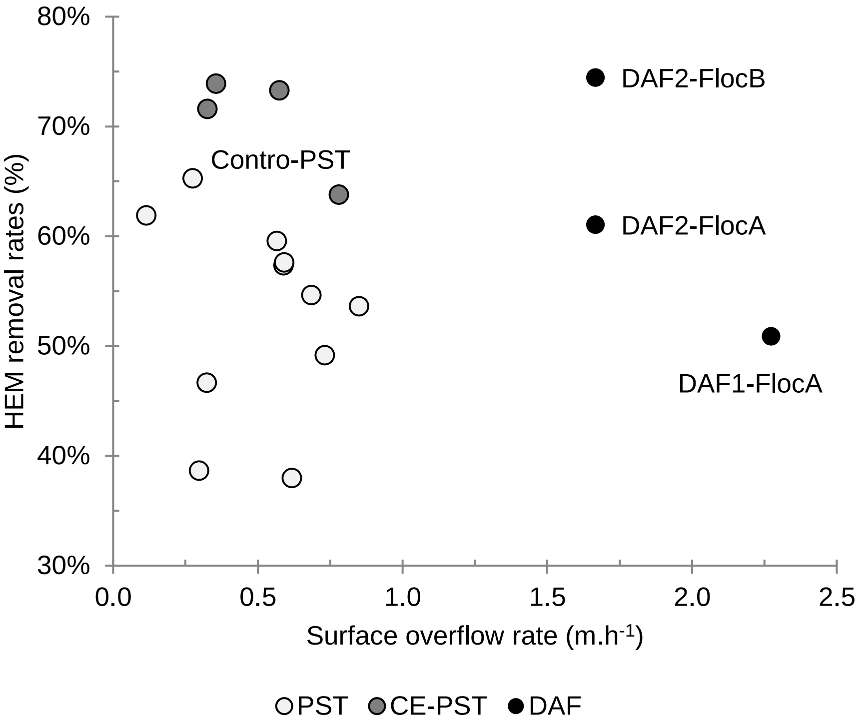

**Figure 6-3** HEM removal rates reported against SOR for each site.

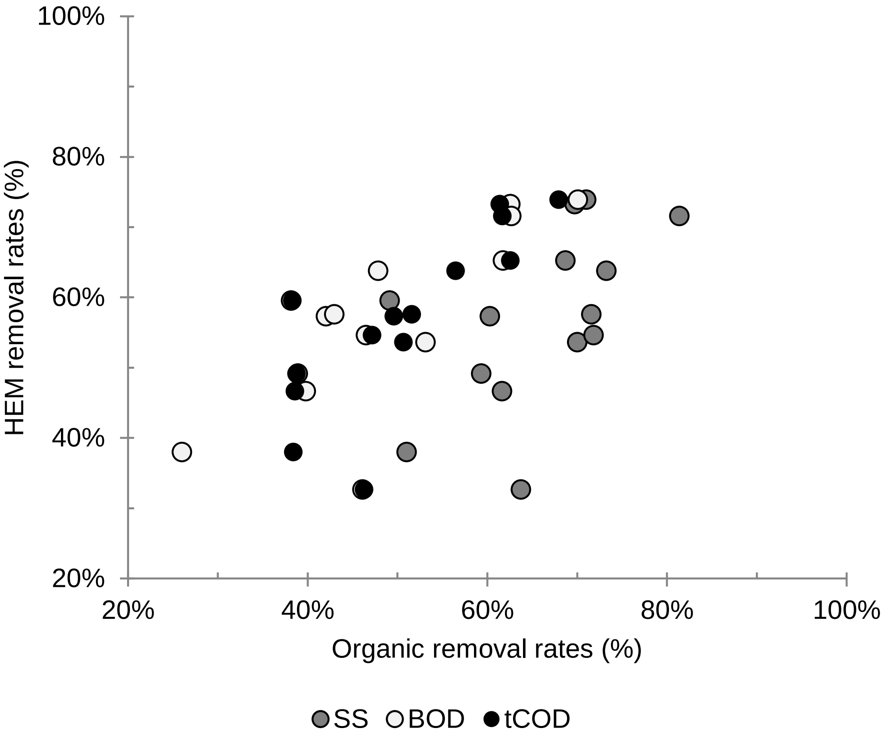

**Figure 6-4** HEM removal rates reported against tCOD, $BOD_5$ and SS removal rates for each site.

### 6.3.3 DAF pilot-scale experiments

Two DAF pilot-scale systems were trialled with the aim to compare their performance to that of PSTs gathered during the extensive sampling. Comparison of the control CE-PST and the three DAF trials revealed HEM removal efficiencies of 65±10%, 51±12%, 61±11% and 74±10% for the CE-PST, DAF1-FlocA, DAF2-FlocA and DAF2-FlocB (Table 6-5). The corresponding effluent concentrations were 14±7, 20±12, 16±8 and 10±4 mg.L$^{-1}$ from a relatively low influent concentration of 40±30 mg.L$^{-1}$. There was a significant difference between effluents from the control-PST and DAF2-FlocB at a confidence level of 90% (one-way ANOVA, *p*-value 0.07). Accordingly, the nature of the polymer appeared to have a significant impact on the efficacy of the processes with the best results observed for the anionic, very large molecular weight polymer. The importance of appropriate polymer selection has been reported before with charge, size and structure all known to influence the outcome as polymers are able to work through a number of different mechanisms such as charge neutralisation, steric hindrance and bridging (Murujew et al., 2020).

The levels reported for DAF2-Floc B were comparable to previous reported FOG removal rates in municipal sewage at 72% (Kuo and Goh, 1992) and from FOG-rich industrial wastewaters (Jensen et al., 2014; Monroy et al., 1995). Whilst DAF2-FlocB achieved relatively comparable performance in removing FOG as CE-PSTs, the process was operated at much higher SOR providing significant opportunities in terms of footprint reduction (Figure 6-3). In addition, it should be noted that chemical dosing is included to improve solids or phosphorus removal and not specifically FOG. Comparison during the trial revealed improved solids removal with the DAF compared to the CE-PST at 75±7% and 69±5% respectively but slightly poorer phosphorus removal at 49±4% compared to 54±3% respectively. Removal efficiencies of tCOD and BOD were also slightly higher for the DAF plant but the greatest difference was observed with regards to HEM.

**Table 6-5** Influent and effluents characteristics for HEM, $BOD_5$, COD and SS. Removal rates were calculated based on average concentrations in influent and effluents. HEM removed and lipids in sludge were calculated, and are expressed with their associated uncertainties.

| **Parameter** | **Inlet** | **control-PST** | **DAF1-FlocA** | **DAF2-FlocA** | **DAF2-FlocB** |
|---|---|---|---|---|---|
| **HEM** | 40±30<br>n=47 | 14±7<br>65%<br>n=22 | 20±12<br>51%<br>n=9 | 16±8<br>61%<br>n=11 | 10±4<br>74%<br>n=17 |
| **$BOD_5$** | 154±84<br>n=88 | 59±18<br>62%<br>n=83 | 66±19<br>57%<br>n=19 | 67±37<br>64%<br>n=20 | 51±13<br>67%<br>n=26 |
| **tCOD** | 452±247<br>n=77 | 169±49<br>62%<br>n=69 | 185±54<br>59%<br>n=19 | 173±91<br>62%<br>n=20 | 158±41<br>65%<br>n=13 |
| **SS** | 290±133<br>n=89 | 91±37<br>69%<br>n=84 | 96±27<br>67%<br>n=19 | 92±35<br>68%<br>n=20 | 74±27<br>75%<br>n=26 |
| **Total P** | 8.2±3.5<br>n=66 | 3.8±0.7<br>54%<br>n=63 | 4.1±0.9<br>50%<br>n=16 | 3.9±1.1<br>52%<br>n=17 | 4.2±1.0<br>49%<br>n=12 |
| **DS (%)** | | 3.1±1.0 | 6.6±1.4 | 7.1±1.1 | 4.9±1.4 |
| **HEM removed (kg.m$^{-3}$ sludge)** | | 3.4±0.4 | 7.0±1.5 | 9.3±1.6 | 6.6±0.7 |
| **Lipids in sludge (kg.m$^{-3}$ sludge)** | | 2.3±0.2 | 5.7±0.8 | 8.8±1.1 | 6.7±0.9 |

The sludge produced from the different primary treatments had a DS level of 3.1±1.0%, 6.6±1.4%, 7.1±1.1% and 4.9±1.4% for the control-PST, DAF1-FlocA, DAF2-FlocA and DAF2-FlocB respectively. Accordingly, flocculant A appeared to be more appropriate for dewatering rather than primary removal. Lipid analysis revealed that not only was the sludge from control-PST less concentrated but it also contained fewer lipids. To illustrate, lipids concentrations as a fraction of the DS were 7.0±3.0% for the control-PST compared to 9.1±2.9% for DAF1-FlocA, 12.2±4.3% for DAF2-FlocA and 13.0±6.6% for DAF2-FlocB (Table 6-5). A one-way ANOVA showed there were significant differences, at a confidence interval of 99% ($p$-value = $7.10^{-5}$), in the lipids content of control-PST sludge and DAF2 flotation sludge. Comparison to literature revealed relatively low levels in the current study with reported ranges of 6.2 up to 19.4% DS with an average at 10.8% DS for primary sludge from sedimentation (Barber, 2014; Giacalone, 2017; Gonzalez, 2006) and 20.0-44.1% of DS for flotation sludge (Donoso-Bravo and Fdz-Polanco, 2013; Perez et al., 2012; Silvestre et al., 2011). A few authors have reported very high levels of up to 94.5% with a median of 31.7% in terms of DS for FOG harvested at STWs (Martín-González et al., 2011; Williams et al., 2012; Chapter 2). It is therefore possible that the low levels reported here reflect the low influent FOG concentrations in the sewage. To verify this hypothesis, HEM removed and lipids in sludge were normalised based on $m^3$ of sludge produced (Table 6-5). The DAF pilot-scale systems were found better at removing FOG, generating between 5.7±0.8 to 8.8±1.1 kg.lipids.$m^{-3}$ sludge produced, compared to the control-PST calculated at 2.3±0.2 kg lipids.$m^{-3}$ sludge produced (Table 6-5). For the DAF plants, normalised qualities of lipids in sludge represented 82% and 101% of the quantities of lipids found in sludge confirming that the final concentrations were fed limited. In the case of DAF1, sampling before and after the screens indicated removal of 16%, 32% and 32% of the incoming $BOD_5$, tCOD and SS loads respectively. Consequently, this also had a direct impact on the sludge quality reducing lipids content.

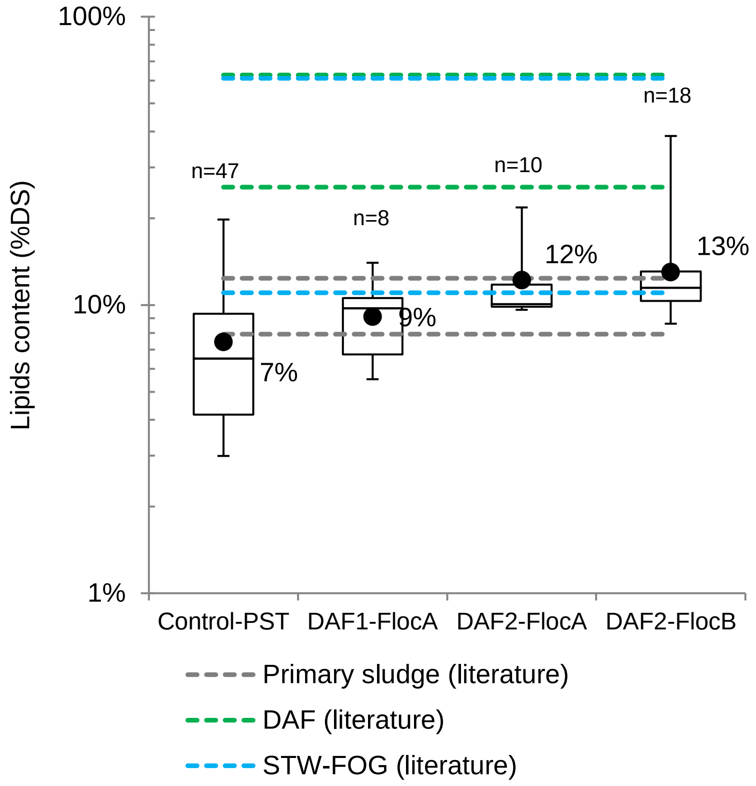

**Figure 6-5** Lipids content (in dry basis) measured in primary sludge from the control-PST and DAF pilot-scale systems. Values from literature are represented with their first and third quartiles. Lipids contents from DAF sludge at STW are reported as "DAF (literature); "STW-FOG (literature)" describes FOG collected at STW.

### 6.3.4 Economic evaluation

An economic evaluation was performed at a hypothetical STW serving 500,000 PE relying on conventional sedimentation techniques (i.e. baseline scenario). Excluding the energy demand of the process, the impact of using enhanced primary treatment in terms of the energy gain revealed a net positive change of 3,460 MWh.year$^{-1}$ and 4,801 MWh.year$^{-1}$ for the CE-PST and DAF systems respectively (Table 6-5). In both cases, the majority of the benefit was observed with respect to reduction in energy demand for aeration as opposed to energy generation in anaerobic digestion. For instance, the reduction in energy demand generated by the enhanced removal of the DAF plant accounted for 70% of the total benefits. Energy generation from flotation sludge was estimated at 7,592 MWh.year$^{-1}$, with FOG providing an additional 651 MWh.year$^{-1}$, corresponding to an increase of 19% to the baseline scenario (i.e. 6,137 MWh.year$^{-1}$ generated with conventional primary treatment). Furthermore, the improved management of FOG contributed to 42% of the total benefits for the DAF and 52% for the CE-PST.

The significantly higher energy benefit of the DAF plant is reduced by the increased energy demand for operation compared to the CE-PST at 2,555 and 38 MWh.year$^{-1}$ respectively. The overall net energy balance is therefore 2,269 MWh.year$^{-1}$ for the DAF compared to 3,445 MWh.year$^{-1}$ for the CE-PST.

**Table 6-6** Energy required for aeration and generated through anaerobic digestion. The base case considers a conventional PST. Positive values indicate savings whilst negative ones represent demands.

| **Parameter (in MWh.year$^{-1}$)** | **PST** | **CE-PST** | **DAF** |
|---|---|---|---|
| Energy demand from primary treatment | -23 | -38 | -2,555 |
| Total energy demand for $BOD_5$ | -7,763 | -5,220 | -4,417 |
| Energy demand for FOG | -2,526 | -1,465 | -665 |
| Total energy from anaerobic digestion | +6,137 | +7,054 | +7,592 |
| Energy generation from anaerobic digestion of FOG | | +375 | +651 |
| Net energy | -1,649 | +1,796 | +620 |
| Net change from base case | | +3,445 | +2,269 |

The net OpEx cost when using enhanced primary treatment revealed a net saving of £0.13M.year$^{-1}$ for DAF prior to inclusion of capital costs. By contrast, CE-PST was associated with net OpEx of £0.06M.year$^{-1}$. It is important to note that these results were based on CE-PSTs motivated by phosphorous removal with dosing rates around 30 mg.L$^{-1}$. If switching to chemical enhancement was purely motivated by a need to deliver load reduction across the primary process to cope with population growth (i.e. increased flow or solid demands), lower quantities of coagulant, estimated at 10 mg.L$^{-1}$ from TWUL's asset standards, will be required providing a net saving of £0.20M.year$^{-1}$. The CapEx for the DAF plant including dosing plant was estimated at £6.20M for this hypothetical STW serving 500,000 PE which equated to an ACC of £0.26M.year$^{-1}$. Accordingly, the savings made did not offset the cost of the plant indicating that there is not a convincing case to switch from sedimentation to DAF purely on an economic basis associated with solids and FOG. In comparison, retrofitting chemical dosing to conventional sedimentation processes was found an economically favourable option due to significantly lower capital investment required. However, if the works requires upgrade to flow or load increases that can no longer be resiliently met by the existing sedimentation processes then DAF appears a sensible option. Further, the current economic analysis is based on a low lipid content sludge due to low influent concentrations. Should the FOG levels increase or further optimisation work improve overall solid removal then the case for change can be made of a purely economic basis. For instance, if effluent $BOD_5$, SS and HEM concentrations respectively as low as 51, 74 and 10 mg.L$^{-1}$, as obtained with DAF2-FlocB (Table 6-5), were to be achieved, the current analysis would be adjusted to slightly higher than cost neutrality (Figure 6-6). In turn, retrofitting conventional sedimentation processes with DAF would be justified from an economic point of view providing significant load reduction to secondary treatment.

The cost associated to installing new PSTs equated to an ACC of £0.99M.year$^{-1}$ whereas retrofitting chemical dosing equated to an ACC of £0.04M.year$^{-1}$ indicating that it provides a feasible economic basis for upgrading primary treatments. Disadvantages of sedimentation tanks include low SORs and hence large footprints and limited ability to control sludge dry solids. In contrast, DAF plants, operated at significantly higher SORs, can be turned up/down by altering the mass of bubbles introduced and can generate thicker sludge with levels appropriate for anaerobic digestion negating the need for thickening processes. These additional features have not been accounted for in the current case but can become critical depending on the specific circumstance of the site in question. In relation to the context of FOG management, upgrading all STWs is not realistic and will require understanding where the benefits would be the highest. Managing FOG at STWs further implies on-going OpEx on sewerage systems. Therefore, more research is required in the field to capture the potential benefits of FOG-control at source to lead to more clarity as to the overall FOG management strategy.

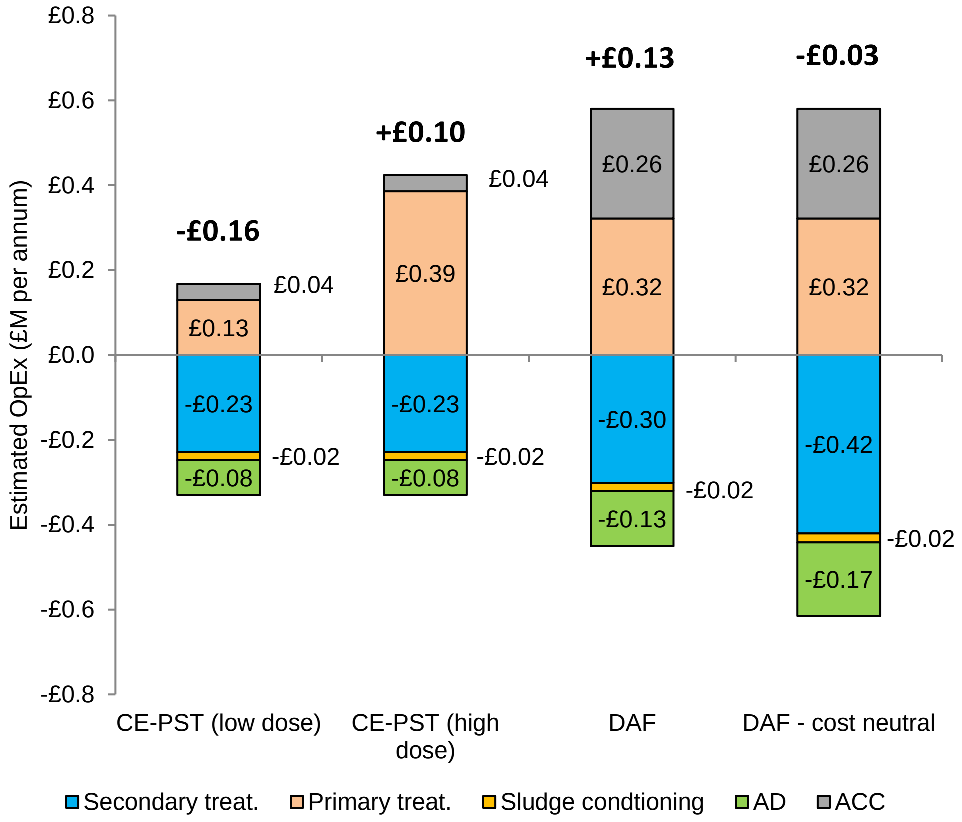

**Figure 6-6** Estimated net OpEx for CE-PST and DAF from baseline scenario. Positive values indicate costs incurred, whilst negative ones represent savings. Net OpEx are represented in bold for each scenario.

## 6.4 Conclusions

Based on a monthly sampling conducted over a five-year period, FOG as HEM was found occurring in urban wastewater at concentrations averaging 57±11 mg.L$^{-1}$. FOG removal efficiencies were reported on average at 50% and 71% respectively from conventional and CE primary sedimentation. By contrast, DAF achieved removal rates of 74% with effluent HEM concentrations of 10±4 mg.L$^{-1}$.

Whilst DAF was evaluated providing significant benefits reducing aeration demand from biological treatment and increasing energy generation through anaerobic digestion, the case to switch from sedimentation to DAF purely on an economic basis was not supported. Yet, DAF, with lower capital investment and footprint required, appeared as a sensible option over sedimentation if the works require upgrading. In relation to FOG management, upgrading all STWs is not realistic. Managing FOG at STWs would imply on-going OpEx in sewerage systems, therefore enhancing primary treatments for FOG removal would require a case-by-case approach to identify where benefits would be the highest.

## 6.5 Acknowledgments

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# 7 Towards the sustainable management of fats, oils and greases (FOG)

The overall aim of this research was to provide guidance for water utilities on the management of fats, oils and greases (FOG). The case for optimising current management strategies is twofold: (1) protect wastewater infrastructure as well as the surrounding environment from complications caused by uncontrolled emissions and (2) maximise energy recovery from FOG to offset operational costs.

A number of key options require consideration when examining sustainable management strategies for FOG:

- Where to harvest the FOG from: at source, in the networks or at sewage treatment works (STWs)?
- Which sources (i.e. domestic and food outlets) to include and at what coverage?
- How is the value generated from collection and processing of FOG?

Harvesting FOG from sewer systems or STWs has been considered but is viewed as an undesirable option due to reduction in feedstock quality and technical challenges in extracting the wastes (Chapters 2 and 6). Accordingly, the focus of the current investigation is collection at source and processing to generate an energy product in the form of biodiesel or biomethane. To assess this, the overall balances of FOG have been estimated using Thames Water Utilities Ltd. (TWUL) catchment as a study area, and the energy and economic balances modelled for the different options were compared. The objectives of the following section are to (1) identify barriers to the implementation of a FOG control programme and (2) establish potential economically viable options for FOG management. Industrial sources of FOG were not included in this assessment as they were assumed controlled by water authorities under trade effluent consents.

## 7.1 Material and methods

### 7.1.1 Capital investment

Capital expenditure (CapEx), which is the amount of capital required to finance the project, would be incurred from collection systems (i.e. collection banks for domestic FOG and grease separators for foot outlets) and retrofitting existing plants for the treatment of FOG (i.e. biodiesel production and anaerobic co-digestion). Capital costs were annualised over their lifetime at an interest rate of 2.8% (Ofwat, 2017). Collection systems and plant infrastructures were assumed with respective lifetimes of 5 and 20 years. The annual cost of capital (ACC) was then calculated as follows:

$$ACC = CapEx \times \frac{i\,(1+i)^n}{i\,(1+i)^n - 1} \quad \textbf{(7-1)}$$

Where $n$ is the project estimated lifetime in years, and $i$ is the interest rate for capital loan.

The base year of this economic evaluation was 2018. Some cost data was collected in EUR or USD and were converted at the rate EUR:GBP of 0.61 (2001), USD:GBP of 0.55 (2004) and USD:GBP of 0.78 (2018). All the equipment cost values collected before 2018 were adjusted using the Chemical Engineering Plant Cost Index, and calculated as follows (Ludwig, 1999):

$$EC_{2018} = EC_1 \times \left(\frac{I_{2018}}{I_1}\right) \quad \textbf{(7-2)}$$

Where $I_{2018}$ is the index value for 2018 (at 603.1), $EC_{2018}$ is the equipment estimated cost for 2018, $EC_1$ is the equipment cost referenced. These index values were $I_{2001}$ = 394.3, $I_{2004}$ = 444.2 and $I_{2009}$ = 521.9.

### 7.1.2 FOG estimation

The energy and economic balances were estimated for this case study based on TWUL catchment. Data with a geographic component was manipulated using ArcGIS. Domestic properties were extracted from AddressBase® Premium (Ordnance Survey, 2017). Domestic oil use was estimated assuming 27% of households owned a deep fat fryer (Sabanoglu, 2018) and that 50% used it regularly. Therefore, these households were projected to produce more FOG at rates of 9.0 kg.year$^{-1}$ (McKinney, 2012) whilst the remaining 86% were assumed producing 2.3 kg of FOG per year (Chapter 3). Further, it was considered that these volumes corresponded to 76% of the total volumes of FOG generated reflecting previous studies (Quested et al., 2013).

**Table 7-1** Estimated FOG production rates from FSEs in g per capita.

| FHRS classification | Correction factor[a] | This study (Chapter 5) | Grenz and Patel (2007) | Doherty (2009) | Merged datasets |
|---|---|---|---|---|---|
| **Catchment population** | | 15,449 | 873 | 4,459,300 | |
| **Catchment visiting population** | | 2,070,205 | 5,404,494 | 5,687,000 | |
| **Hospital, childcare and caring premise** | 55% | 0.05 | nd | 0.21 | 0.13 |
| **Hotel, bed and breakfast and guest house** | 83% | 0.19 | 0.27 | 0.07 | 0.18 |
| **Other catering premises** | 50% | nd | nd | 0.02 | 0.02 |
| **Pub, bar and nightclub** | 58% | 0.09 | 0.10 | 0.07 | 0.09 |
| **Restaurant, café and canteen** | 88% | 0.09 | 0.16 | 0.07 | 0.12 |
| **Supermarket and hypermarket** | 50% | nd | nd | 0.05 | 0.05 |
| **School, college and university** | 100% | 0.05 | nd | 1.29 | 0.47 |
| **Takeaway and sandwich shop** | 50% | nd | 0.11 | 0.36 | 0.19 |

[a] A correction factor was applied based on the number of premises likely to produce FOG as by-products from their activities observed during the survey; nd: no data.

Data associated with food service establishments (FSEs) was extracted from the Food Hygiene Rating Scheme (FHRS) (Food Standards Agency, 2019). Data available from literature and produced during this research was adapted to reflect FOG production rates per capita (Table 7-1). The contribution from visiting population was included in this assessment and data was obtained from Kantar TNS (2018b) for each local authority within the catchment. During the survey of FSEs, it was highlighted that not all businesses registered under the FHRS were producing FOG. As such, a correction factor was introduced for each business type reflecting the percentage of FSEs likely to generate FOG as determined from the survey (Chapters 2 and 4).

### 7.1.3 FOG capture

#### 7.1.3.1 From food outlets

Currently, only 15% of FSEs physically separate FOG from their effluents (Chapter 4). Thus, the model assumed that FOG management systems, either in the form of grease removal units (GRUs) or passive grease separators, will be required for 85% of the remaining FSEs. The maximum efficiency achievable for these systems was 82% (Gallimore et al., 2011). CapEx and operational expenditure (OpEx) on the installation and maintenance of grease separators were adapted from Aqua Cure Ltd. (2016) (Table 7-2). The cost of collection of FOG wastes was estimated at £15 per m$^3$ based on figures produced for kerbside food waste collection (Hogg, 2001).

**Table 7-2** Costs comparison between GRU and passive grease separator per unit adapted from Aqua Cure Ltd. (2016).

| Unit type | CapEx (£) | Maintenance and power (£.year$^{-1}$) | Pump out (£.year$^{-1}$) |
|---|---|---|---|
| **Passive grease separator** | £550 | N/A | £2,975 |
| **GRU** | £2,784 | £667 | £700 |

Compliance and enforcement will be a necessary part to the FOG control programme. Associated staff employment costs were calculated at £11.2 per hour from the UK weekly rate at £361.4 for service occupations (Office for National Statistics, 2017b) to which an increment of 123.7% was added to cover employers' national insurance, pension contribution, training and administration charges (Yang et al., 2018). It was assumed that a number of four FSEs could be visited per day (McIlvaine and Flynn, 2011). Compliance after the first visit was 51% (Thames Water internal data). It was further assumed that all FSEs will become compliant after three visits, and that there will be one inspection per year for each premise.

#### 7.1.3.2 From domestic properties

The economics for the capture of FOG from domestic properties were investigated via (1) collection banks installed at *bring sites* or (2) kerbside food waste collection organised by local authorities. The number of drop-off locations operated per employee was assumed to be 25 based on experience from Dallas Water Utilities (Helms and Dulac, 2016). Capital costs were calculated for collection banks assuming a volume of 1 $m^3$. The number of collection banks required was estimated based on the total number of households divided by the number of households that could fil one collection banks based on estimated production rates. This number was then adjusted based on expected participation rates. Investment costs of *bring sites* were adapted from Eunomia Research & Consulting (2001) and evaluated at £855 per site. OpEx from staff employment was calculated using the same assumptions presented in section 7.1.3.1.

### 7.1.4 Water utility

Net operational costs for the water utility were determined as the sum of OpEx from (1) sewerage systems, (2) pumping stations and (3) STWs subtracted from the (4) electricity generation from degradation of FOG removed with primary sludge through anaerobic digestion.

#### 7.1.4.1 Sewerage systems

Within the water utility, sewer incidents are recorded by field operators with a geographical component. This data was extracted from 01/01/2012 to 13/09/2018 for blockages, cleaning (reactive, planned and emergency), flooding (internal and external) and other activities possibly caused by FOG discharges (e.g. CCTV investigation, sewer failure). Internal data was used to estimate the average OpEx for each activity. Experience within water utilities suggested that FOG deposits form between 50 and 200 metres from FSEs (Keener et al., 2008). As such, sewer incidents recorded within 150 metres of a FSE were attributed to FSEs whilst the remaining events being within 150 metres of a household were considered as "domestic" events. Field crews attending blockage incidents generally record the cause of blockage. The analysis of this data demonstrated that the most common recorded causes of blockage were: paper and rag (50%), FOG (32%), debris (5%) and defects (5%). Therefore, it was assumed that around 32% of all sewer incidents were caused by FOG.

OpEx incurred from sewage pumping stations were also obtained from TWUL. Three OpEx categories were assumed impacted by FOG (at 32%): maintenance, wet well cleaning and tanker hiring. Due to commercial sensitivity, details on costs data from sewers are restricted to overall values in this study.

#### 7.1.4.2 Sewage treatment works

Volumes of FOG reaching sewage treatment works were estimated using FOG concentrations measured at 57 $mg.L^{-1}$ in crude sewage (Chapter 6) and flow data from TWUL STWs. To only estimate amounts of FOG from vegetable oils and animal fats, the contribution of lipids from faeces and soaps, respectively of 4.1 $g.capita^{-1}.day^{-1}$ (Rose et al., 2015) and 1.5 $g.capita^{-1}.day^{-1}$ (Ram et al., 2018), were subtracted from total volumes calculated.

FOG was assumed increasing the biochemical oxygen demand ($BOD_5$) load to STWs further representing for around 23% of this organic load (Chapter 6). Average FOG removal rates from both conventional and chemically-assisted primary sedimentation produced of 59% were used (Chapter 6). This percentage of FOG removed alongside primary sludge was estimated degradable at 74% of through anaerobic digestion (Chapter 5). Biomethane yields were then assumed at 795 $m^3$ per ton $VS_{destroyed}$ (Chapter 2). The remaining FOG fraction found in settled sewage is assumed degraded during the secondary treatment. It was calculated from TWUL flow data that 85% of the total volume of wastewater is treated through aeration lanes. The energy requirement from aeration lanes was estimated at 1.5 kWh per kg $BOD_5$.

### 7.1.5 Energy recovery

Due to their energy-rich nature, FOG can be turned from waste to resource. The economics for their use as a substrate for energy recovery are based on the characteristics summarised in Table 7-3.

**Table 7-3** Key characteristics of FOG wastes used to determine their value for energy recovery (Chapters 3 and 5).

| Parameter | Domestic FOG | FSE FOG |
|---|---|---|
| **Dry solids (%wt.)** | 95.8% | 89.2% |
| **Lipids (% dry solids)** | 94.3% | 88.0% |
| **Lower heating value ($MJ.kg^{-1}$)** | 37.0 | 31.4 |
| **Biomethane yields ($m^3$ $CH_4$.ton $FOG_{destroyed}^{-1}$)** | 942 | 1,073 |
| **FOG destruction (%)** | 87% | 80% |
| **Biodiesel conversion yield (%)** | 26% | 75%[1] |

[1] Canakci and Van Gerpen (2001a)

Four recovery routes were considered for valorisation of the collected FOG using existing facilities: (1) disposal to landfill, (2) incineration, (3) anaerobic co-digestion and (4) biodiesel. Assumptions used to estimate the potential revenue streams and OpEx for each route are summarised in Table 7-4. Revenue from biodiesel was calculated as follows:

$$V_{biodiesel} = V_{diesel} - (VAT \times V_{diesel} + Fuel\ duty\ rate) \qquad \textbf{(7-3)}$$

Where $V_{biodiesel}$ is the calculated market value of biodiesel, $V_{diesel}$ is the market value of diesel (£1.311 per L), value added tax (VAT) is 20% and fuel duty rate is £0.5795 per L (HM Revenue & Customs, 2014).

**Table 7-4** Assumptions used for the energy recovery from FOG.

| Parameter | Value | Source |
|---|---|---|
| **1 – Landfill** | | |
| Landfill tax | £88.9 per ton | HM Revenue & Customs (2018) |
| Methane yields in landfill | 43 $m^3$ per ton of municipal solid waste | Themelis and Ulloa (2007) |
| **2 – Incineration** | | |
| Energy conversion efficiency of incineration | 20% | CIWEM (n.d.) |
| Gross operating costs | £70 per ton | McKendry (2008) |
| **3 – Anaerobic co-digestion** | | |
| Calorific value of methane | 36 $MJ.m^{-3}$ | |
| Electrical efficiency of CHP engines | 30% | Goss et al. (2017) |
| ACC for retrofitting facility | £78.6 per ton | adapted from Laborde (2009) |
| Operational costs | £15.9 per ton | adapted from Laborde (2009) |
| **4 – Biodiesel conversion** | | |
| Value of biodiesel | 0.47 £ per L | Calculated from (7-3 |
| Equivalent of 1 $m^3$ of biodiesel | 0.78 tonne of oil equivalent | Eurostat (2018) |
| Energetic value of biodiesel | 11.6 MWh per ton equivalent | Eurostat (2018) |
| ACC for retrofitting facility | £71.3 per ton | adapted from Laborde (2009) |
| Operational costs | £19.9 per ton | adapted from Laborde (2009) and Canakci and Van Gerpen (2001b) |

### 7.1.6 Scenario modelling

A baseline scenario was developed to evaluate the current costs incurred by FOG for the water utility in sewer networks and at the STWs (i.e. increased aeration demand) and the benefits of energy generation from FOG through anaerobic digestion (Figure 7-1).

In terms of FOG from commercial sources, three scenarios were developed: (1) targeting all FSEs within TWUL catchment, (2) FSEs within high- and medium-risk areas and (3) FSEs within high-risk areas. Sewer drainage area catchments (SDACs) were categorised into low-, medium- and high-risk areas derived from the normalised number of sewer blockages per kilometres of sewers (Table 7-5).

**Table 7-5** Risk-ranking of SDACs. SDACs were categorised based on the number of blockages recorded in the area over its sewer length.

| **Parameter** | **High-risk** | **Medium-risk** | **Low-risk** |
|---|---|---|---|
| **Blockage per km of sewers** | > 2 | Between 1 and 2 | < 1 |
| **Number of SDACs** | 1,107 | 1,047 | 982 |
| **Number of FSEs** | 21,065 | 34,674 | 13,227 |

Two main scenarios were tested for the collection of domestic FOG: (1) through collection banks or (2) through kerbside collection with food waste (Chapter 3). A summary of the inputs and outputs from these scenarios is provided in Figure 7-2.

**Figure 7-1** Inputs used for the baseline scenario (i.e. current state). Positive signs indicate savings while negative ones represent losses. FHRS: Food Hygiene Rating Scheme, AD: anaerobic digestion, Acc.: accumulation.

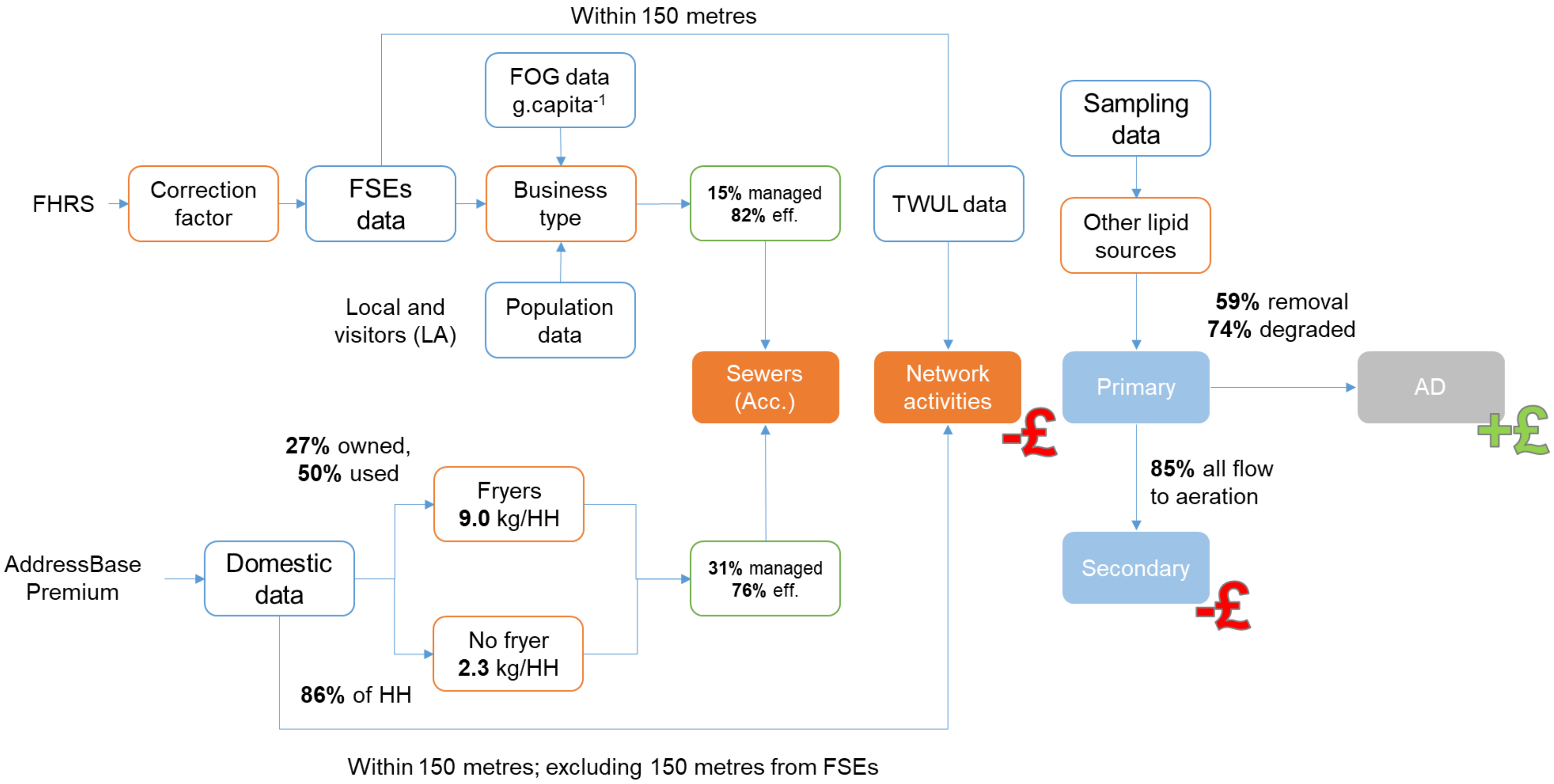
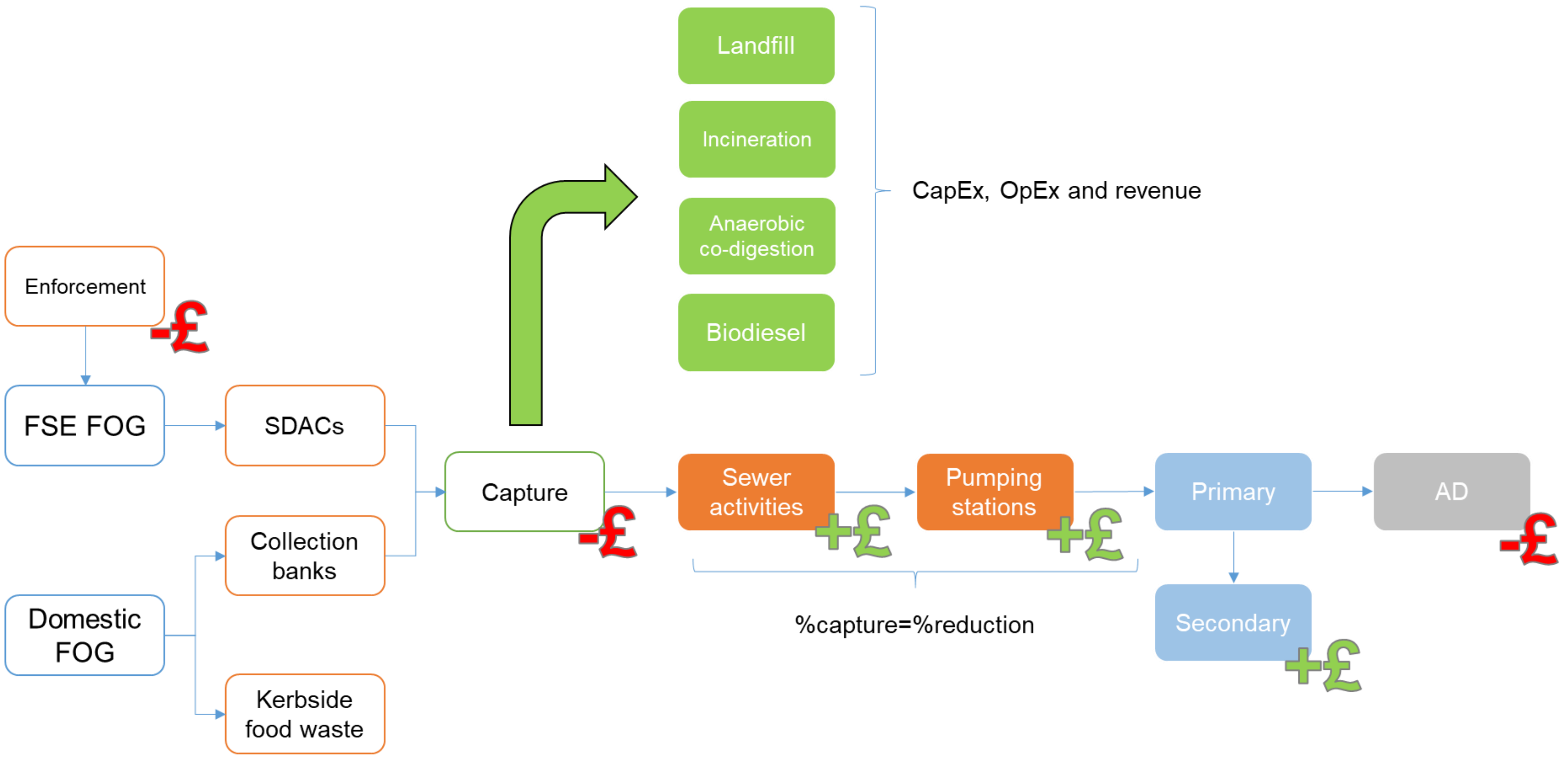


**Figure 7-2** Summary of the inputs and outputs of the scenario modelling. Positive signs indicate savings while negative ones represent losses.



## 7.2 Results and discussion

### 7.2.1 Current mass, energy and economic balances

To be able to determine future directions in terms of FOG management, it is necessary to understand the scale of the problem in the first place. Based on data gathered during this thesis, around 60,470 tonnes of FOG were estimated produced yearly from FSEs within the study catchment. Currently, only 15% of FSEs manage their FOG discharges in an efficient manner (Chapter 4), potentially contributing to 51,400 tonnes of FOG from FSEs discharged into the sewers yearly.

In comparison, volumes of FOG generated from domestic sources were initially evaluated at 14,920 tonnes per year solely based on data presented in Chapter 3. This estimate was re-evaluated to account for population using larger volumes of cooking oils (McKinney, 2012; Sabanoglu, 2018). Total volumes generated from the 6.3M households within the catchment were re-estimated at 26,085 tonnes per year. It was further assumed that 14% of the total volume is currently diverted from the sewers (i.e. share of households owning a frequently used fryer) further estimating discharges of FOG from domestic sources at 23,395 tonnes per year (Figure 7-3).

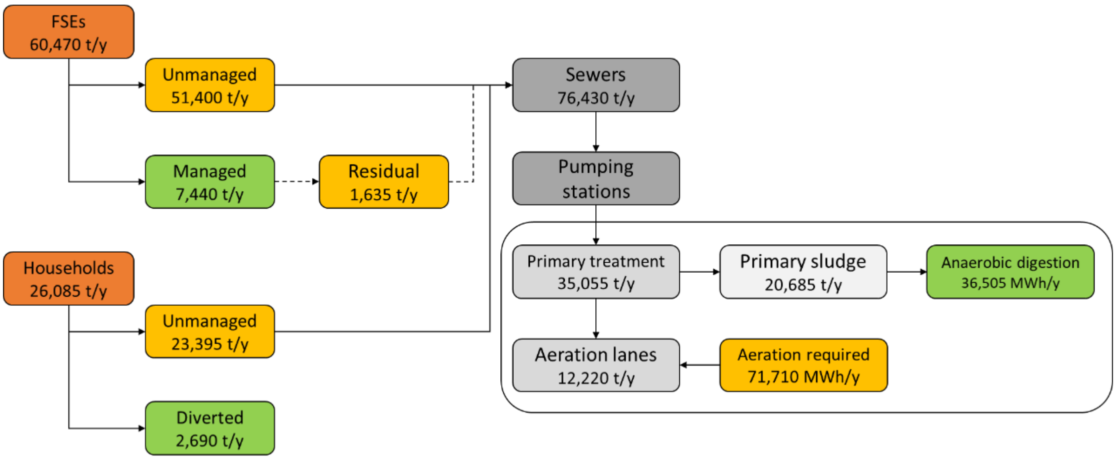


**Figure 7-3** Current mass and energy balances of FOG across the catchment area

The current FOG load into the sewers was therefore calculated at 76,430 tonnes.year$^{-1}$ which was estimated to generate 15,745 sewer blockages at an operating costs of £7.8M.year$^{-1}$. In pumping stations, FOG was evaluated contributing to a further £2.0M.year$^{-1}$ to OpEx. Around 35,055 tonnes of FOG were estimated reaching TWUL's STWs corresponding to a 51% difference from source-based estimates. Based on previous trials, 59% of the FOG would partition into the sludge line during primary treatment (Chapter 6), equivalent to 20,685 tonnes.year$^{-1}$. At the assumed conversion rates this will generate 36,505 MWh.year$^{-1}$ through anaerobic digestion equivalent a revenue of £3.5M.year$^{-1}$ from electricity generation. However, the remaining FOG fraction in the liquid line will exert an additional aeration demand of 71,710 MWh.year$^{-1}$ contributing to OpEx of £6.8M.year$^{-1}$. Consequently, there is a net deficit in electricity of 32,205 MWh.year$^{-1}$ due to current management practices at the STWs. Overall, the current position related to FOG management is estimated to result in a total operating costs of £16.6M.year$^{-1}$ with an income generation of £3.5M.year$^{-1}$ resulting in a net deficit of £13.1M.year$^{-1}$

### 7.2.2 Alternative mass, energy and economic balances

#### 7.2.2.1 FSEs options

The majority of the FOG entering the network was from FSEs which did not have separators and represented 69% of the current estimated total operating costs associated with FOG. The enforcement cost to make approximatively 42,080 FSEs compliant was estimated at £1.9M.year$^{-1}$. Diversion of these wastes from sewerage systems was estimated saving £9.0M.year$^{-1}$ in operating costs associated with the network, pumping stations and STWs. However, the reduced FOG in the sludge decreased energy production of 21,185 MWh.year$^{-1}$ at a loss of revenue of £2.0M.year$^{-1}$ resulting in a total net benefit of £5.1M per year for the all SDACs case (Table 7-6).

Furthermore, there is an additional source of revenue for converting the collected FOG into energy through the four proposed routes. Assuming a 82% efficiency of grease separator (Gallimore et al., 2011), 42,145 tonnes of FOG per annum were estimated collectable from FSEs. The lowest energy potentials were calculated for landfill disposal at 5,435 MWh.year$^{-1}$ (Figure 7-4a). Utilisation of the high energy content of the collected FOG was better realised through incineration or anaerobic co-digestion which produced 59,025 MWh.year$^{-1}$ and 79,690 MWh.year$^{-1}$ respectively for the all SDAC case. The highest energy generation value was estimated to come through biodiesel production which generated 202,850 MWh.year$^{-1}$. This equates to 343% and 254% of the incineration and anaerobic co-digestion routes respectively. The corresponding net revenue (subtracting OpEx associated with collection and processing) generation from the additional value generation was estimated at -£3.8M.year$^{-1}$, £2.0M.year$^{-1}$, £6.1M.year$^{-1}$ and £9.0M.year$^{-1}$ for the landfill, incineration, anaerobic co-digestion and biodiesel scenarios respectively (Table 7-6). Combining with the other benefits the total net revenue (excluding purchase of the separators) is estimated to be £1.3M.year$^{-1}$, £7.1M.year$^{-1}$, £11.2M.year$^{-1}$ and 14.1M.year$^{-1}$ for the respective cases.

Capital investment for passive grease separators was estimated at £23.1M for the 42,080 non-compliant FSEs. Payback periods based on CapEx were evaluated at 4.5 years, 17.3 years, 3.2 years, 2.1 years and 1.6 years respectively for the enforcement only, landfill, incineration, anaerobic co-digestion and biodiesel scenarios. Nevertheless, this would require the pre-treatment of FOG to remove the large volumes of water associated with these grease trap wastes (GTW). By contrast, GRUs were demonstrated to produce a concentrated type of waste thus making collection and utilisation more efficient (Chapter 5) but were also associated with large CapEx further estimated at £117.1M. Thus, payback periods were found much higher at 22.6 years, 87.8 years, 16.2 years, 10.4 years and 8.3 years for the respective scenarios. From a FSE point of view, GRUs offer lower operational costs (Table 7-2) but are also associated with increased daily maintenance from kitchen staff in disposing of the collected FOG.

In addition to the economic analysis there are a number of additional factors that needs consideration associated to different routes. Incineration currently has a negative perception due to due to the emissions of substances polluting the environment and the requirement to dispose of heavily contaminated ashes

(National Research Council, 2000). Using FOG as a co-substrate in anaerobic digesters is an attractive option for UK water utilities which possess the largest asset base in the country. Government incentives, under the Renewable Obligation certificates, would provide an additional revenue stream however this scheme is closed to new energy generator and FOG would have to be diverted to existing renewable energy generation plants to benefit from it. Despite these potential benefits, currently in the UK anaerobic co-digestion with sewage sludge is limited due to different governing regimes. The digestate produced from anaerobic digestion of sewage sludge is covered under the Sludge Regulations and the Safe Sludge Matrix which governs its application to land. On the other hand, source-segregated FOG would be covered by the Anaerobic Digestate quality protocol and the Publicly Available Specification 110. The digestate produced from co-digestion would no longer fall into the sludge regulations and would be considered as a waste under the Waste Framework Directive possibly requiring expensive environmental permitting for disposal to land.

Targeting all the FSEs within the catchment would provide the highest benefits not only in terms of relieving wastewater infrastructures but also for energy generation. However, whilst in its early days, enforcement and compliance are conducted by water utilities requiring significant efforts to target all premises. As such, prioritisation could be considered to keep costs reasonable. SDACs were categorised according to their normalised blockage rate (Table 7-5). A FOG strategy solely based on enforcement and compliance was estimated to produce savings between £3.6-5.2M.year$^{-1}$ depending on SDAC targeting (Figure 7-4b). Favouring biodiesel production or anaerobic co-digestion from FOG collected from medium- and high-risk SDACs was offsetting current OpEx spent on FOG from food outlets. It is important to mention that for this analysis, using FOG as a biodiesel feedstock assumed processing at an existing UK plant. However, if a new facility had to be built, this will be associated with large CapEx evaluated around £60.9M (adapted from Laborde, 2009). In turn, this would make the biodiesel route significantly less economically attractive over anaerobic co-digestion. Accordingly, complete subsidy of the separators does not appear economically appropriate and so partial subsidy to enhance participation appears more sensible.

**Table 7-6** OpEx and revenues from energy generation from FSE FOG based on SDAC targeting.

| **Parameter** | **High-risk SDACs** | **Medium and high-risk SDACs** | **All SDACs** |
|---|---|---|---|
| **Number of non-compliant FSEs** | 12,623 | 33,539 | 42,080 |
| **GRUs CapEx (£M)** | £35.1 | £93.4 | £117.1 |
| **Grease separators CapEx (£M)** | £6.9 | £18.4 | £23.1 |
| **FOG collectable (tonnes.year$^{-1}$)** | 12,580 | 32,010 | 41,245 |
| **1 – OpEx (£M.year$^{-1}$)** | | | |
| Enforcement | £0.6 | £1.5 | £1.9 |
| FOG collection | £0.2 | £0.5 | £0.6 |
| Landfill OpEx | £1.1 | £2.8 | £3.7 |
| Incineration OpEx | £0.9 | £2.2 | £3.0 |
| Anaerobic co-digestion OpEx | £0.3 | £0.7 | £0.9 |
| Biodiesel OpEx | £0.3 | £0.7 | £0.9 |
| **2 – Revenue stream (£M.year$^{-1}$)** | | | |
| Reduction in AD biogas from sludge line | -£0.6 | -£1.6 | -£2.0 |
| Reduction in networks costs | £3.3 | £3.2 | £4.0 |
| Reduction in pumping station costs | £0.3 | £0.9 | £1.1 |
| Reduction is STW costs | £1.2 | £3.0 | £3.9 |
| Landfill | £0.2 | £0.4 | £0.5 |
| Incineration | £1.7 | £4.3 | £5.6 |
| Anaerobic co-digestion | £2.3 | £5.7 | £7.6 |
| Biodiesel | £3.1 | £8.0 | £10.5 |

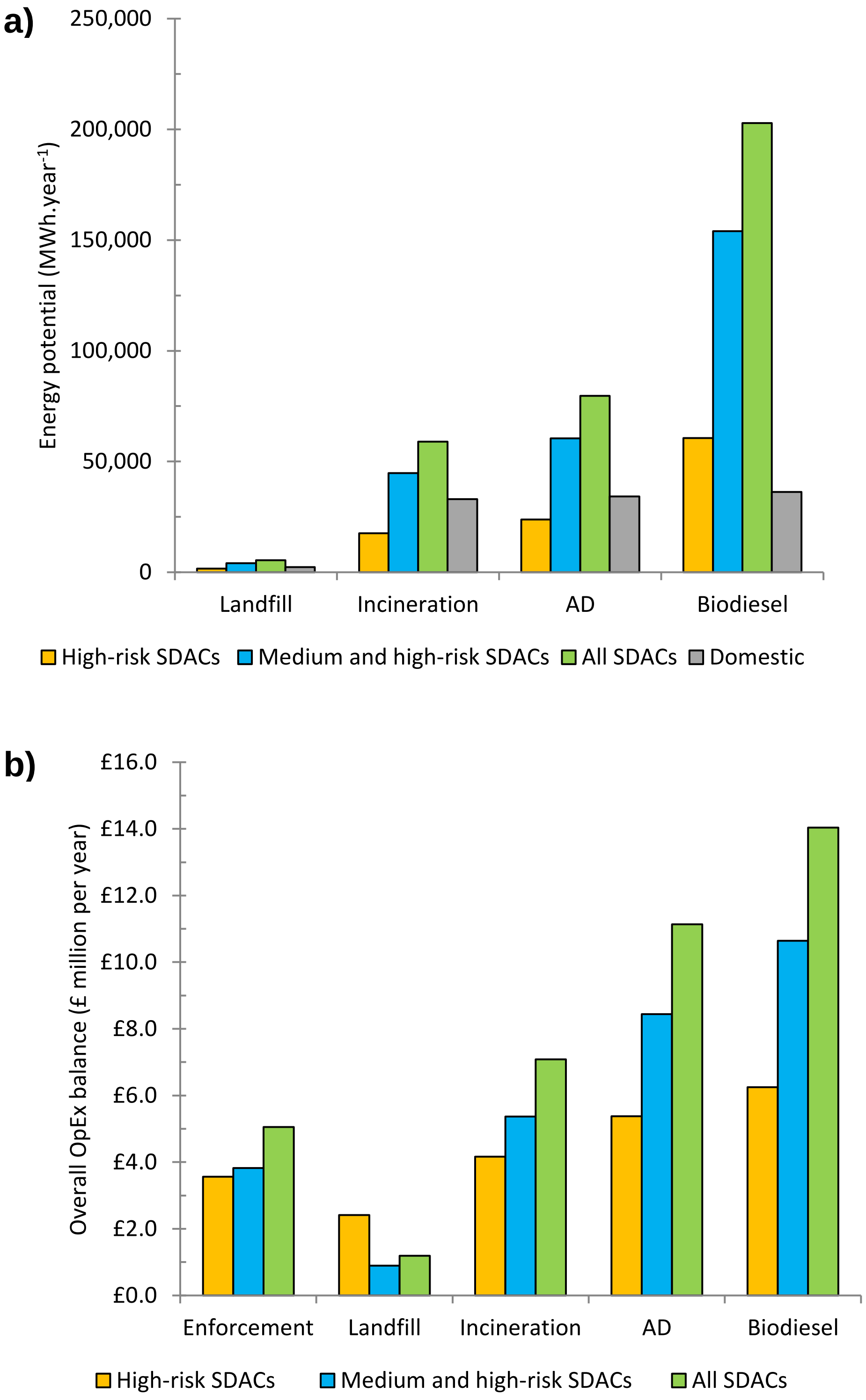


**Figure 7-4** Total energy potential (a) and associated OpEx balance (b) from FSE FOG from the baseline scenario.

#### 7.2.2.2 Households options

Across the 6.3M households within the catchment, 26,085 tonnes of FOG are produced per year. Currently, it was estimated that FOG discharges from households represented an annual spend of £5.5M.year$^{-1}$ for the water utility with a net OpEx of £4.4M.year$^{-1}$.

The first scenario explored collection using drop-off locations. Energy generation from incineration, anaerobic co-digestion and biodiesel was estimated between 33,060-36,195 MWh.year$^{-1}$. The corresponding net revenue from the additional value generation combined with other benefits was calculated at £2.8M.year$^{-1}$, £3.9M.year$^{-1}$ and £2.5M.year$^{-1}$ respectively for incineration, anaerobic co-digestion and biodiesel. Despite high energy potentials, the total capture of FOG is unrealistic. In light of households' participation rates observed in Chapter 3, it is further evaluated that only 21% of these benefits are achievable through collection banks (Chapter 3), corresponding to net revenue of £0.6M.year$^{-1}$, £0.8M.year$^{-1}$ and £0.5M.year$^{-1}$ for the respective cases.

Penetration rates were posited maximised relying on kerbside food waste collection schemes organised by local councils (Foden et al., 2017). With collection rates assumed at 30% (WRAP, 2016c), net savings were estimated at £1.5M.year$^{-1}$ for the water utility. Energy generation from co-digestion of FOG and food waste including other benefits was estimated at £2.3M.year$^{-1}$. In this case, any benefits from energy generation only profit the third party waste company treating food waste. Ultimately, to maximise penetration rates a significant effort in terms of customer engagement will be required compared to FSEs, further contributing to OpEx. However, this educational message could fit in a wider context to raise awareness on other issues (e.g. unflushable items).

**Table 7-7** OpEx and revenues from domestic FOG energy recovery based on participation rates.

| Parameter | All councils | | Only kerbside scheme | |
|---|---|---|---|---|
| **FOG collectable (tonnes.year$^{-1}$)** | 17,870 | | 15,540 | |
| **Participation rates** | 21% | 100% | 30% | 100% |
| **1 – OpEx (£M.year$^{-1}$)** | | | | |
| FOG collection[1] | £0.5 | £2.4 | £0.07 | £0.2 |
| Landfill OpEx | £0.3 | £1.6 | | |
| Incineration OpEx | £0.3 | £1.3 | | |
| Anaerobic co-digestion OpEx | £0.1 | £0.4 | £0.1 | £0.4 |
| Biodiesel OpEx | £0.05 | £0.2 | | |
| **2 – Revenue stream (£M.year$^{-1}$)** | | | | |
| Reduction in AD biogas from sludge line | -£0.2 | -£0.8 | -£0.2 | -£0.9 |
| Reduction in networks costs | £0.4 | £2.1 | £0.9 | £2.1 |
| Reduction in pumping station costs | £0.1 | £0.5 | £0.2 | £0.5 |
| Reduction is STW costs | £0.4 | £1.7 | £0.8 | £1.7 |
| Landfill | £0.05 | £0.2 | | |
| Incineration | £0.6 | £3.1 | | |
| Anaerobic co-digestion | £0.7 | £3.4 | £1.0 | £3.4 |
| Biodiesel | £0.4 | £1.8 | | |

[1] including ACC for collection banks; [2] including ACC from retrofitting existing plant

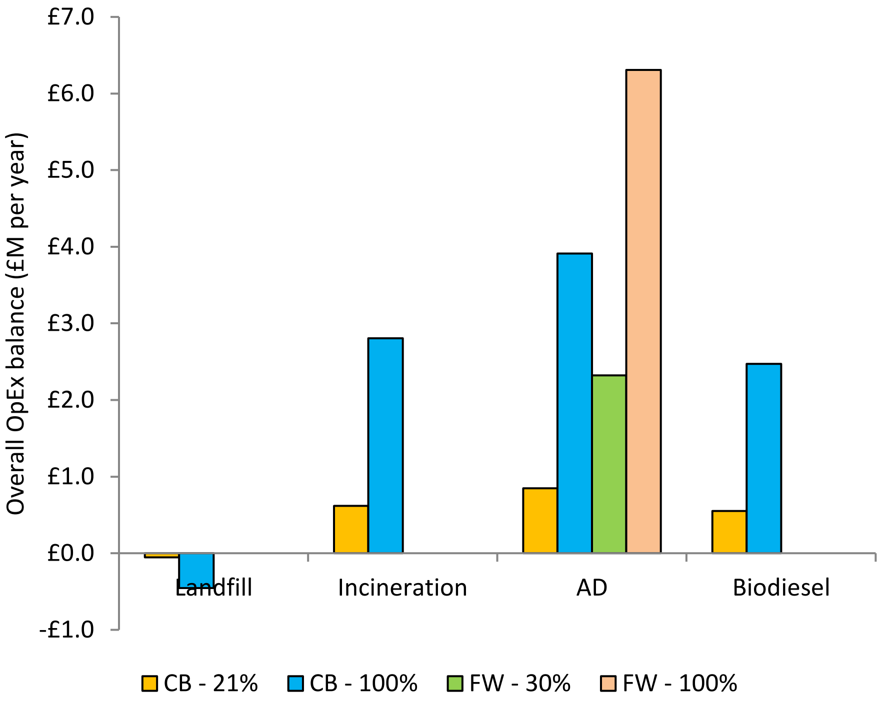


**Figure 7-5** Overall OpEx balance from domestic FOG accounting for energy generation from baseline scenario based on the collection mode: collection banks (CB) or kerbside food waste (FW) schemes.

## 7.3 Conclusions

A prediction of the economic viability of developing a FOG programme focusing on currently unmanaged discharges of FOG from FSEs and promoting FOG kerbside collection with food waste can be provided (Figure 7-6). Current energy generation from FOG through anaerobic digestion was found decreased from 36,505 MWh.year$^{-1}$ down to 12,360 MWh.year$^{-1}$ corresponding to a revenue loss of £2.3M.year$^{-1}$ for the water utility. However, OpEx savings of £10.7M.year$^{-1}$ were achieved in networks, pumping stations and STWs. Thus, net OpEx including enforcement were estimated at £6.5M.year$^{-1}$. Collection and utilisation of FOG from FSEs through anaerobic digestion has the potential generate a revenue of £6.1M.year$^{-1}$ further contributing to a net revenue of £12.6M.year$^{-1}$. The period payback for the installation of GRUs in non-compliant FSEs was therefore estimated at 9.3 years. The main barrier to full-scale implementation comes from the existence of different waste regulatory regimes. However, as TWUL is moving towards advanced energy recovery integrating pyrolysis treatment into the sludge treatment, there is an opportunity to treat the co-digestate through this process therefore diverting it from land application and in turn bypassing the need for environmental permitting. Yet, there is a need for more research to evaluate whether pyrolysis as a standalone process can achieve better energy recovery rates from FOG.

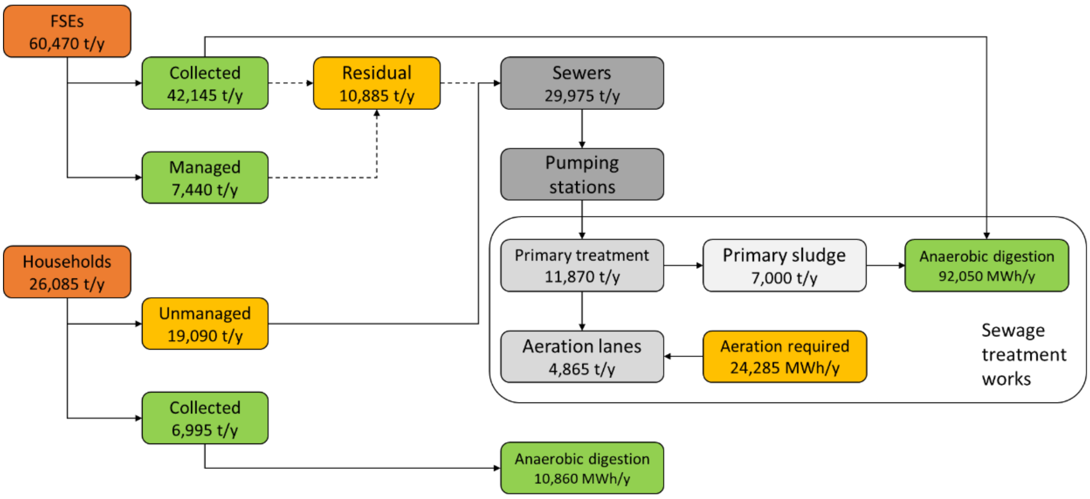


**Figure 7-6** Mass and energy balances of FOG across TWUL catchment area after implementation of source-control programme.

# 8 Implications of this work for the management of FOG

Consideration of the above findings as a whole enables identification of a number of recommendations with respect to the design and implementation of future fats, oils and grease (FOG) programmes.

## 8.1 Perspectives for a water utility

Until now, FOG collected from food service establishments (FSEs), in the form of grease trap waste (GTW) has been the most studied feedstock for energy recovery. By contrast, other materials, found in the sewers, at pumping stations or at the treatment works, have attracted little attention. In this research, we demonstrated that these other types of FOG would be desirable substrates for anaerobic co-digestion when looking at their biomethane potentials (Chapter 2). Of particular interest for water utilities, it could be argued that these materials do not classify as food waste anymore (i.e. conversely to FOG at source). Therefore, utilising these wastes in anaerobic digesters treating sludge would not require any additional permitting for disposing of the digestate. However, as FOG get exposed to sewage, it gets enriched in water and other contaminants (e.g. unflushable items) possibly requiring an additional conditioning step prior utilisation. The practicality of harvesting FOG deposited in the networks purely on an energy generation motivation was further questioned. However, it is common practice within the industry to landfill or discharge the extracted materials at the inlet works. Diverting these wastes would avoid exerting an additional organic load onto the biological treatment step at the sewage treatment works (STWs). Further investigations are recommended to identify a suitable technology for conditioning prior to utilisation and avoid hindering anaerobic digesters.

A more practicable option would be to improve FOG capture at the STW. In municipal wastewater treatment, a dedicated pre-treatment for the removal of FOG at the front end of the STW is not common as it requires the inclusion of additional assets (Pastore et al., 2015). This research, based on historical data, provided evidence of poor FOG removal achieved in conventional primary sedimentation tanks (33-64% removal). Whilst enhancing FOG removal, upgrading primary treatment with dissolved air flotation (DAF) was not found to be economically conclusive purely based on FOG-related economics. By contrast, chemical dosing, often implemented for phosphorous removal, demonstrated the potential to reduce downstream impacts on the biological treatment step by increasing FOG removal rates to 64-76%. Ultimately, if an end-of-pipe approach was to be considered, existing primary systems could be upgraded to incorporate chemical dosing for enhanced FOG capture at significantly lower investment costs (Chapter 6).

## 8.2 Perspectives for a source-based approach

In Chapter 3, we demonstrated that households, which are often a lower priority for FOG interventions in comparison to FSEs, would have a significant impact on sewer systems in densely populated area. Information gathered from trial participants identified several FOG pathways in a domestic kitchen and could be further used by stakeholders, including local councils overseeing household waste management and water utilities concerned about the impact of FOG on their assets, to develop a comprehensive recommendation guide.

Similarly, through a survey of FSEs, we extended the knowledge on FOG pathways in commercial kitchens (Chapter 4). This work also highlighted that FOG is not perceived as an issue for FSEs, possibly as they are removed from the problem they create, and as such do not justify investments in FOG management. Raising awareness on the FOG problem as well as on remediation techniques available should therefore be an integral part of a utility FOG programme.

Currently, such interventions are conducted by water utilities individually. Whilst there are opportunities to engage with other stakeholders (e.g. local councils) to design a more robust approach, a case-by-case approach could help maximising cost effectiveness of enforcement. For instance, experimental data gathered in Chapters 3 and 5 could be used to model catchments and identify hotspots where benefits from customer education would be the highest in terms of (1) opportunities to reduce sewer incidents and (2) volumes of FOG collectable.

### 8.2.1 Perspectives for energy recovery from FOG

A significant number of publications have covered the treatment of GTW, recovered from grease interceptors or separators in FSEs. Still, automated units, referred to as grease removal units (GRUs), have become more popular over the last years. In comparison, these units remove periodically the FOG component from the device further posited to be “purer” than GTW. In Chapter 5, we demonstrated that GRU wastes were associated with smaller volumes of water than GTW, possibly making transportation more efficient and reducing the need for conditioning prior to utilisation. The second hypothesis formulated in this Chapter was around the utilisation of FOG as a co-substrate with thermo-hydrolysed (TH) sewage sludge. Over recent years, the UK water sector has largely invested in sludge pre-treatment technologies such as thermal hydrolysis. This technique helps overcoming slow degradation rates by solubilising organic matter and making it more readily available to microorganisms. It was suggested that adding FOG into TH sludge could pose a potential risk as the easily degradable material may limit uptake of the more complex FOG compounds and hence may restrict the value proposition of FOG through this route. By operating continuously stirred anaerobic reactors at lab-scale, we demonstrated there was no reduction in the utilisation of FOG when co-digested with TH sludge. Nevertheless, we highlighted a possible impact on the dewaterability behaviour of the co-digestate requiring further research before full-scale implementation as this would affect process economics. Further to this, to improve operators’ confidence, the determination of safe boundaries and optimal parameters would be required. For the water utility, adopting a strategy around co-digestion would also require identifying assets with spare capacity and optimise FOG transportation accordingly. On this latter point, with smaller volumes of water associated with the collected wastes, GRUs could help optimise logistics to support the case for energy recovery.

Whilst we demonstrated FOG collected at source to be a valuable substrate for anaerobic co-digestion with sewage sludge, the current legal framework in the UK does not support the process economics. Utilities would have to seek alternative uses of the digestate to land-based application such as incineration, gasification or pyrolysis. If using advanced thermal treatment is to be considered, it is possible that by-passing anaerobic digestion for FOG treatment could become more attractive as it would also reduce operational risks linked to the accumulation of long-chain fatty acids (Chapters 2, 3 and 5).

# 9 Conclusions and future work

## 9.1 Conclusions

The overall findings of this research enabled the development of a business case for an improved management of fats, oils and greases (FOG) emissions from households and food service establishments (FSEs). Managing FOG at source would provide significant reduction in operational costs for the water utility. The utilisation of collected FOG for energy recovery has the potential to offset these costs creating a revenue stream. The specific conclusions in relation to the original objectives are as follows:

**Objective 1**: To clarify the variation amongst FOG wastes collected at different locations in the wastewater catchment (i.e. source, networks and STWs), in terms of their physicochemical properties, through laboratory analyses, and biomethane potentials using batch reactors.

- From source to treatment, FOG can be collected at several points in a wastewater catchment: at source, in the sewers, at pumping stations and at treatment works. A comparative study of their respective physicochemical properties demonstrated that not only FOG collected further away from source was getting enriched in contaminants (e.g. unflushable items) but also in water further reducing the lipids fraction by mass and thus its energy value. In turn, this reduces the value proposition of FOG collected further away from source (Chapter 2).
- The lipid-rich nature of the FOG wastes translated into high methane potentials ranging between 773-981 STP mL $CH_4$.g $VS_{added}^{-1}$. Yet, whilst FOG from networks and STW still has high values for biogas generation through anaerobic digestion, the practicality and feasibility of collecting these wastes was suggested to reduce any benefits from energy recovery (Chapter 2).
- Long chain fatty acids (LCFAs) with greater inhibitory actions on the anaerobic digestion process were found in higher concentrations in FOG collected at source. Further away from source, a shift from unsaturated to

saturated fatty acids happened possibly reducing the toxicity of FOG wastes (Chapter 2).

**Objective 2**: To collect FOG from a set number of households to generate quantitative and qualitative data on FOG from domestic sources to inform decision making for energy recovery from these materials in relation with current regulatory framework.

- The pilot trial collecting FOG from 31 households demonstrated production rates ranging between 0.1 and 6.4 kg.year$^{-1}$ per household with an average of 2.3 kg.year$^{-1}$ per household, further enabling the validation of theoretical estimates available in literature. To put this into context, this equates to around 62,380 tonnes.year$^{-1}$ theoretically collectable (Chapter 3). If only the London area (i.e. Thames Water Utilities' catchment) were to be considered, this would represent 14,920 tonnes.year$^{-1}$ (Chapter 2).
- FOG from domestic sources had lower water and free fatty acids (FFA) contents than those reported in literature for FOG from FSEs, suggesting potential applications in biodiesel production (Chapter 3).
- From these results, domestic FOG presented a high value as a co-substrate for anaerobic digestion yielding 942±36 STP mL $CH_4$.g $VS_{destroyed}^{-1}$. In comparison, thermo-hydrolysed (TH) sewage sludge yielded 645±141 STP mL $CH_4$.g $VS_{destroyed}^{-1}$ (Chapter 3).
- Whilst the most predominant collection routes for FOG advised by local councils were collection point at their household waste recycling centre (HWRC) (56%), food waste collection (15%) and into the waste bin (9%), none of the trial participants were disposing of their FOG this way. To maximise energy recovery from FOG, it was suggested to design schemes presenting a better fit with households' daily rhythms. Several opportunities could be considered including joined-up approach with other stakeholders (e.g. local authorities) to ensure sustainable domestic FOG management (Chapter 3).

**Objective 3**: To produce a comprehensive understanding of the contribution of current practices in kitchens to FOG discharges.

- Whilst UCOs were identified as the main source of FOG by FSE operators, the survey of 107 premises demonstrated other pathways related to washing and cleaning of kitchen appliances were often unacknowledged possibly contributing to FOG emissions into drainage systems. From information gathered on operation and cleaning practices, four appliances were suggested with high risks of contributing to FOG discharges in the networks namely: pre-rinse sinks, dishwashers, extraction hoods and fryers (Chapter 4).
- However, most of the FSEs surveyed (69%) did not have a remediation system installed in their kitchen to minimise these emissions. Ultimately, only 15% of the respondents were physically removing FOG from these effluents (Chapter 4).
- Educational campaigns raising awareness on FOG are posited to be a critical part of any FOG control programme, particularly when considering that 65% of the respondents never had experienced FOG-related problems, leaving sewer operators dealing with their discharges (Chapter 4).
- From field experience using grease removal units (GRUs), not all grease contamination points were found contributing to emissions further suggesting a case-by-case approach to FOG management rather than a one-size-fits-all one. Similarly, other parameters such as food serving were suggested impacting FOG generation in FSEs (Chapter 5).

**Objective 4**: To use GRUs to quantify volumes of FOG collectable from FSEs and investigate the utilisation of FOG in anaerobic digesters treating thermo-hydrolysed (TH) sewage sludge.

- FOG was collected from 14 FSEs using GRUs, and quantities produced were found ranging from 0.9 up to 18.0 g per meal served with a median value of 3.9 g per meal. Whilst the sample population was relatively small, it was suggested that FOG production rates were affected by food serving (e.g. more FOG being discharged from food served on ceramic plates). FOG generation was also dependant on the type of cuisine, and not always related to the use of cooking oils (Chapter 5).

- GRUs generated FOG wastes with significantly lower water content, varying from 2.7% to 32.3%, than that reported in literature from FOG collected through passive grease separators. In turn, this indicated a potential benefit for GRUs over passive systems when considering collection and utilisation of FOG from FSEs (Chapter 5).
- The successful utilisation of FOG as a co-substrate was demonstrated in semi-continuous conditions with both conventional and TH sewage sludge. Using TH sludge in anaerobic digesters with FOG did not lead to a reduction in the utilisation of the added material. However, with higher loading rates, tested at 4.9 kg $VS.m^{-3}.d^{-3}$, TH sludge would allow the processing of higher FOG loadings compared to that of conventional anaerobic digesters (Chapter 5).

**Objective 5**: To use historical datasets to determine the performance of conventional primary clarifiers at STWs in removing FOG and to investigate the potential of enhanced primary treatment as an alternative treatment in order to understand the business case for end-of-pipe FOG strategies.

- Using historical data, FOG (measured as hexane extractable material) were found at concentrations of 57±11 $mg.L^{-1}$ across the 15 STWs sampled (Chapter 6).
- Conventional primary clarifiers used in the treatment of sewage at STWs were able to remove 52% on average of the incoming FOG load with effluent concentrations ranging from 18.2 $mg.L^{-1}$ to 35.8 $mg.L^{-1}$. Effluents from chemically-assisted primary treatments ranged from 14.1 $mg.L^{-1}$ to 21.9 $mg.L^{-1}$ corresponding to a 71% FOG removal on average. In comparison, dissolved air flotation (DAF) achieved FOG concentrations effluents as low as 10.3 $mg.L^{-1}$, and removing 74% of the incoming FOG loads (Chapter 6).

- Ultimately, chemical dosing was found able to provide significant savings in terms of operational costs for the water utility (e.g. load reduction to secondary biological treatment and improvement of the quality of digesters' feed). By contrast, the case to switch from sedimentation to DAF purely on an economic basis was not supported. Yet, with lower CapEx and footprint, DAF appeared as a sensible option over primary clarifiers if the works require upgrading to accommodate increased flows or loadings (Chapter 6).
- In terms of FOG management, as upgrading all STWs is not realistic, it requires identifying where the benefits would be highest to be weighed against on-going operational costs from sewers (Chapter 6).

**Objective 6**: To examine the current economic impact of FOG in a wastewater catchment and develop a business case for the collection of FOG at source.

- Using Thames Water Utilities Ltd. (TWUL) catchment as a study area, it was initially estimated that 79,810 and 14,920 tonnes of FOG per year were generated respectively from around 50,000 food outlets and 6.3M households (Chapter 2).
- Using data collected from the pilot trials, this assessment was re-evaluated at 60,470 and 26,085 tonnes of FOG per year produced respectively from food outlets households (Chapter 7).
- For TWUL, the annual operating costs in dealing with FOG discharges were estimated at £16.6M. The energy generation from FOG removed with primary sludge was 36,505 MWh.year$^{-1}$ (Chapter 7).
- A prediction model for the management of FOG at source was developed. Whilst diverting FOG from the sewers contributed to a loss of £2.3M.year$^{-1}$ from the sludge stream through anaerobic digestion, OpEx savings of £10.7M.year$^{-1}$ were found achievable in the networks, pumping stations and STWs. Furthermore, collection and utilisation of FOG from FSEs through anaerobic co-digestion with sewage sludge contributed to a net revenue of £12.6M.year$^{-1}$ equating to a payback period of 9.3 years to repay capital investment of grease separators (Chapter 7).

- The existence of different regulatory regimes governing the treatment of FOG and sewage sludge is currently the main limitation to full-scale applications. Yet, TWUL moving towards advanced energy recovery with pyrolysis processes is a considerable opportunity to divert the co-digestate from conventional application to agricultural land (Chapter 7).

## 9.2 Suggestions for further works

In the course of this project further areas for research have been identified:

- The survey of FSEs highlighted several kitchen practices contributing to FOG emissions. To date the characterisation of kitchen effluents has been often limited to pre-rinse sinks and dishwashers (Chung and Young, 2013; Gurd, 2018). To fully understand the relative contribution of cleaning regimes to FOG emissions, and in turn to be able to select the suitable remediation technique, there is a broad need for further work to be conducted. It is posited that the Gerber method developed by Gurd et al., (2018) allowing the measurement of both free and emulsified FOG would provide a more reliable and complete picture over liquid-liquid and solid phase extraction techniques. Providing a reliable characterisation of the different FOG streams in a kitchen would help tailoring suitable approaches, both educational and technical, for the management of FOG in kitchens rather than relying on a one-size-fits-all strategy. As FOG management can represent a significant cost for a FSE, such work would help prioritising and targeting sources of FOG to maximise cost effectiveness.
- It has been hypothesised that the collection of FOG from domestic sources would be maximised by designing schemes presenting better fits with households' daily routine (Foden et al., 2017; Seyring et al., 2015). However, disparities were found between advices provided by local councils and water utilities for the management of FOG. For instance, only 15 out of the 73 local authorities organising food waste kerbside collection were accepting FOG. To drive forward sustainable practices, it is

recommended to understand what the drivers and barriers to implementation for each stakeholder are.

- The successful anaerobic digestion of FOG with both non-hydrolysed and thermo-hydrolysed sewage sludge was demonstrated in semi-continuous conditions. However, this research suggested the addition of FOG to reduce the digestate dewaterability when using the capillary suction time (CST) as a proxy. Critically, affecting the digestate rheological behaviour would have an impact on polymer requirement and in turn on process economics. The long-term operation of anaerobic co-digesters at larger scale is recommended to (1) evaluate the process performance when scaling up and (2) quantify any impacts on downstream dewatering of digestate.
- As TWUL is moving towards advanced energy recovery from sludge incorporating pyrolysis technologies, there is an opportunity for anaerobic co-digestion bypassing co-digestate disposal to agricultural land. Yet, research is required to compare recovery rates from FOG through pyrolysis as a standalone process against its incorporation into the advanced energy recovery stream (i.e. anaerobic co-digestion coupled with pyrolysis).

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# Appendices

## Appendix A International management of FOG

**Table A 1** Comparison of key parameters for the testing of grease abatement devices.

| **Standard** | **BS EN 1825** | **A112.14.3** | **A112.14.4** | **B481 Series 12** | **PDI G101** |
|---|---|---|---|---|---|
| **Origin** | European Union | USA | USA | Canada | USA |
| **Unit type** | Grease interceptor | Grease interceptor (of 380 L/min or less) | GRU (of 380 L/min or less) | Indoor (under-the-counter) and outdoor (underground) grease interceptor (26 to 380 L/min) | GRU (automated function must be deactivated) |
| **Media** | Fuel oil (ρ: 0.85g.cm$^{-3}$) | Lard (ρ: 0.87g.cm$^{-3}$) | Lard (ρ: 0.87g.cm$^{-3}$) | Sunflower oil (ρ: 0.92g.cm$^{-3}$)<br>Lard (in accordance with ASME A112.14.3) | Lard (ρ: 0.87g.cm$^{-3}$) |
| **Temperature** | 4 to 20˚C | 66 to 71˚C | 43±3˚C | 23±3˚C | 66 to 71˚C |
| **Compliance** | 99-ish% removal | 90% removal / 80% removal with retention | Volume of grease removed not less than 50% of its rated capacity | Removal efficiency to be determined by target effluent quality or amount of FOG to be removed | 90% removal / 80% removal with retention |
| **Cleaning and maintenance** | Emptied, cleaned and refilled at least once a month (preferably every two weeks) | Once a week to once in several weeks | Provided by manufacturer | Several times a week to once every few weeks | Once a week to once in several weeks |

**Table A 2** International experience with FOG management. Information has been collected from personal communication with stakeholders involved in FOG programmes (as of 2016) and information available in literature.

| Location | Domestic | FSEs – Enforcement | FOG utilisation |
|---|---|---|---|
| **USA, Dallas** (Helms and Dulac, 2016) | Residential FOG programme ("Cease the Grease") was promoted through extensive educational and awareness campaigns.<br>To divert this FOG from the sewers, drop-off locations have been installed around the city of Dallas. | The FOG programme was initiated through a local ordinance issued amending the City Code to make the installation of grease abatement devices compulsory. Their sizing is determined by the Development Services Plumbing Inspectors. A consent limit of 200 mg.L$^{-1}$ of FOG in kitchen effluent has been defined. The Liquid Waste Section regularly conducts inspections unannounced and avoiding rush hours. Interceptors' cleaning frequency is required on a 90-day basis or based on a 25% rule (i.e. volumes of FOG and sludge representing 25% of the total interceptor volume). Proofs of maintenance are a legal requirement and recorded through a Liquid Waste manifest tracking any activity from collection to disposal of the GTW. | The relatively small volumes of domestic FOG collected are fed into the anaerobic digesters treating sludge at the STW.<br>At the time of communication, GTW collected by licensed waste haulers was not used at the STW for co-digestion. |

| | | | |
|---|---|---|---|
| | | The use of bio-additives is allowed but grease abatement devices are still required and shall be maintained at the same frequency. | |
| **USA, Orange County** (Seiler, 2016) | Domestic FOG is mainly targeted through education and outreach programmes focusing on best disposal practices. | Each FSE is inspected at least once a year to ensure compliance. Grease interceptors' cleaning frequency is either set at 90 days or based on the 25% rule. FSEs are required to send a record of their activities regarding GTW and UCO collection to the Orange County Sanitation District twice a year. There is no consent limit for kitchen effluents as this was difficult to enforce.<br>The use of bio-additives is forbidden unless a specific authorisation is obtained. | |
| **USA, San Francisco (Ho and Miot, 2016)** | For the management of this FOG, drop-off units are dispersed in town and collected once or twice a week. | | The water utility provides food outlets with barrels to dispose of their UCOs and collected free of charge. UCOs are refined through a heated sedimentation process. Food, water and particulates are collected and fed to anaerobic digesters (treating sewage sludge). |

| | | | |
|---|---|---|---|
| | | | At another one of their plants, the utility has trialled a FOG-to-biodiesel project using GTW as a feedstock. The process was a combination of mixing and heating in three consecutive tanks but was abandoned after a succession of technical issues. Since then, the utility has been successfully co-digesting pre-treated GTW and sewage sludge (Miot et al., 2013) – at the time of communication, co-digestion was halted due to transitioning to thermophilic digestion. |
| **USA, Oakland** (Hake, 2016) | Through their FOG residential programmes, drop-off locations are available for the disposal of FOG. | Similarly to other US municipalities, a local ordinance was issued to make the presence of grease interceptors compulsory in FSEs. Grease trap maintenance log should be kept and make available for inspectors. | GTW and other high-strength wastes (e.g. dairy and food processing, slaughterhouse) are successfully co-digested with sewage sludge at the STW (Suto et al., 2006). |
| **Ireland** (Wallace et al., 2017) | | Since 2008, FSEs have been required to have suitable grease trapping system installed. FSEs have to apply for a trade effluent licence | |

| | | |
|---|---|---|
| | to cover their FOG discharges. A discharge consent of 100 mg.L$^{-1}$ is set. | |
| **Singapore** | Grease management systems have been imposed to commercial and industrial food facilities since 1981. Designs are recommended by the Code of Practice for Sewerage and Sanitary Works. The utility runs grease traps inspection programme to ensure compliance of FSEs. Premises are advised to determine their optimum cleaning frequency by monitoring discharges. Nevertheless, they recommend to clear and clean these systems at least once every two weeks. | Grease waste are collected by Licensed Waste Collectors' vacuum tankers from the food facilities and disposed at the STW for treatment. Screens are used to filter large particles, and dissolved air flotation DAF is used to concentrate greasy waste further used as a co-substrate with sewage sludge. |
| **Australia** | FSE discharges are defined as trade waste and required to have a grease abatement device. A formal consent is required for the installation of grease separators. Proofs of maintenance are required (Georges et al., 2017). | |

# Appendix B Supplementary materials to paper 1

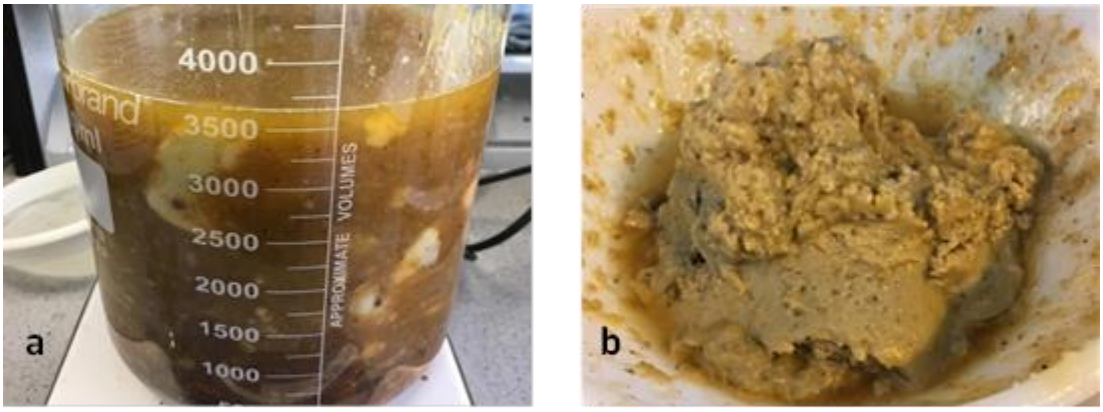

**Figure B-1** FOG collected from (a) households and (b) a FSE’s grease removal unit.

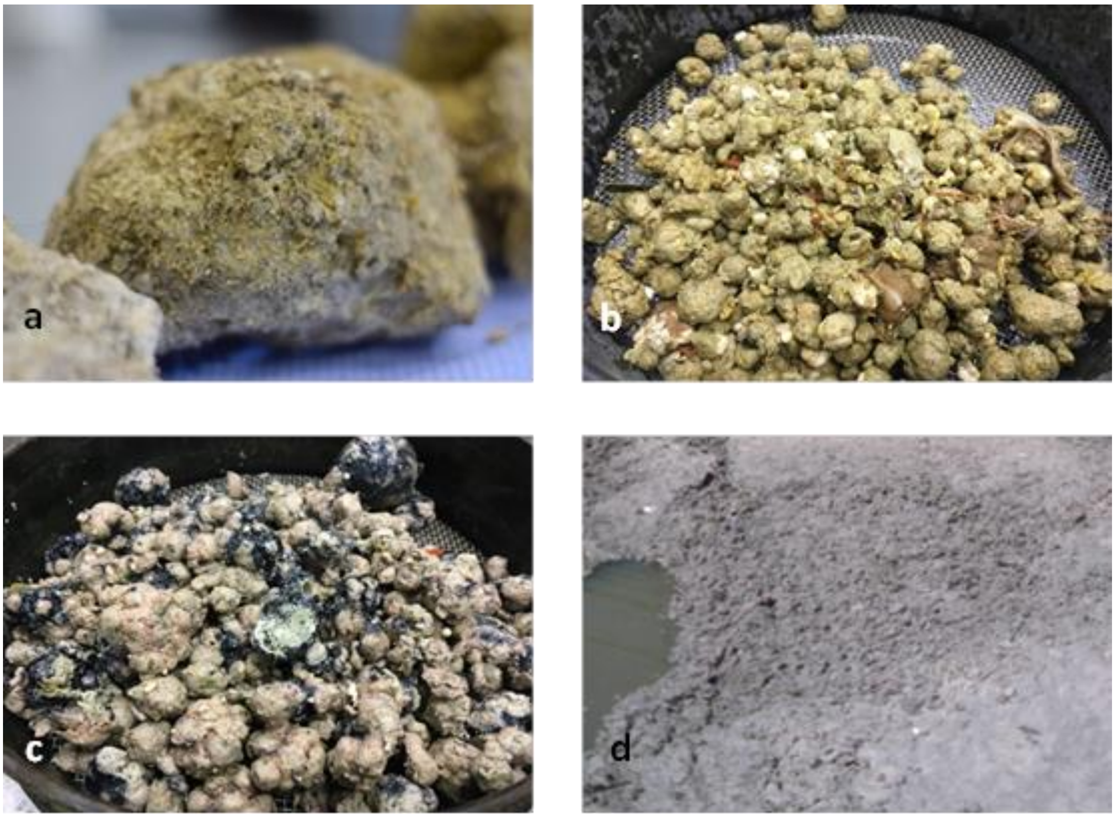

**Figure B-2** FOG collected in the wastewater system from (a) sewers (fatberg), (b) pumping station (fat balls), (c) inlet works (fat balls) and (d) floating scum from inlet works.

# Appendix C Supplementary materials to paper 2

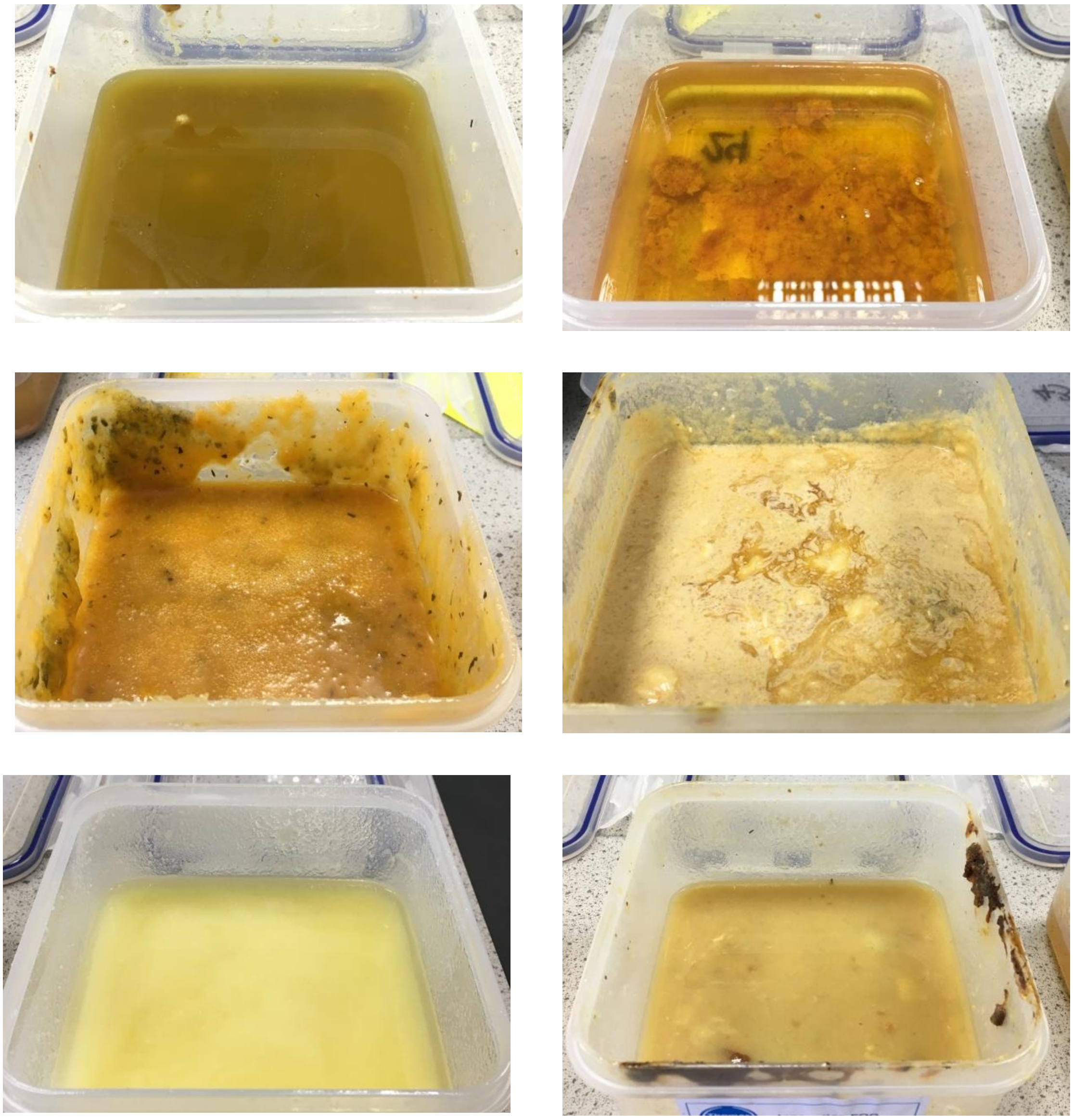

**Figure C-1** FOG collected from six different households.

# Appendix D Survey of households

## D.1 Initial survey

- How do you currently manage FOG in our household? (e.g. pour down the sink, put in bin, food waste or compost heap)
- How many people live in your household? And how many will be contributing to the FOG collection?
- What is the age demographic of those cooking in your household?
- What nationality are you? Do you tend to cook traditional food?
- Are you aware of Thames Water's "Bin it, don't block it" campaign? If so, has it affected your kitchen waste routine?

## D.2 Mid-trial survey

- What types of FOG do you use when cooking? (e.g. vegetable oil, butter, ghee)
- What types of FOG did you collect?
- Where did you collect the FOG from?
- Which appliances collected the largest volumes?
- Please comment on any issues you have encountered during the trial (e.g. design of pot, odour).

# Appendix E Additional information on GRUs

**Table D-1** Details on GRUs installed at the 14 FSEs.

| GRU | FSE | Unit type | Connected appliances |
|---|---|---|---|
| **1** | 1 | Big Dipper W-250-ISE | Double bowl sink |
| **2** | 2 | GS1850-S-PF-DUAL | Wash sink and dishwasher |
| **3** | 3 | GS1850-S-PF-DUAL | Wash sink and dishwasher |
| **4** | 3 | GS1000-AST-PF | Combination oven |
| **5** | 4 | Big Dipper W-250-ISE | Double bowl sink |
| **6** | 5 | Big Dipper W-250-ISE | Wash sink and dishwasher |
| **7** | 6 | GS1850-L-PF-DUAL | Triple bowl sink and dishwasher |
| **8** | 6 | GS1000-LL-FLF | Combination oven |
| **9** | 7 | GS1850-L-PF-DUAL | Double bowl sink and dishwasher |
| **10** | 8 | GS1850-L-PF | Triple bowl sink and dishwasher |
| **11** | 9 | GS1850-L-PF | Double bowl sink |
| **12** | 9 | GS1850-L-PF | Pre-rinse sink and dishwasher |
| **13** | 9 | GS1850-L-PF | Double bowl sink |
| **14** | 9 | GS1850-L-PF | Pre-rinse sink and dishwasher |
| **15** | 9 | GS1850-L-PF | Pre-rinse sink and dishwasher |
| **16** | 10 | GS1850-L-PF | Pre-rinse sink and dishwasher |
| **17** | 10 | GS1850-LL-PF | Combination oven |
| **18** | 10 | GS-1850-AST-PF | Pot wash sink |
| **19** | 10 | GS1850-AST-PF | Pot wash sink |
| **20** | 11 | GS1850-L-PF | Pre-rinse sink |
| **21** | 12 | GGX7 | Triple bowl sink |
| **22** | 13 | GGX15 | Pre-rinse sink |
| **23** | 13 | GGX25 | Pot wash sink |
| **24** | 14 | GGX25 | Double bowl sink |

# Appendix F Additional data from semi-continuous anaerobic co-digestion of FOG

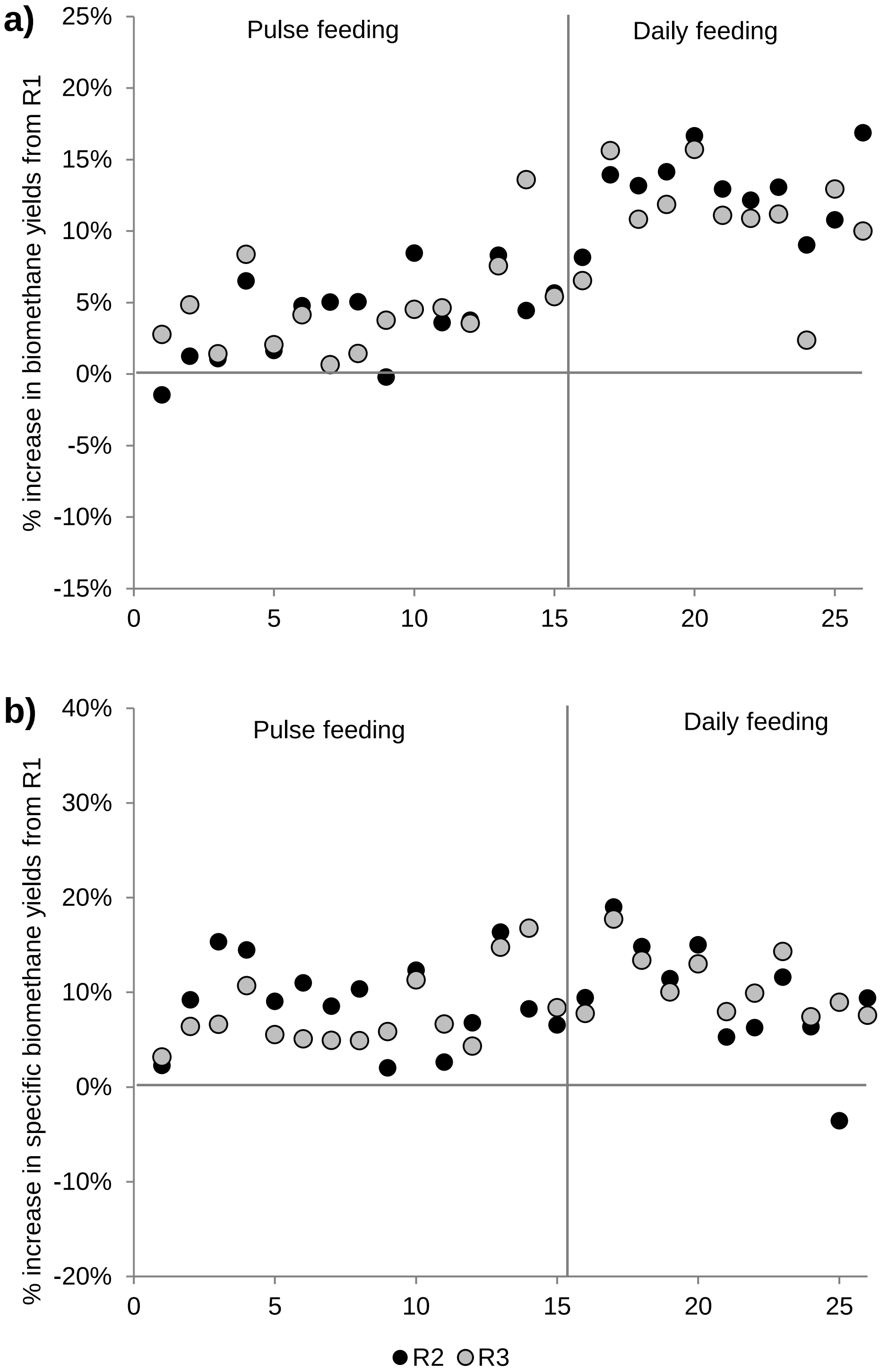

**Figure F-1** Increase in biomethane yields per gram VS added (a) and destroyed (b) for R2 and R3 compared to R1.

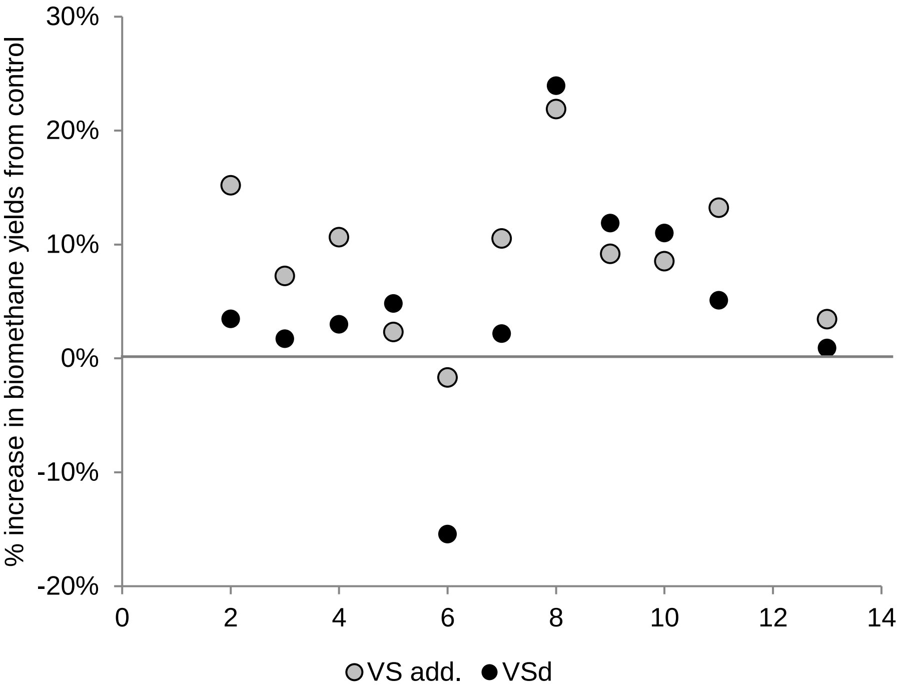

**Figure F-2** Increase in biomethane yields per gram VS added and stroyed for R5 compared to R4.