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DIVERTING FOOD WASTE FROM LANDFILL: A CHALLENGE FOR THE WATER INDUSTRY

A thesis submitted to Imperial College London for the degree of Doctor of Philosophy

By

Eleni Iacovidou

Centre for Environmental Policy

Faculty of Natural Sciences

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DECLARATION OF OWN WORK

I declare that this thesis

'Diverting food waste from landfill: A challenge for the water industry' is entirely my own work and that where any material could be construed as the work of others, it is fully cited and referenced, and/or with appropriate acknowledgement given.

Signature:.....

Name of student: Eleni Iacovidou

Name of supervisor: Nikolaos Voulvoulis

ABSTRACT

Food waste management is currently attracting both regulatory and public attention due to new greenhouse gas emission targets, targets relating to waste reuse, recycling and the diversion of waste from landfill, as well as other policies related to sustainability. The research presented in this thesis addressed two of the management options currently available namely the use of food waste disposal units (FWDs) and the anaerobic co-digestion of food waste with sewage sludge, because of their relevance to the water industry.

The use of FWDs was studied and results demonstrated that they can bring substantial benefits to local authorities through reductions in cost from waste collection, but can also present challenges to the water industry due to the need to treat additional wastewater. Policy implications were identified, and given the prospect of FWDs application, the incorporation of anaerobic treatment processes into municipal wastewater treatment operations was investigated as a viable technology which could be implemented to give environmental and economic benefits independent of the prevailing climatic conditions.

The anaerobic co-digestion of food waste with sewage sludge, well-established in other European countries but still in its infancy in the UK, was investigated as an alternative food waste management option. With sustainability becoming a driver for change in waste management practices, this option was found to have considerable potential, but significant capital investment and operational expenditure could be required to create new assets or upgrade existing ones.

A multi-criteria framework was developed to support decision-making aim to select the appropriate food waste management option based on sustainability. It was concluded that diverting food waste from landfill is not only a challenge, but an opportunity that can lead to more sustainable solutions. Balancing the relationship between water, food and energy is the real means to viable solutions and the ultimate approach to deliver overall benefits.

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Nomenclature

ABP	Animal by-products
ABPR	Animal by-products regulations
AD	Anaerobic digestion
Ag	Silver
Al	Aluminium
AMP	Asset management cycle
As	Arsenic
B	Specific microorganism decay rate
B	Boron
BOD	Biochemical oxygen demand
BT	Biological treatment
Ca	Calcium
$CaCO_3$	Calcium carbonate
Cd	Cadmium
CH_3COOH	Acetate
CH_3OH	Methanol
CH_4	Methane
CMAS	Completely-mixed activated sludge reactor
CO	Carbon monoxide
Co	Cobalt
CO_2	Carbon dioxide
COD	Chemical oxygen demand
Cr	Chromium
Cu	Copper
d_{CH_4}	Density of methane
Defra	Department for Environment, Food and Rural Affairs
EA	Environment agency of England and Wales
EGSB	Expanded granular sludge bed
EU	European union

Fe	Iron
FOG	Fat, oil and grease
F_W	Volumetric flow rate of the organic substrate
FW	Food waste
FWDs	Food waste disposal units
GHG(s)	Greenhouse gas emissions
GPS-X	General purpose simulator
GWP	Global warming potential
H_2	Hydrogen
H_2O	Water
H_2S	Hydrogen sulphide
HCO_3^-	Bicarbonate ion
$HCOOH$	Formic acid
Hg	Mercury
HI	Health impacts
HRT	Hydraulic retention time
IPPC	EU pollution, prevention and control permit
k	Maximum specific substrate utilisation rate
K	Potassium
k_h	First order hydrolytic kinetic constant rate
K_S	Saturation constant
LAs	Local authorities
LCA	Life cycle assessment
LCFAs	Long chain fatty acids
MCA	Multi-criteria approach
$MeSH$	Methane thiol
Mg	Magnesium
Mn	Manganese
Mo	Molybdenum
MSW	Municipal solid waste

N	Nitrogen
Na	Sodium
NH_3	Ammonia
$NH_3 - N$	Ammonia-nitrogen
NH_4^+	Ammonium ion
NH_4HCO_3	Ammonium bicarbonate
Ni	Nickel
NI	Noise impacts
NO_2^-	Nitrites
NO_3^-	Nitrates
O_2	Oxygen
OC	Operational cost
OFWAT	Office of Water
OLR	Organic loading rate
P	Phosphorus
PA	Public acceptance
PAS	Publicly available specification
Pb	Lead
PCS-BT	Primary chemically enhanced sedimentation followed by biological treatment
PS	Primary sludge
PS-BT	Primary sedimentation followed by biological treatment
RC	Resources consumption
RE	Renewable energy generation
RN	Revenue
ROC	Renewable obligation certificates
rWFD	Revised Waste Framework Directive
r_x	Bacterial growth rate
S	Substrate concentration
S	Sulphur
S^{2-}	Sulphide

Se	Selenium
SI	Smell impacts
Sn	Tin
S_0	Influent substrate concentration
SO_4^{2-}	Sulphate
SRB	Sulphate reducing bacteria
SRT	Solids retention time
SS	Suspended solids
t	Time
TKN	Total kjeldahl nitrogen
TP	Total phosphorus
TSS	Total suspended solids
UASB	Upflow anaerobic sludge bed
VFAs	Volatile fatty acids
VOCs	Volatile organic compounds
V_R	Anaerobic reactor volume
VS	Volatile solids
W	Tungsten
WW	Wastewater
WAS	Waste activated sludge
WWTPs	Wastewater treatment plants
X	Biomass concentration
X_{degr}	Concentration of biodegradable organic matter
Zn	Zink
μ	Specific bacterial growth rate
μ_{max}	Maximum specific growth rate

1 INTRODUCTION

The disposal to landfill of the increasing amount of municipal solid wastes (MSW) as a result of increased population and living standards has raised environmental concerns, that have been the main drivers for environmental regulations across the world (Marashlian & El-Fadel, 2005). This is because MSW disposed of to landfills leads to the formation of landfill gas and leachate, which can be associated with negative environmental impacts.

Landfill gas and leachate are produced during the anaerobic decomposition of biodegradable waste, which is the largest proportion of MSW (Lou & Nair, 2009). Biodegradable waste is "*any waste that is capable of undergoing anaerobic or aerobic decomposition, such as food waste and garden waste, and paper and paperboard*" (European Council, 1999). Food waste is perhaps the most challenging fraction of biodegradable waste, mainly because of its high moisture and easily degradable organic content that make its handling a rather complicated task.

As a result of increased environmental concerns, other food waste management options as an alternative to landfill have been introduced. These are kerbside collection and recycling, centralised incineration, home and centralised composting (De Koning & Van der Graaf, 1996; Diggelman & Ham, 2003; Wang et al., 2001), anaerobic digestion with and without sewage sludge (Edelmann, 2000a; GarcIa et al., 2005; Hamzawi, 1998; Hamzawi et al., 1998; Kim et al., 2004; la Cour Jansen, 2004; Mata-Alvarez et al., 2000; Sosnowski et al., 2003), thermolysis, gasification and the use of food waste disposal units (FWDs).

In the UK waste management is controlled by local authorities (LAs). LAs have the responsibility for waste collection and disposal, and activities related to waste management. They also have the responsibility for promoting different waste management options and support waste reduction (Defra, 2011c). The use of FWDs has been one of these options. FWDs practice the grinding of food waste into a homogenised mixture that is disposed into the sewer for treatment with the wastewater. While this option removes the burden of waste collection and disposal from LAs, it adds the burden of food waste treatment to the water industry.

Another food waste management option, also related to water industry operations, is anaerobic co-digestion with sewage sludge. This option involves the digestion of two substrates together as a way to improve digestion efficiency and increase the energy output. Although well implemented in other European countries, this option is still trying to find its way into the UK economy and waste management sector, as it offers important benefits, including the one of renewable energy generation. However, its implementation can be a great challenge for the water industry mainly because of its associated environmental, economic and social costs.

The following Chapter aims to provide information on waste and wastewater authorities in the UK and the adverse environmental impacts of organic waste disposal to landfills, highlighting the importance of diverting it from landfills and the significance of European legislation in achieving this goal. It also provides information on the science of anaerobic treatment and outlines the factors controlling its operation, underpinning the importance of its implementation for the treatment of food waste, and an in-depth understanding for its successful operation.

2 BACKGROUND

2.1 Waste and wastewater management in the UK

In the 19th century, water supply in the UK was managed by the private sector, regulated by local governance and controlled by parliament. In the late 19th and early 20th centuries, due to the perception that private control was not facilitating equitable service provision for all, companies responsible for the water supply and sewerage management were taken over by local authorities (LAs), which at that time were also responsible for waste management (Johnson & Handmer, 2002).

Waste management, however, had not always been the responsibility of LAs. At the beginning waste was controlled by the public, as many people made their living by selling what they found in other people's waste (Waste Online, 2004). Dust from coal fires was taken in dust-yards where men, women and children used to sieve the dust to use it as a soil conditioner and brick ingredient (Waste Online, 2004). This was until 1875, where the introduction of the Public Health Act made LAs responsible for regular waste collection and disposal, and required householders to keep their waste into "moveable receptacles" that LAs had to empty every week (Herbert, 2007; Waste Online, 2004). In 1889, the Association of Cleansing Superintendents, known today as the Chartered Institution of Wastes Management (CIWM), was established. Cleansing Superintendents were responsible for controlling the proper management of water, sewerage and waste, and dealing with any problems related to these (Herbert, 2007). In 1907, amendments to the Public Health Act gave authority to LAs to charge for waste collection. The Public Health Act in 1936 allowed LAs to prosecute over uncontrolled dumping and scavenging, and a few years later, in 1947, they were given the power to plan new waste management sites by the newly established Town and Country Planning Act (Waste Online, 2004). Because of the bad economy as a result of the World Wars waste management options could not be developed, and the large surplus of holes in the ground created by the extractive industries in the UK, landfill had become the dominant waste disposal practice, without any recognition of the associated environmental impacts (Waste Online, 2004). In 1956, the introduction of the Clean Air Act reduced the number of open fires in homes and these were replaced by central heating fuelled by oil, gas or

electricity, so that wastes such as food and paper, that previously would have been burnt, became part of the waste collected and sent to landfills (Waste Online, 2004).

Almost two decades later in 1973, the introduction of the Water Act shifted the responsibility for water and wastewater services from LAs to the newly established Regional Water Authorities. Water Authorities had the duty to regulate and manage the use of water, and wastewater treatment and disposal (Environment Agency, 2011; Johnson & Handmer, 2002). But, concerns over the capability of the Water Authorities to control the risks for which they were responsible shifted the responsibility for water supply security to the government. Although the government succeeded in not increasing the prices to customers, it failed in the management of environmental and water supply risks because of its unwillingness to invest in the ageing infrastructure and in water quality improvements (Johnson & Handmer, 2002). This was the catalyst that led the water industry to its full privatization in 1989 aiming to transform water supply and wastewater treatment and disposal from a public service focused on equity to a business organization focused on efficiency (Environment Agency, 2011; Johnson & Handmer, 2002).

As a result of these changes, wastewater and waste management in the UK have since then been controlled by different authorities. Wastewater management is controlled by the water industry, whereas waste management has remained the responsibility of LAs.

- ***Waste management***

LAs are responsible for the waste collection, disposal and other related activities such as street cleaning. They have an important role in improving waste management in their area, supporting waste reduction, reuse and recycling, while providing frequent and reliable collection services (Defra, 2011c). Upon the introduction of the EU Landfill Directive, that requires the diversion of biodegradable municipal waste from landfills, and the revised Waste Framework Directive, that requires the increase in the reuse, recycling and recovery of waste, LAs have started to make some progress in this area.

Municipal waste is the waste stream for whose collection and disposal, LAs have responsibility. Of the 44million tonnes of municipal waste collected by LAs each year, only 12million is diverted from landfills (Defra, 2011c). Although this is a great achievement compared to the waste landfilled in the past, there is still a great potential for reducing and recycling waste in even greater amounts. To achieve these expectations, LAs should make efforts to progress. Even though they have experience in the traditional and established waste management techniques, their familiarity with innovative and emerging technologies is limited (Communities, 2004). Therefore LAs investment into new and more advanced facilities that would enable the treatment of large quantities of waste in a more efficient and sustainable way is desirable (Communities, 2004). The design, planning and construction of such facilities, however, are controlled by different regulations compared to the traditional ones, and a large capital investment is also required, which may be associated with regulatory and economic constraints (Communities, 2004).

As such, mechanisms through which LAs can procure assets are required. With that intention, the Waste Infrastructure Delivery Programme (WIDP) was established to support LAs to accelerate investment in the large-scale infrastructure required to treat residual waste, without compromising efforts to minimize waste and increase recycling levels (Defra, 2011c). This programme aims to ensure cost-effective and timely delivery of the major infrastructure required to bridge the shortfall in residual waste treatment capacity needed in order for England to meet its share of the UK's Landfill Directive diversion targets (Defra, 2011c).

The UK government is also trying to support LAs by improving its policies on waste as a way to move towards a zero waste economy, where prevention is the main goal (Defra, 2011c). Flexibility in waste regulations would turn waste into a resource and provide positive environmental impacts. Through this flexibility, LAs would be able to focus more on providing waste management services in a sustainable way for their local area.

- ***Wastewater management***

The water industry has a key role in providing high quality drinking water and ensuring improved wastewater treatment. It has responsibility for providing renewable energy generation and promoting the development of clean technologies and energy efficiency activities that could assist on its commitment to reduce its greenhouse gas (GHG) emissions, thus meeting the requirements for sustainable development (Defra, 2011b; Environment Agency, 2010b; Water UK, 2010).

The delivery and treatment of water and wastewater to the high quality standards required by environmental regulations (Charlton & Arnell, 2011) have been the key drivers for increased energy use (Water UK, 2010). The water industry uses up to four per cent of the total energy used in the UK. This energy consumption has been a major problem to the water industry, mainly because of the rising cost of energy and the pressures associated with carbon emissions (Brown et al., 2010). Renewable energy generation that covers eight and a half per cent of the energy use in the water industry, has partly relieved the pressures associated with the increased energy use and the reliance of the water industry on fossil fuels. Nevertheless, the water industry has the obligation to generate more of its energy from renewable sources.

Increased renewable energy generation is therefore expected to play a key role in the water industry's sustainable development, as it will reduce carbon emissions and create financial benefits through the reduction of electricity import (Environment Agency, 2010b). A range of renewable energy sources such as hydropower, wind power, thermal generation and anaerobic digestion are currently accessible to the water industry, and can help meet its renewable energy expectations (Environment Agency, 2010b).

Anaerobic treatment has recently received considerable attention mainly because apart from the benefit of renewable energy generation, it offers the benefit of dealing with another problem the water industry is currently facing, that of sewage sludge management. Over 90% of the current renewable energy generation in the water industry is through sewage sludge digestion. The haulage, treatment and disposal of sewage sludge incurs a significant environmental and economic cost to the water

industry (Zakkour et al., 2001). This cost is expected to rise in the future as sewage sludge production is subject to increase due to the increasing population and increased levels of treatment required by the continuing implementation of the European Urban Waste Water Treatment Directive and the related legislation (Brown et al., 2010). As a result, the water industry has to investigate and promote the implementation of options such as anaerobic treatment that can either treat sewage sludge efficiently or minimise its production.

The implementation of such processes, however, can be associated with economic, regulatory and technical challenges that form a barrier to water industry development. These challenges are likely to become more critical due to the effects of climate change, the reduction of water resources, the increasing population and the ageing infrastructure (APPWG, 2008; Brown et al., 2010; Environment Agency, 2010b). As such, the major goal of the water industry is to address these challenges through holistic thinking, planning and decision making in a sustainable and cost-effective way (Brown et al., 2010).

Financing the water industry has significant implications for innovation intended to promote water sector sustainability. In England and Wales, investment follows a 5-yearly Asset Management Period (AMP) overseen by the Office of Water (OFWAT), the economic regulator of the water industry in the UK. AMP cycles aim to strike the right balance between keeping bills down and allowing Water Companies to invest in essential infrastructure and secure water. However, these five year planning cycles are arguably too short to promote long range planning for sustainability (Brown et al., 2010).

Furthermore, the regulatory system that governs the water industry operations is complex and unable to provide the industry with the right incentives that would enable it to develop more sustainable solutions to the many environmental problems that it is facing. Currently the entire amount of renewable energy generated by the water industry is subsequently used in the water and wastewater treatment processes rather than being exported for use in other purposes (Environment Agency, 2010b). This represents a great challenge for the water industry as the energy produced is not exploited beneficially. As a result, the water industry is in a continuing battle with the regulators for

recognition of its generated renewable energy value, and creation of a system that satisfies both demand and supply (Environment Agency, 2010b).

Companies and regulators have started to collaborate in order to develop a coherent framework that would enable the UK water industry to make a positive contribution to the UK efforts towards sustainable development.

2.2 The environmental impacts of landfilling

Landfill gas and leachate formation from waste landfilling have raised new environmental concerns, due to their associated environmental impacts.

Landfill gas mainly consists of methane (CH_4) and carbon dioxide (CO_2), together with small amounts of hydrogen sulphide (H_2S) and trace amounts of volatile organic compounds ($VOCs$) (Nastev et al., 2001; Qin et al., 2001). When food waste is being disposed of to landfill, the initial decomposition of organic matter occurs under aerobic conditions, where CO_2 is produced. Soon after the oxygen (O_2) is depleted, the degradation continues under anaerobic conditions, during which the organic materials are converted into CH_4 and CO_2, based on a three to four-stage process that is well described by many authors (Barlaz & Ham, 1993; Bingemer & Crutzen, 1987; El-Fadel et al., 1997a; El-Fadel & Massoud, 2000; Kjeldsen et al., 2002; Themelis & Ulloa, 2007). The process of anaerobic decomposition can last for many years (Bingemer & Crutzen, 1987; Brown & Maunder, 1994; El-Fadel et al., 1997a; Flower et al., 1981; Nastev et al., 2001; Themelis & Ulloa, 2007).

Leachate is mainly a result of the rainwater percolation through the organic mass which has been disposed of to landfills. Another pathway for its production is through the water generated from the degradation of food wastes themselves (Christensen et al., 2001; El-Fadel et al., 1997a; El-Fadel et al., 1997b; Koerner & Soong, 2000; Renou et al., 2008; Robinson & Maris, 1983). When water percolates through the wastes, it dissolves the organic and inorganic components and decomposition products generating a solution that contains many pollutants including organic matter, ammonia-nitrogen, heavy metals, and chlorinated and inorganic components. The composition and quality of leachate can vary depending on the climatic and hydro-geological conditions, waste characteristics, fermentation stage and operational procedures occurring within the landfills (Christensen et al., 2001; Chu et al., 1994; El-Fadel et al., 1997b; Kjeldsen et al., 2002; Lo, 1996; Rapti-Caputo & Vaccaro, 2006; Tatsi & Zouboulis, 2002).

- ***Global Warming***

Although energy can be recovered from landfill gas, its potential leakage to the environment is an important risk. CH_4, CO_2 and $VOCs$ (heptanes, octane, nonane, hexane, n-pentane, iso-pentane, propane, n-butane, iso-butane, acetone, dichloromethane, benzene, butyl alcohol, chloroform, terpene, limonene, toluene, xylene) found in landfill gas are powerful greenhouse gases (GHG) that can contribute significantly to global warming. Nevertheless, CH_4 has received more attention as it has a global warming potential (GWP) 21 times greater than that of CO_2 (UK WIR, 2008). It is also more effective than other GHGs at trapping infrared irradiation; controlling the oxidizing capacity in the atmosphere and affecting the stratospheric ozone layer (Cicerone & Oremland, 1988; Khalil & Rasmussen, 1994; Milich, 1999).

Historically, CH_4 emissions from landfills have been the greatest source of GHG emissions from the waste sector (Bogner et al., 2007). Global CH_4 emissions from landfills were estimated at 9-70 tera grams per year (Tg/yr) in 1987, with the latest estimates being in the range of 19-40 Tg/yr (Bingemer & Crutzen, 1987; Bogner & Matthews, 2003; Spokas et al., 2006). In the United States (U.S), landfills account for approximately 23% of the total anthropogenic CH_4 emissions, representing the second largest source of CH_4 emissions. In 2007, about 137 million tonnes of MSW were discarded in landfills, of which 11.9% was food waste (US EPA, 2008). That year, landfill CH_4 emissions were estimated at approximately 6,327 giga grams (Gg), of which 90% was from MSW landfilling, meaning that CH_4 emissions from food waste were approximately 677.6 Gg (US EPA, 2009). In Europe, the landfilling of waste represents the second largest source of CH_4 emissions (EEA, 2009). In 2007, in the European Union (EU-15) CH_4 emissions from landfills were estimated at approximately 3,745 Gg, of which 90% corresponded to MSW landfilling, with CH_4 emissions from MSW landfilling being at 3,370.5 Gg (EEA, 2009). The fraction of food waste going to landfill varies widely from country to country, making it difficult to estimate overall emissions from food waste disposal. It must be noted, however, that some European counties such as, Germany and Sweden, have introduced total bans on food waste going to landfill, and thus contribute less to CH_4 emissions.

- ***Fires and Explosions***

Besides its contribution to global warming, CH_4 has the ability to migrate away from landfills resulting in fire and explosion hazards, and vegetation damage (Christophersen & Kjeldsen, 2001; El-Fadel et al., 1997a; Kjeldsen & Fischer, 1995; Nastev et al., 2001; Raybould & Anderson, 1987; Scheutz et al., 2009). CH_4 is a flammable gas and when its concentration in air is in the range of 5-15%, an explosion can occur (Christophersen & Kjeldsen, 2001; Nastev et al., 2001; Raybould & Anderson, 1987), while fire hazards are also possible (Calvo et al., 2007). Onsite fires can occur in the subsurface of the landfill due to air entrapment and the formation of a mixture of CH_4 and O_2 that can sustain a fire (El-Fadel et al., 1997a; Stearns & Petoyan, 1984). Incidents of fires and explosions due to gas migration away from landfills have been reported in the literature (Kjeldsen & Fischer, 1995; Raybould & Anderson, 1987). Among them was an accident that took place at Skellinsted, Denmark in 1991 that was a result of CH_4 migration away from landfill through the underground soil.

- ***Vegetation Damage***

The migration of CH_4 to the root zone of the landfill area and its oxidation near the soil surface can both contribute to O_2 deficiency that causes damage to vegetation and affects plant growth (Börjesson et al., 2004; El-Fadel et al., 1997a; Flower et al., 1981; Gilman et al., 1982; Wong, 1988). Heat release during CH_4 oxidation increases the soil temperature creating a potential for plant asphyxia, while CO_2 from CH_4 oxidation can be harmful to plant growth especially at high concentrations (El-Fadel et al., 1997a). Other factors including the toxicity of landfill gas, low nutrient levels, low moisture and poor soil structure have also been reported to affect plant growth at landfill sites (Flower et al., 1981; Gilman et al., 1982).

- ***Groundwater Pollution***

Leachate production can cause significant pollution to the surrounding soil and to ground or surface water (Tatsi & Zouboulis, 2002). In uncontrolled landfills, leachate accumulates at the bottom of the landfill and percolates through the soil where it can reach the aquifer lines and contaminate the

underground water (Mor et al., 2006). The impact of landfill leachate on surface and ground water has been widely reported in the literature (Albaiges et al., 1986; Assmuth & Strandberg, 1993; Christensen et al., 1998; Kjeldsen, 1993; MacFarlane et al., 1983; Mor et al., 2006; Rapti-Caputo & Vaccaro, 2006).

- ***Malodours***

Landfills can act as a potential source of unpleasant odours (Sarkar et al., 2003). The presence of small concentrations of certain esters, hydrogen sulphide, alkyl benzenes, limonene, and organosulphur compounds in landfill gas is mainly responsible for the odour nuisance (Dincer et al., 2006; El-Fadel et al., 1997a). This nuisance can vary from relatively sweet to bitter or acidic, depending on the concentration of odorous compounds in the gas. The distance at which odours can travel away from landfills is also important and depends on the wind patterns and on weather conditions in general (Saral et al., 2009).

- ***Air Pollution and Health Hazards***

Although the amount of $VOCs$ in landfill gas is usually below 1% of the total emissions, their effects on the environment are significant (Dincer et al., 2006; Saral et al., 2009). $VOCs$ are considered to be among the most hazardous air pollutants in the atmosphere, and hence, can affect the ambient air quality especially in areas close to landfills (Brosseau & Heitz, 1994; El-Fadel et al., 1997a). VOC emissions can contribute to stratospheric ozone depletion, but they can also cause several health hazards such as toxicity and carcinogenicity (Brosseau & Heitz, 1994; Dincer et al., 2006). For instance, when $VOCs$ are combined with nitrogen oxides (NO_x) in the atmosphere, they can cause lung irritation due to ozone production. It should be highlighted that exposure to $VOCs$ can contribute to the risk of congenital anomalies and low birth weight (Elliott et al., 2001).

2.2.1 European Union legislation on waste

In Europe, legislation has been established to promote sustainable development in the field of waste management (EUROPA, 2009). As a result, the European Union (EU) has adopted a number of Directives to harmonise waste management and disposal policies throughout Europe and to ensure environmental protection.

To improve the sustainability of waste management, EU member states have agreed that the reduction of biodegradable waste going to landfill is indispensable in order to prevent and reduce the damage caused to the environment. This agreement was established by the EU landfill Directive which has set out the requirements for all EU member states to reduce the amount of biodegradable waste sent to landfill (European Commission, 2011b).

- ***The EU Landfill Directive (1999/31/EC)***

The European Directive 1999/31/EC on the Landfill of waste provides measures, procedures and guidance to prevent or reduce the negative effects on the environment from the landfilling of waste (European Council, 1999). It sets the minimum standards for the location, design, construction and operation of landfills and controls the nature of waste accepted to landfills. It defines the different categories of waste, and applies to all landfills. It also improves the environmental standards of landfills by requiring greater use of landfill gas collection and energy recovery from it, as a way to minimise GHG emissions (European Commission, 2011b).

This Directive also sets the targets for member states to reduce the amount of biodegradable municipal waste going to landfill to 75% by 2006, 50% by 2009 and 35% by 2016, based on the production levels of 1995 (European Council, 1999). EU member states that have been heavily reliant on landfill have an additional four years to comply with these targets, which means that they have to reduce the amount of their biodegradable waste going to landfill to 75% by 2010, to 50% by 2013 and to 35% by 2020 (European Council, 1999).

The targets set in this Directive do not apply to the spreading of sludge from sewage treatment works and dredging operations, inert waste for redevelopment or restoration, unpolluted soil or non-hazardous inert waste resulting from prospecting and extraction, treatment and storage of mineral sources as well as material from the operation of quarries, non-hazardous dredging sludge alongside small waterways from which they have been dredged and non-hazardous sludge in surface water, including the bed and its subsoil, but these wastes must be treated before being disposed of to landfill (European Council, 1999). Generally, waste must follow the criteria laid down in Annex II of the Directive for its acceptance in landfills, in order to avoid the risks of pollution (European Commission, 2011b; European Council, 1999).

Wastes that cannot be accepted in a landfill include liquid waste, flammable, explosive or oxidising waste, hospital and other clinical waste that is infectious, used tyres (with some exceptions) and any other type of waste which does not meet the acceptance criteria laid down in Annex II of the Directive (European Commission, 2011b; European Council, 1999).

Finally, EU member states must ensure that existing landfills operate based on the provisions set by the Directive and must report to the Commission every three years on the implementation of the Directive (European Commission, 2011b).

After the implementation of the EU landfill Directive, the Waste Framework Directive came into force requiring member states to manage waste without endangering human health and harming the environment.

- ***Waste Framework Directive (2006/12/EC) and the revised Waste Framework Directive (2008/98/EC)***

The Waste Framework Directive (WFD) requires member states to establish a waste management system to protect human health and the environment against harmful effects caused by the collection, transport, treatment, storage and tipping of waste (European Parliament and Council, 2006). To achieve a high level of environmental protection, member states are required to take appropriate measures to encourage the prevention or reduction of waste production and its harmfulness, to

encourage the recovery of waste by means of recycling, re-use or reclamation or any other process to extract secondary raw materials from the waste or to use the waste as a source of energy, and to ensure the disposal of waste without endangering human health and the environment (European Parliament and Council, 2006).

In November 2008, the revised WFD (Directive 2008/98/EC) (rWFD) was adopted with amendments in some of the articles of the previous form of the Directive. It draws together many existing concepts and policies, but it also incorporates new ones.

The rWFD outlines five steps for dealing with waste, ranked according to their environmental impact, forming the "waste hierarchy". In this waste hierarchy, prevention is at the top of the priority order, followed by re-use, recycling, other recovery and disposal, in descending order according to their increasing environmental impact with the last being the most harmful (Figure 2-1) (European Commission, 2011a; European Parliament and Council, 2008).



Figure 2-1 The waste hierarchy (adopted from European Commission)

The rWFD sets the basic concepts and definitions related to waste management. It explains when waste ceases to be waste and becomes a secondary raw material (end-of-waste criteria), and how to distinguish between waste and by-products. It includes the "polluter pays principle" and the "extended

producer responsibility" (European Commission, 2011a). It incorporates requirements for the management of hazardous waste and waste oils and sets out two new targets for increasing recycling and recovery rates for both household and construction and demolition waste (European Commission, 2011a). These targets aim to achieve by 2020:

- 50% of re-use and recycling of certain waste materials collected from households and other sources similar to households
- 70% for re-use, recycling and other recovery of collected construction and demolition waste

In addition, it requires member states to set up a separate collection of paper, metal, plastic and glass from the household waste stream by 2015 (letsrecycle.com, 2012).

Overall, it requires member states to adopt waste management plans and waste prevention programmes (European Commission, 2011a).

2.3 The science of anaerobic treatment – An overview

Anaerobic decomposition occurs naturally in the environment but, can also be mechanically employed for the stabilization of waste and wastewater (CADDET, 1998; Gujer & Zehnder, 1983; McCarty, 1964; Parkin & Owen, 1986; Toerien & Hatting, 1969). Is a biological process in which organic matter is converted into CH_4 and CO_2, through a series of microbiological reactions, in the absence of O_2 (Gujer & Zehnder, 1983; Toerien & Hatting, 1969).

- ***The development of anaerobic treatment***

The existence of CH_4 and CO_2 gases has been known since antiquity, but in the 17th century Jan Baptita Van Helmont was the first person to determine that flammable gas could evolve from the decaying of organic matter (Lusk, 1998). Nevertheless, the natural production of CH_4 was first discovered by Alessandro Volta in 1778, who made the observation that the gas collected from the stirred up sediment in Lake Maggiore, Italy was flammable (Dalton, 2005; McCarty, 2001). Dalton called the gas he collected 'carburetted hydrogen' and suggested that it consisted of an atom of carbon and two atoms of hydrogen (Dalton, 2005). It was not until 1804, that Dalton gave the correct formula of CH_4 (Neves, 2009). In 1808, Sir Humphry Davy determined that CH_4 was present in the gases produced during the anaerobic decomposition of cattle manure (Dalton, 2005).

It was only in the middle of the 19th century that anaerobic treatment started to be seriously developed. Its first application dated back in 1859 at a leper colony in Bombay, India, where it was used for waste disposal, whereas in 1860 it made its appearance in Europe and particularly in France (Lusk, 1998). There, anaerobic treatment was applied for the handling of domestic wastewater by the development of a simple air-tight chamber called "Mouras' Automatic Scavenger" (McCarty, 2001). In 1895, anaerobic treatment reached England when a similar anaerobic system referred to as a "septic tank" was constructed in the city of Exeter to test its efficiency to treat domestic wastewater (Lusk, 1998; McCarty, 2001). Because of its success, the city of Exeter approved the treatment of the entire city's wastewater by this system in 1897 (McCarty, 2001).

Although the wastewater was sufficiently treated, the indigestible material that remained after the process, also known as sludge, was considered a serious problem due to its offensive nature and unpleasant odour. At the beginning of the 20th century, Karl Imhoff developed an advanced tank which combined sedimentation and digestion in a single compartment, allowing sludge to become inoffensive and odourless (McCarty, 2001). The Imhoff tank became very popular and by the end of 1914, about 75 cities and many institutions in the United States were licensed to use this tank (McCarty, 2001). It was after that, that anaerobic treatment began to attract more interest for the management of settled sludge. Several methods were tried for separate treatment of sludge, and this was successfully achieved in 1927 when the Ruhryerband at Essen-Rellinghausen installed the first sludge-heating apparatus in a separate tank (McCarty, 2001). The efficiency of the treatment exceeded that of Imhoff tanks and separate processes were subsequently applied for the treatment of wastewater and sludge (McCarty, 2001).

The value of CH_4 produced during the anaerobic treatment of wastewater and sludge was first recognised in Exeter in 1885, where it was used for the heating and lighting at the disposal works (McCarty, 2001). Over the years, its value became even more recognised as it was used for other purposes, such as to generate power for a biological treatment plant and for cooling water from the motors for heating the digestion tanks (Babel et al., 2009).

Nowadays, anaerobic treatment is widely used not only for the stabilisation of sludge produced during the treatment of wastewater, but also for the treatment of the wastewater itself (Gunaseelan, 1997; Tchobanoglous & Burton, 1991). In addition, the need for renewable energy generation has pushed anaerobic treatment into the market for the management of other types of organic wastes, such as biomass and bio-waste. The term biomass includes fruit and vegetable waste, garden waste, crops, wood, weeds, marine and fresh water biomass and any other waste that is formed by photosynthetic capture of solar energy and stored as chemical energy (Gunaseelan, 1997). Bio-waste is waste of biological origin, such as animal manures and slurries, and sewage sludge from biomass based industries (Monnet, 2003).

- ***The reaction***

Waste and wastewater composition is often of a complex nature, which is difficult to describe in detail. An easy way to characterise their composition is by the use of two parameters, namely the chemical oxygen demand (COD) and the volatile solids (VS) content. COD is usually used to characterise wastewaters, while VS is used for solid wastes (Angelidaki & Sanders, 2004).

When waste and wastewater is degraded anaerobically, the end product is carbon in its most oxidized (CO_2) and most reduced form (CH_4), i.e. an electron transfer between carbon atoms takes place (Angelidaki & Sanders, 2004).

Therefore, based on a known substrate composition schematically represented as $C_nH_aO_bN_c$ and the assumption that the substrate is completely converted into biogas when anaerobically digested, the CH_4 and CO_2 yields can be calculated using the following reaction, first developed by Buswell and co-workers and then modified by others (Gujer & Zehnder, 1983; Parkin & Owen, 1986):

$$\boldsymbol{C_nH_aO_bN_c + \left[n - \left(\frac{a}{4}\right) - \left(\frac{b}{2}\right) + \left(\frac{3c}{4}\right)\right] H_2O \rightarrow \left[\left(\frac{n}{2}\right) + \left(\frac{a}{8}\right) - \left(\frac{b}{4}\right) - \left(\frac{3c}{8}\right)\right] CH_4 + \left[\left(\frac{n}{2}\right) - \left(\frac{a}{8}\right) - \left(\frac{b}{4}\right) + \left(\frac{3c}{8}\right)\right] CO_2 + cNH_3} \quad \textbf{Equation 2-1}$$

The ratio between CH_4 and CO_2 depends on the oxidation state of the carbon present in the organic material. As such, the more reduced the organic carbon content is, the more CH_4 is produced (Gujer & Zehnder, 1983). Estimating CO_2 is more complex as some of it remains dissolved in the liquid phase or is converted to bicarbonate as a function of the ammonia concentration (Gujer & Zehnder, 1983).

As a consequence, it can be suggested that the composition of the biogas depends on the mean oxidation state of carbon in the organic matter, the saturation of CO_2 in the liquid phase and the nitrogen content of the organic material degraded.

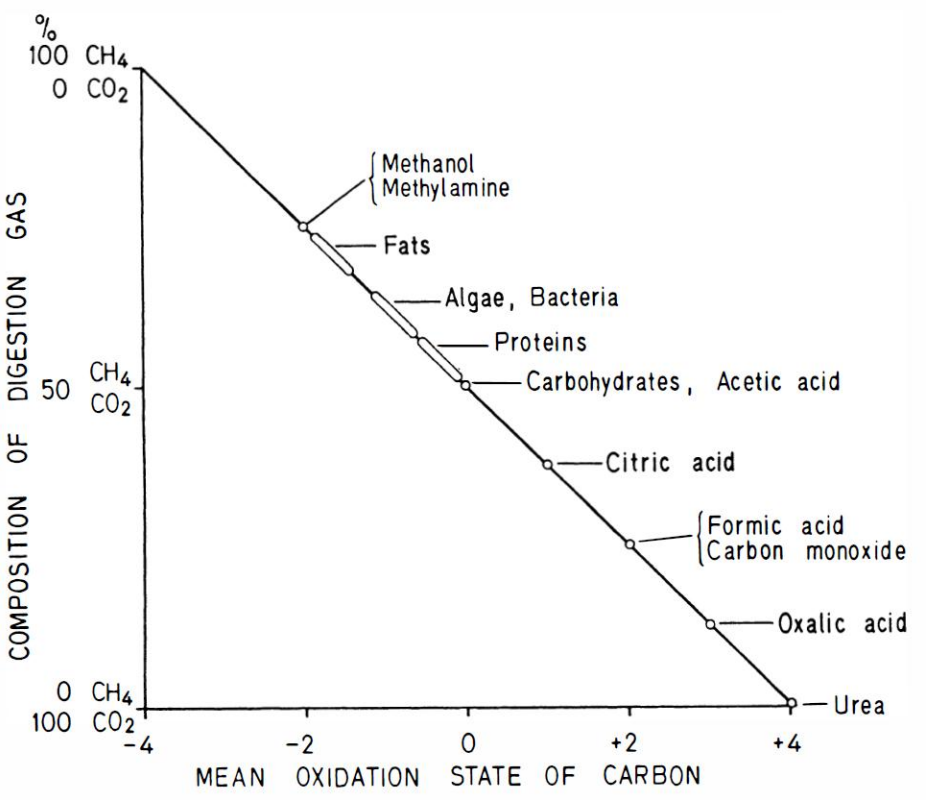


Figure 2-2 Composition of biogas based on the mean oxidation state of carbon of different components of a substrate (adopted from Gujer and Zehnder, 1983)

From the major groups of organic content, fats have the higher CH_4 potential with a mean oxidation state of carbon of -2, followed by proteins and carbohydrates (Figure 2-2). Carbohydrates are converted into equal amounts of CH_4 and CO_2, since they are characterised by a mean oxidation state of carbon of 0. Urea has a CH_4 potential of zero due to a mean oxidation state of +4.

- ***Advantages and disadvantages of anaerobic treatment***

Anaerobic treatment is a process that has many advantages to offer compared to aerobic processes and other waste management options with regard to wastewater and organic wastes treatment, respectively (Table 2-1). But like every other process, some potential disadvantages may also exist (Table 2-1) (Lettinga, 1995; McCarty, 1964; Seghezzo et al., 1998; Tchobanoglous & Burton, 2003).

Table 2-1 Advantages and disadvantages of anaerobic treatment

Advantages	Disadvantages
Biogas production	Long start-up time
Less sludge production	Low pathogen and nutrient removal
High degree of sludge stabilisation	Sensitivity to possible toxic compounds
Low nutrient requirements	Sensitivity to changes in temperature
High loading rates	Potential of malodours
High efficiency	Potential corrosiveness of the biogas
Microorganisms preservation	Potential alkalinity requirements
Degradation of toxic compounds	Necessity of post-treatment
Flexibility	

These are analysed as follows:

Advantages

- *Biogas production*. During the anaerobic treatment the organic matter is degraded into CH_4 and CO_2 and they form biogas. The produced biogas can be used for energy generation (Lettinga, 1995; Seghezzo et al., 1998).

- *Less sludge production*. The growth rate of the anaerobic bacteria is slow and only a small portion of the waste is converted into new cells, whereas most of it is turned into CH_4 (McCarty, 1964; Seghezzo et al., 1998). In fact, 50% of organic carbon is converted into biomass under aerobic conditions, whereas only 5% is converted into biomass under anaerobic conditions (Bitton, 2005).

- *High degree of sludge stabilisation*. Waste stabilisation is succeeded by the conversion of a high proportion of the waste into CH_4 (McCarty, 1964). The well stabilised sludge has also good dewatering characteristics (Lettinga, 1995; Seghezzo et al., 1998).

- *Low nutrient requirements*. Because of the slow growth rate of anaerobic bacteria and their smaller population number, fewer nutrients are required (Tchobanoglous & Burton, 2003).

- *High loading rates*. Anaerobic digesters can accommodate very high loading rates leading to another advantage that of the smaller space requirements (Lettinga, 1995; Seghezzo et al., 1998).

- *High efficiency*. Good removal efficiency can be achieved even at high loading rates (Seghezzo et al., 1998).

- *Microorganisms' preservation***.** The anaerobic organisms can be preserved unfed for long periods of time (exceeding one year) without a significant reduction of their activity (Bitton, 2005; Lettinga, 1995).

- *Degradation of toxic compounds*. Xenobiotic compounds such as chlorinated aliphatic hydrocarbons and recalcitrant natural compounds such as lignin can be degraded (Tchobanoglous & Burton, 2003).

- *Flexibility*. It can be applied at any place and scale (Lettinga, 1995; Seghezzo et al., 1998).

Disadvantages

- *Long start-up time*. Because of the low growth rate of methanogenic bacteria, long start-up times are usually required. However, the use of readily available organic material can speed up the process (Bitton, 2005; McCarty, 1964; Seghezzo et al., 1998).

- *Low pathogen and nutrient removal.* Pathogens are only partially removed during anaerobic treatment, and the biological removal of nitrogen and phosphorus is less likely (Seghezzo et al., 1998; Tchobanoglous & Burton, 2003).

- *Sensitivity to possible toxic compounds.* Methanogenic bacteria are very sensitive to toxic compounds and the presence of high concentrations of them, can limit the process (Tchobanoglous & Burton, 2003).

- *Sensitivity to changes in temperature*. Fluctuations in temperature can affect the performance of anaerobic treatment, mostly due to the slow growth rate of methanogenic bacteria that are severely affected by these changes (Bitton, 2005; Tchobanoglous & Burton, 2003).

- *Potential of malodours.* H_2S is produced during the anaerobic process in quantities which can be higher when sulphate (SO_4^{2-}) concentration in the influent is high (Seghezzo et al., 1998).

- *Potential corrosiveness of the biogas.* The presence of H_2S in the biogas can be a cause of damage to the digesters due to its corrosive potential (Tchobanoglous & Burton, 2003).

- *Alkalinity requirements*. In cases where alkalinity is not sufficient in the digester or cannot be produced by the degradation of proteins, it must be added in the digester in order to preserve a stable performance. This can incur a significant cost (Tchobanoglous & Burton, 2003).

- *Necessity of post-treatment*. Post-treatment of the anaerobic influents may be required to meet discharge standards and quality requirements (Seghezzo et al., 1998).

Many of the potential disadvantages of anaerobic treatment are attributed to the lack of sufficient knowledge of the process fundamentals and can be avoided or managed with proper waste characterisation and process design. When the process is clearly understood, an efficient performance can be achieved and the potential disadvantages will no longer be valid (McCarty, 1964; Parkin & Owen, 1986).

2.3.1 Microbiology of anaerobic treatment

Due to the synergistic interactions between a consortium of microorganisms that exist in an anaerobic environment, complex high-molecular-weight organic compounds are converted into CH_4 (Bitton, 2005; Rittman & McCarty, 2001). Although some fungi and protozoa may be found in anaerobic environments, bacteria are undoubtedly the dominant microorganisms (Bitton, 2005).

Bacteria are complete living entities that consist of only one cell, hence single cellular. They are able to grow, reproduce, organise and restrict what crosses their boundaries by taking up necessary elements, electrons and energy in a process called metabolism (Rittman & McCarty, 2001). They have the ability to transform a variety of inorganic and organic pollutants into harmless minerals that can be recycled back into the environment. Bacteria can also oxidise many organic chemicals synthesised naturally and industrially. One of their important characteristics is the wide variety of energy sources they can use for growth and maintenance such as light and thus called phototrophs, and chemical reactions and so called chemotrophs. Chemotrophs can be either autotrophic or heterotrophic, depending on whether they use inorganic or organic carbon for their cell synthesis, respectively (Rittman & McCarty, 2001).

Another characteristic that differentiates bacteria species is their response to O_2. Accordingly, bacteria are divided into three groups, the strict aerobes, facultative anaerobes, and anaerobes (Gerardi, 2003). Strict aerobes are active and degrade substrate only in the presence of O_2, where they perform a significant role in the degradation of wastes (Table 2-2). However, strict aerobes die in an anaerobic digester in which O_2 is absent (Gerardi, 2003; Rittman & McCarty, 2001).

Facultative anaerobes are active in the presence or absence of O_2. If present, O_2 is used for enzymatic activity and the degradation of wastes. If O_2 is absent another molecule, for example, nitrate ion (NO_3^-), is used to degrade wastes such as methanol (CH_3OH) (Equation 2-2).When NO_3^- are used, de-nitrification occurs and di-nitrogen gas (N_2) is produced.

$$\mathbf{6NO_3^- + 5CH_3OH \rightarrow 2N_2 + 5CO_2 + 7H_2O + 6OH^-} \qquad \textbf{Equation 2-2}$$

Facultative bacteria are found in relatively large numbers in anaerobic processes. They produce a variety of acids and alcohols, CO_2, and hydrogen (H_2) from the degradation of complex organic compounds (Table 2-2) (Gerardi, 2003).

Table 2-2 Classification of bacteria species according to their response in oxygen and their role

Bacteria	Response to oxygen	Role
Strict aerobes	Active only in the presence of oxygen	Degrade soluble organic compounds Oxidise nitrites (NO_2^-) to nitrates (NO_3^-) Oxidise ammonium ion (NH_4^+) to NO_2^-
Facultative anaerobes	Active in the presence or absence of oxygen	Degrade soluble organic compounds
Anaerobes	Inactive in the presence of oxygen	
a. Aerotolerant anaerobes	Oxygen tolerant	Reduce sulphate (SO_4^{2-}) to hydrogen sulphide (H_2S)
b. Strict anaerobes	Oxygen intolerant	Produce methane (CH_4)

Anaerobes are inactive in the presence of O_2 and may be divided into two subgroups: oxygen-tolerant species also called aerotolerant anaerobes, (Rittman & McCarty, 2001) and oxygen-intolerant species also called strict (or obligate) anaerobes (Table 2-2). Aerotolerant anaerobes can survive and grow in the presence of O_2, but cannot use it for enzymatic activity or any degradation purposes (Rittman & McCarty, 2001). It has also been reported that in the presence of O_2 they cannot perform normal cellular activities, such as the degradation of substrate (Gerardi, 2003). Aerotolerant anaerobes are strong acid producers and can also reduce SO_4^{2-} to H_2S (Equation 2-3) (Gerardi, 2003).

$$\boldsymbol{CH_3COOH + SO_4^{2-} \rightarrow 2CO_2 + 2H_2O + H_2S} \qquad \textbf{Equation 2-3}$$

Strict anaerobes, including methane-forming bacteria, are O_2 intolerant, meaning that in the presence of O_2 they simply cannot exist (Gerardi, 2003; Rittman & McCarty, 2001).

The various classifications of bacteria show that environmental conditions are the determining factor of their survival. Therefore, changes in these conditions can result in changes in the dominant bacteria species that control the anaerobic process (Gerardi, 2003; McCarty, 2001).

- ***Anaerobic metabolism***

Anaerobic metabolism can be described as a multistep process, during which a large number of reactions can take place through the microbial activity of different groups of bacteria (Gujer & Zehnder, 1983; Lyberatos & Skiadas, 1999; Pavlostathis & Giraldo-Gomez, 1991). Efficient metabolism is a result of the synergistic relationship between the different groups of bacteria (Parkin & Owen, 1986).

Four major groups of bacteria are involved in the transformation of complex organic compounds into simple molecules such as CH_4 and CO_2 (Gavala et al., 2003b). These are the:

1. Hydrolytic bacteria
2. Fermentative acidogenic bacteria
3. Acetogenic bacteria
4. Methanogens

Accordingly, the anaerobic metabolism can be presented as a four step process, the four steps being hydrolysis, acidogenesis, acetogenesis and methanogenesis (Bitton, 2005). A schematic representation of the anaerobic metabolism can be as follows (Figure 2-3).

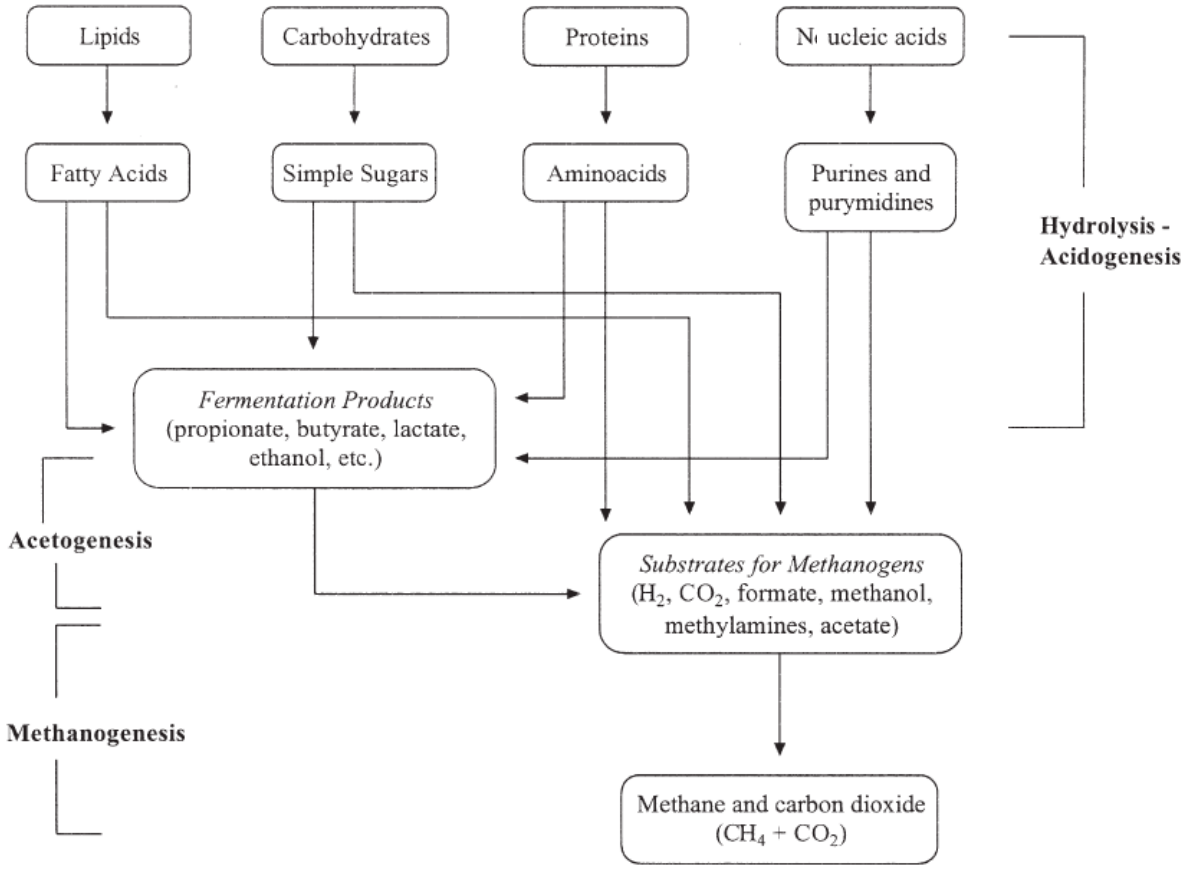


Figure 2-3 Anaerobic conversion of organic compounds into CH_4 and CO_2 through a four step process (adopted from Gavala et al., 2003a)

During hydrolysis, carbohydrates lipids and proteins are broken down into soluble monomer molecules such as sugars, fatty and amino acids, by the secretion of catalytic enzymes by hydrolytic bacteria (Ahring, 2003; Bitton, 2005; Gerardi, 2003). In the acidogenesis step, the corresponding bacteria convert the soluble monomer molecules into organic acids, alcohols, ketones, acetate, CO_2 and H_2 to make them available for the acetogenic bacteria (Figure 2-3). Acetogenic bacteria convert these into acetate (CH_3COOH), H_2 and CO_2 in the acetogenesis step (Bitton, 2005). Finally, methanogens produce CH_4 and CO_2 either by the use of CH_3COOH, or the reduction of CO_2 by H_2 in the final and most critical step of anaerobic treatment, the methanogenesis (Bitton, 2005; Hamzawi, 1998; Lusk, 1998; Mitchell & Gu, 2010; Neves, 2009; Sosnowski et al., 2003) (Figure 2-3).

1. ***Hydrolysis***

Complex organics, such as carbohydrates, proteins and lipids, must be hydrolysed into soluble forms that can be utilised by the bacteria for energy or nutrient sources. This is accomplished by the use of enzymes called hydrolases that are produced by the hydrolytic bacteria for this specific purpose (Bitton, 2005; Gavala et al., 2003a; Gerardi, 2003; Parkin & Owen, 1986). Two types of enzymes are involved in the degradation process, endoenzymes and exoenzymes. Endoenzymes degrade soluble substrate within the cell, whereas exoenzymes solubilise particulate and colloidal compounds that cannot be directly used by the bacteria, in order to make them permeable to the cell wall (Gerardi, 2003). According to the substrate they degrade, exoenzymes can be separated into cellulases, proteases and lipases (Table 2-3) (Bitton, 2005; Gerardi, 2003).

Table 2-3 Exoenzymes produced by the hydrolytic bacteria and their role

Exoenzymes	Role	Substrate	Product
Cellulase	Saccharolytic	Carbohydrates	Simple sugars
Protease	Proteolytic	Proteins	Amino acids
Lipase	Lipolytic	Lipids	Fatty acids

Although all bacteria produce endoenzymes, no single bacterium produces all the exoenzymes that are needed to degrade carbohydrates, proteins and lipids (Gerardi, 2003). Each exoenzyme is produced by different bacteria and hence the degradation of carbohydrates, lipids and proteins occurs in a parallel enzymatic activity (Gerardi, 2003). Therefore, for efficient hydrolysis, a large and active bacterial population, a concentrated organic substrate, and uniform mixing and temperature are required (Parkin & Owen, 1986).

Further, solubilisation is catalysed not only by the exoenzymes produced by the bacteria but also by physicochemical reactions (Gavala et al., 2003a). It should be noted that hydrolysis may depend on parameters such as the particle size, pH, and diffusion and adsorption of enzymes to particles (Bitton, 2005; Gavala et al., 2003a; Parkin & Owen, 1986).

2. Acidogenesis

During this step, dissolved organic matter such as simple sugars, amino acids and fatty acids are degraded into volatile fatty acids ($VFAs$), $alcohols$, $ketones$, CH_3COOH, CO_2 and H_2 by a heterogeneous microbial population (Bitton, 2005). The products produced vary according to the type of acidogenic microbes and the conditions that prevail in the digester. The dominant species in this step is bacteria while small populations of fungi and protozoa may also be present (Gavala et al., 2003a). Although both facultative and obligate anaerobic bacteria are present in acidogenesis, the obligate bacteria are the dominant group (Toerien & Hatting, 1969). Important factors that influence acidogenesis are: (1) interspecies H_2 transfer, (2) pH, (3) hydraulic retention time (HRT – the time organic substrates is treated), and (4) previous acclimation of the anaerobic culture.

3. Acetogenesis

This step determines the continuation of the anaerobic metabolism as it leads to the production of CH_3COOH which is the precursor of CH_4 production. Two different types of mechanisms can be distinguished during acetogenesis (Gavala et al., 2003a):

a) Acetogenic hydrogenation
b) Acetogenic dehydrogenation

Acetogenic hydrogenation is the production of CH_3COOH through the anaerobic reduction of CO_2 with H_2, of carbon monoxide (CO) and water (H_2O), and of methanol (CH_3OH) and CO_2, which is carried out by obligate hydrogen-consuming bacteria (Table 2-4) (Gavala et al., 2003a; Parkin & Owen, 1986).

Table 2-4 Reactions catalysed by hydrogen-consuming bacteria

Acetogenic hydrogenation
$2CO_2 + 4H_2 \rightarrow CH_3COOH + 2H_2O$
$4CO + 2H_2O \rightarrow CH_3COOH + 2CO_2$
$4CH_3OH + 2CO_2 \rightarrow 3CH_3COOH + 2H_2O$

Acetogenic dehydrogenation results in the production of CH_3COOH and H_2, from the fermentation of hexoses and organic long- and short-chain fatty acids. These reactions are catalysed by obligate hydrogen-producing bacteria (Table 2-5) (Gavala et al., 2003a).

Table 2-5 Reactions catalysed by hydrogen-producing bacteria

Acetogenic dehydrogenation
$C_6H_{12}O_6 \rightarrow 2CH_3COOH + 2CO + 2H_2$
$CH_3CH_2OH + H_2O \rightarrow CH_3COOH + 2H_2$
$CH_3CH_2COOH + 2H_2O \rightarrow CH_3COOH + CO_2 + 3H_2$
$CH_3CH_2CH_2COOH + 2H_2O \rightarrow 2CH_3COOH + 2H_2$

Obligate hydrogen-producing bacteria can survive only if their metabolic waste-H_2 is continuously removed. However, H_2 plays a key role in regulating CH_3COOH production and consumption (Parkin & Owen, 1986). Under relatively high H_2 concentrations, significant H_2 partial pressure occurs, which limits the activity of hydrogen-producing bacteria and, as a consequence, the methanogenesis step (Bitton, 2005; Gavala et al., 2003a; Gerardi, 2003; Parkin & Owen, 1986). In fact, when the H_2 partial pressure is above 0.0001 atmospheres, propionate and butyrate are likely to persist in the digester and the production of acetate will be minimized (Parkin & Owen, 1986; Polpraset, 2007). That would prevent CH_4 production and hence, the completion of the process. Therefore the symbiotic relationship of hydrogen-producing bacteria with hydrogen-consuming bacteria or methane-forming bacteria is detrimental for a successful process performance (Gerardi, 2003; Parkin & Owen, 1986).

Emphasis should be given to the fact that acetogenic bacteria reproduce very slowly, and any disturbances to their population can limit the anaerobic treatment process (Gerardi, 2003).

4. Methanogenesis

Methanogenesis is the final and most critical step of the anaerobic metabolism at which waste stabilization occurs through the conversion of the final products into CH_4. The bacteria that catalyse

this step are known by many names such as methanogens, methanogenic bacteria, methane-forming bacteria and methane-producing bacteria (Gerardi, 2003; McCarty, 1964). These bacteria belong in the domain Archaea-bacteria and are some of the oldest on earth (Gerardi, 2003). They are also the only microorganisms that can produce CH_4.

CH_4 generation can be attained through the ability of methanogens to obtain energy from the degradation of the final products of the anaerobic metabolism which are primarily CH_3COOH, H_2, CO_2, but also from one-carbon compounds, such as formic acid ($HCOOH$), methanol (CH_3OH), CO, tri-methylamine $[(CH_3)_3 - N]$, and dimethyl sulphide $[(CH_3)_2 - S]$ (Bitton, 2005; Gavala et al., 2003a; Gerardi, 2003). Nevertheless, no single species of methane-forming bacteria can utilize all substrates. As a result, the presence of a large number and diversity of methanogenic bacteria is required for a successful treatment (Gerardi, 2003). According to which substrate they use for obtaining their energy, methanogenic bacteria are divided into three categories. These are the a) hydrogenotrophic methanogens, b) acetotrophic methanogens and c) methylotrophic methanogens.

a) Hydrogenotrophic methanogens

These use H_2 as an electron donor to convert CO_2 -the electron acceptor-, into CH_4 by attaining at the same time a low H_2 partial pressure. By keeping a low H_2 pressure, they help acetogenic bacteria to survive which is very important for the oxidation of fatty acids into CH_3COOH (Bitton, 2005; Gavala et al., 2003a; Gerardi, 2003; Rittman & McCarty, 2001). It must be noted, that some hydrogenotrophic methanogens use CO to produce CH_4 (Table 2-6) (Gerardi, 2003).

b) Acetotrophic methanogens

These split CH_3COOH into CH_4 and CO_2 in a process known as acetoclastic cleavage (Table 2-6) (Gerardi, 2003; McCarty, 1964; Parkin & Owen, 1986). As an extension of this reaction, these methanogens are also called acetoclastic methanogens or acetate-splitting methanogens (Bitton, 2005; Parkin & Owen, 1986). The CO_2 produced from the acetoclastic cleavage is used by hydrogenotrophic

methanogens for CH_4 production (Bitton, 2005; Gerardi, 2003). The acetoclastic cleavage is responsible for the production of 65-70% of CH_4 (Gavala et al., 2003a).

Acetotrophic methanogens reproduce more slowly than hydrogenotrophic methanogens and like the acetogenic bacteria are also affected by the accumulation of H_2. Therefore the maintenance of low partial H_2 pressure is favourable for the activity of both acetogenic bacteria and acetoclastic methanogens (Gerardi, 2003).

c) Methylotrophic methanogens

This group of methanogens grow on substrates that contain the methyl group ($-CH_3$), and produce CH_4 directly from compounds that are composed by the ($-CH_3$) group. However, the production of CH_4 by this group of methanogens is very low if not negligible (Table 2-6) (Gerardi, 2003).

Table 2-6 Methanogenesis reactions of each group of methanogens

Hydrogenotrophic methanogenesis:	$CO_2 + 4H_2 \rightarrow CH_4 + 2H_2O$
	$2CO + 2H_2O \rightarrow CH_4 + 2CO_2$
Acetotrophic methanogenesis:	$CH_3COOH \rightarrow CH_4 + CO_2$
Methylotrophic methanogenesis:	$3CH_3OH + 6H \rightarrow 3CH_4 + 3H_2O$
	$4(CH_3)_3 - N + 6H_2O \rightarrow 9CH_4 + 3CO_2 + 4NH_3$

Methanogenesis is extremely sensitive to temperature and pH fluctuations, loading rate and the presence of toxic compounds. It can also be inhibited by the presence of other organic and inorganic compounds (Bitton, 2005; Parkin & Owen, 1986).

- ***Sulphate-reducing bacteria***

Another group of strictly anaerobic bacteria commonly present in anaerobic treatment is the sulphate-reducing bacteria (SRB). The main problem associated with SRB results from their higher affinity for CH_3COOH, which makes them interact competitively with methanogens under low CH_3COOH concentrations (Bitton, 2005). Sulphate reducing bacteria are able to use SO_4^{2-} as a terminal electron

acceptor for the oxidation of H_2 and a variety of organic compounds, resulting in the production of H_2S rather than CH_4 (Mitchell & Gu, 2010).

The problems related to the widespread occurrence of SRB in anaerobic treatment include:

1. Reduction in CH_4 yield
2. Inhibition of microbial groups by H_2S
3. Malodours and corrosion of pipes due to H_2S
4. Need to clean up H_2S from biogas

2.3.2 Kinetics of anaerobic treatment

Biological growth kinetics are based on two fundamental relationships: growth rate and substrate utilization (Pavlostathis & Giraldo-Gomez, 1991). Cell growth involves a respiratory conversion of the substrate into products and energy (catabolism), which is used for both the synthesis of new cells and the maintenance of old ones (anabolism) (Gavala et al., 2003a). The net effect of these processes is the metabolism of the substrate into new microorganisms and microbial products (Gavala et al., 2003a; Pavlostathis & Giraldo-Gomez, 1991). To measure this effect on the bacterial growth, a microorganism growth and decay rate are usually used (Pavlostathis & Giraldo-Gomez, 1991).

Various kinetic models have been developed over the years to describe the bacterial growth rate, based on a series of mathematical expressions, according to the following equation,

$$\boldsymbol{r_x = \mu(S, X) \cdot X} \qquad \textbf{Equation 2-4}$$

where, r_x is the bacterial growth rate, μ symbolises the specific bacterial growth rate and S, X symbolise the concentration of the biodegradable substrate and biomass, respectively (Gavala et al., 2003a). Contois (1959), Chen and Hashimoto (1978) and Grau et al., (1975) have developed models which predict that the effluent substrate concentration (S) is a function of the influent substrate

concentration (S_0) (Table 2-7). These models are an improved description of the Monod model in which, S is independent of the S_0 (Table 2-7) (Gavala et al., 2003a; Pavlostathis & Giraldo-Gomez, 1991).

Table 2-7 Kinetic expressions of the microbial growth rate as a function of the substrate (S) concentration

Model	Specific growth rate, μ
First-order	$\frac{k \cdot S}{S_0 - S} - b$
Monod	$\frac{\mu_{max} \cdot S}{K_S + S}$
Moser	$\mu_{max} \cdot \frac{S^n}{S^n + K_S}$
Contois	$\frac{u_m \cdot S}{B \cdot x + S} - b$
Grau et al.	$\frac{\mu_{max} \cdot S}{S_0}$
Chen and Hashimoto	$\frac{\mu_{max} \cdot S}{K \cdot S_0 + (1 - K) \cdot S}$

k is the maximum specific substrate utilisation rate, μ_{max} is the maximum specific growth rate, K_S is the saturation constant, *b* is the specific microorganism decay rate

Generally, in a series of reactions, it is expected that one step is much slower than the other steps (Pavlostathis & Giraldo-Gomez, 1991). This slow step is usually called the rate-limiting or rate determining step (Pavlostathis & Giraldo-Gomez, 1991). In anaerobic treatment, hydrolysis was found to be the rate-limiting step that normally influences the overall process (Pavlostathis & Giraldo-Gomez, 1991).

- ***Hydrolysis kinetics***

Hydrolysis of the organic compounds is usually described by a first-order kinetic model (Angelidaki & Sanders, 2004; Christ et al., 2000; Gavala et al., 2003a; Gujer & Zehnder, 1983; Pavlostathis & Giraldo-Gomez, 1991; Sanders et al., 2000). The following equation is commonly considered for the hydrolysis of biodegradable organic matter:

$$\frac{dX_{degr}}{dt} = -k_h \cdot X_{degr} \qquad \textbf{Equation 2-5}$$

In this equation, X_{degr} is the concentration of biodegradable organic matter (kg/m^3), t is time in days and k_h is the first order hydrolytic kinetic constant rate ($days^{-1}$).

This is an empirical equation and changes in the k_h value can be associated with changes on the experimental conditions and substrate characteristics. Even if the reactor conditions and substrate type are kept constant, different k_h can be found due to changes in the particle size distribution of the substrate (Sanders et al., 2000).

A wide range of k_h values have been reported in the literature for the hydrolysis of carbohydrates, proteins and lipids, assuming first-order hydrolysis (Gavala et al., 2003a; Pavlostathis & Giraldo-Gomez, 1991). It must be taken into account that the hydrolysis rate of these substrates depends on the origin and previous acclimation of the anaerobic culture (Gavala et al., 2003a). Digester conditions such as temperature and solids retention time (SRT) - that is, the average time the solids spend in the digester- are also of great importance for the hydrolysis rate of carbohydrates, lipids and proteins (Table 2-8) (Christ et al., 2000; Gujer & Zehnder, 1983; Pavlostathis & Giraldo-Gomez, 1991; Vavilin et al., 2008).

Table 2-8 Kinetic hydrolytic constant rates of the major organic groups

Substrate	k_h ($days^{-1}$)	Temperature (^oC)	SRT (days)	Reference
Carbohydrates	0.025-0.200	55	-	Christ et al., 2000
Carbohydrates	0.5-2	-	-	Garcia-Heras, 2003[1]
Proteins	0.01-0.10	35	5-60	O'Rourke, 1968[2,3]
Proteins	0.02-0.03	-	-	Gujer and Zehnder, 1983
Proteins	0.015-0.075	55	-	Christ et al., 2000
Proteins	0.25-0.28	-	-	Garcia-Heras, 2003[1]
Lipids	0.01-0.17	35	5-60	O'Rourke, 1968[2,3]

Lipids	0.08-1.70	-	-	Gujer and Zehnder, 1983
Lipids	0.005-0.010	55	-	Christ et al., 2000
Lipids	0.1-0.7	-	-	Garcia-Heras, 2003[1]

[1]Vavilin et al., 2008; [2]Pavlostathis and Giraldo-Gomez, 1991; [3]Gujer and Zehnder, 1983

It is generally accepted that carbohydrates can be effectively hydrolysed during the anaerobic fermentation (Gavala et al., 2003a; Pavlostathis & Giraldo-Gomez, 1991). However, protein hydrolysis has been reported to be slower than carbohydrate hydrolysis (Gavala et al., 2003a). With regards to protein hydrolysis, it has been reported that it depends on the solubility of each protein, the pH and the origin of the anaerobic culture (Gavala et al., 2003a). For lipids, the information that currently exists in the literature regarding their hydrolysis is vague (Table 2-8).

The hydrolysis potential of complex organic substrates has been examined by many researchers (Gavala et al., 2003a; Gujer & Zehnder, 1983; Pavlostathis & Giraldo-Gomez, 1991; Vavilin et al., 2008). Most of them concluded that hydrolysis remains the less defined step in the anaerobic treatment process. The k_h of different substrates reported in the literature are presented in the following table (Table 2-9).

Table 2-9 Kinetic hydrolytic constant rates of complex organic substrates

Substrate	k_h ($days^{-1}$)	Temperature (^oC)	SRT (days)	Reference
Mixture of primary and waste activated sludge	0.077	25	-	Gujer and Zehnder, 1983
	0.15	35	-	
Municipal solid waste	0.052-0.99	35-60	>15	Pfeffer, 1974[1,2]
	0.25-0.28	35-60	<15	
Primary sludge (domestic)	0.4-1.2	35	-	O'Rourke, 1968[2,4]
Primary sludge	0.99	35	-	Ristow et al., 2006
Raw domestic sludge	0.077-0.182	35	1.5	Eastman and Ferguson, 1981[3]
Secondary sludge	0.168-0.6	35	-	Ghosh, 1981[1,2,4]

Kitchen waste	0.34	35	-	Liebetrau et al., 2004[4]
Biowaste	0.12	35	-	Liebetrau et al., 2004[4]
Municipal solid waste	0.1	15	-	Bolzonella et al., 2005[4]
Food waste	0.55	37	-	Vavilin et al., 2004[4]
Household solid waste	0.1	37	-	Vavilin and Angelidaki, 2005

[1]Pavlostathis and Giraldo-Gomez, 1991; [2]Gavala et al., 2003a; [3]Gujer and Zehnder, 1983; [4]Vavilin et al., 2008

The k_h can take different values depending on the substrate, temperature and SRT. Additionally, the pH, particle size and mixing conditions can also play an important role to the hydrolytic potential of the substrates (Parkin & Owen, 1986). As a consequence, the first order kinetics appear not to be applicable in all circumstances, indicating that an in-depth understanding of the different processes involved is necessary to accurately describe hydrolysis.

- ***Acidogenesis kinetics***

Acidogenesis of simple carbohydrates such as glucose is mostly described by Monod kinetics, whereas limited data exist on the kinetics of acidogenesis of amino acids (Gavala et al., 2003a). However, existing data support the proposition that the acidogenesis of amino acids is faster than the hydrolysis of proteins, indicating that hydrolysis is the rate-limiting step of the anaerobic degradation of proteins (Pavlostathis & Giraldo-Gomez, 1991). It must be noted that the composition and origin of the treated substrate can affect the rate and extent of acidogenesis reactions (Pavlostathis & Giraldo-Gomez, 1991).

- ***Acetogenesis kinetics***

Acetogenesis usually refers to the anaerobic oxidation of long and short chain fatty acids (Gavala et al., 2003a). Numerous studies have been undertaken on the kinetics of long and short chain fatty acids, and particularly of butyrate and propionate. The oxidation reactions of these substrates have been described by Monod kinetics (Pavlostathis & Giraldo-Gomez, 1991).

- ***Methanogenesis kinetics***

The reduction of CO_2 using H_2, for CH_4 generation, is very important since it regulates the levels of H_2, thus controlling its partial pressure. Therefore the kinetics of H_2 utilisation has attracted a lot of interest (Pavlostathis & Giraldo-Gomez, 1991). Kinetic investigations on methanogenic bacteria utilizing acetate also exist (Gavala et al., 2003a). All sub-processes of methanogenesis have been described by Monod kinetics.

2.3.3 Factors controlling the anaerobic treatment

Within the anaerobic environment, various important parameters that govern bacterial growth can affect the performance of the anaerobic process. These are the temperature, pH, presence of toxic substances, solid and hydraulic retention time (SRT and HRT), organic loading rate (OLR) and mixing (bacteria-substrate contact) (Appels et al., 2008; Bitton, 2005; Parkin & Owen, 1986).

- ***Temperature***

In the anaerobic treatment processes, temperature controls the extent of the growth rate and metabolism of microorganisms and hence the population dynamics. This is due to the effect of temperature on the enzymatic activity or reactions in the digester, which increases with increases in temperature and vice versa. There are three common temperature ranges at which anaerobic treatment can be achieved (Lettinga, 1997; Lettinga et al., 2001). These are:

1. Psychrophilic, for temperatures lower than 20ºC
2. Mesophilic, for temperatures within 20–45ºC
3. Thermophilic, for temperatures in the range of 45–60ºC

Although, CH_4 production has been documented under psychrophilic temperatures, anaerobic treatment is usually carried out at an either mesophilic or thermophilic range of temperatures. The average temperature for mesophilic and thermophilic treatment is 35ºC and 55ºC, respectively (Bitton,

2005; Ward et al., 2008). Thermophilic temperatures have the advantages of higher loading rates, increased solubility of the organic compounds and destruction of pathogens, but they also have several drawbacks such as sensitivity to toxicants and generation of high concentrations of free ammonia (NH_3) that plays an inhibiting role for the microorganisms (Appels et al., 2008; Bitton, 2005). Nevertheless, there is no optimal temperature range for the anaerobic treatment, since there are studies on the anaerobic treatment under mesophilic and thermophilic conditions with conflicting results.

Although hydrolytic bacteria are not sensitive to temperature fluctuations, acetate- and methane-forming bacteria are affected by sharp and/or frequent fluctuations in temperature and as a result they can affect the overall treatment performance (Appels et al., 2008). For instance, a change of 10^{o}C in temperature can stop the activity of methane-forming bacteria and consequently CH_4 production (Gerardi, 2003).

Temperature has also a significant effect on the H_2 partial pressure which influences the kinetics of the anaerobic reactions. Reactions, such as the breakdown of propionate into CH_3COOH, CO_2 and H_2, become energetically more favourable at higher temperatures, whereas reactions, such as hydrogenotrophic methanogenesis, are more favoured at low temperatures (Appels et al., 2008).

- ***pH***

Maintenance of the pH levels is a prerequisite of an efficient anaerobic treatment process (Parkin & Owen, 1986). The desired pH range lies between 6.6 and 7.6. Values outside this range can be detrimental to process stability (Rittman & McCarty, 2001). This is because methanogens are extremely sensitive to pH changes, compared to the other groups of bacteria. In fact while acidogenic bacteria can function in a pH range between 4.0 and 8.5, methanogens can only function at a pH range of 6.7-7.4, with an optimum between 7 and 7.2 (Appels et al., 2008; Bitton, 2005).

Deviations from the optimum pH range are usually a result of the increased production and accumulation of acidic or basic products such as $VFAs$ and NH_3 respectively. $VFAs$ accumulation can lower the pH levels, whereas NH_3 accumulation can increase the pH. It is important that pH returns to

its proper range, because deviations far from its optimum range can lead to a system upset (Appels et al., 2008; Bitton, 2005; Parkin & Owen, 1986; Rittman & McCarty, 2001).

- ***Alkalinity***

The digester pH is buffered by the CO_2 concentration in the gas phase and also the bicarbonate ion (HCO_3^-) which is the major source of alkalinity in anaerobic treatment (Appels et al., 2008; Parkin & Owen, 1986; Rittman & McCarty, 2001). During anaerobic treatment, HCO_3^- is produced either by the breakdown of nitrogenous compounds, such as proteins, to NH_3 and its reaction with CO_2 to form ammonium bicarbonate (NH_4HCO_3), or the breakdown of existing compounds such as soaps and organic salts. Alkalinity can be expressed in mg/l of calcium carbonate ($CaCO_3$) (Appels et al., 2008).

- ***Nutrient requirements***

As with all biological systems, trace nutrients must be present to support the growth of microorganisms involved (Table 2-10) (Bitton, 2005; Parkin & Owen, 1986; Rittman & McCarty, 2001). The major inorganic nutrients required for growth are nitrogen and phosphorus. Nitrogen represents about 12% by weight of the cell, while phosphorus represents about 2%. Sulphur is another inorganic nutrient that is required for anaerobic microorganisms, especially methanogens, which require sulphur in the same order of magnitude as phosphorus (Mignone, 2005).

Another group of nutrients equally important as inorganic ones is metals. These are further divided into two other sub-groups, namely the light metal ions, also known as cations, and the heavy metals (Mignone, 2005; Parkin & Owen, 1986; Rittman & McCarty, 2001).

Traces of heavy metals are required for the activation of key enzymes for methanogenesis. Iron, cobalt and nickel are known requirements for key enzymes within methane-producing species and must always be present for effective anaerobic treatment. Lack of sufficient trace heavy metals may be a cause of failure of anaerobic treatment (Mignone, 2005; Parkin & Owen, 1986; Rittman & McCarty, 2001).

Cations such as sodium, potassium, calcium and magnesium are also needed in the anaerobic environment for the growth of microorganisms. However, these must be present in a correctly balanced quantity. If one of them is present in a higher proportion than the others, then the environment may become unbalanced. Such unbalance can be recovered by increasing the proportions of the other cations (Rittman & McCarty, 2001).

Table 2-10 Nutrient requirements for anaerobic treatment (adopted from Rittman and McCarty, 2001)

Nutrients	Desired excess concentration (mg/l)	Typical form for addition
Nitrogen (N^{3+})	50	NH_4, NH_4Cl, NH_4HCO_3
Phosphorus (P^{2+})	10	NaH_2PO_4
Sulphur (S^{2-})	5	$MgSO_4 \cdot 7H_2O$
Iron (Fe^{2+})	10	$FeCl_2 \cdot 4H_2O$
Cobalt (Co^{2+})	0.02	$CoCl_2 \cdot 2H_2O$
Nickel (Ni^{2+})	0.02	$NiCl_2 \cdot 6H_2O$
Zinc (Zn^{2+})	0.02	$ZnCl_2$
Copper (Cu^{2+})	0.02	$CuCl_2 \cdot 2H_2O$
Manganese (Mn^{2+})	0.02	$MnCl_2 \cdot 4H_2O$
Molybdenum (Mo^{5+})	0.05	$NaMoO_4 \cdot 2H_2O$
Selenium (Se^{5+})	0.08	Na_2SeO_3
Tungsten (W^{6+})	0.02	$NaWO_4 \cdot 2H_2O$
Boron (B^{6+})	0.02	H_3BO_3
Sodium (Na^{+})	100-200	$NaCl$, $NaHCO_3$
Potassium (K^{+})	200-400	KCl
Calcium (Ca^{2+})	100-200	$CaCl_2 \cdot 2H_2O$
Magnesium (Mg^{2+})	75-250	$MgCl_2$

- ***Toxicants***

A wide range of toxicants, including light metal ions, heavy metals, ammonia-nitrogen ($NH_3 - N$), sulphide (S^{2-}) and a wide range of organic compounds are responsible for toxicity caused during

anaerobic treatment (Bitton, 2005; Rittman & McCarty, 2001). The extent at which a substance is toxic to the anaerobic environment depends on a number of factors including solubility, concentration and acclimation (Mignone, 2005; Parkin & Owen, 1986). For a substance to be toxic, it must be present in its solubilised form. Only then it can penetrate through the cell membrane and affect the microorganisms (Mignone, 2005; Parkin & Owen, 1986; Rittman & McCarty, 2001). Also, the concentration of a substance is very important, as many substances are stimulatory at low concentrations, but inhibitory or toxic at high concentrations (Mignone, 2005; Rittman & McCarty, 2001). A toxic substance that is slowly increased in the anaerobic environment will probably not cause any toxicity, as a result of the acclimatization of microorganisms to it (Mignone, 2005).

- **Light metal ions**

A high light metal ion or cation concentration can lead to the dehydration of bacterial cells due to osmotic pressure (Chen et al., 2008). Common cations that can cause inhibition at high concentrations are Na^+, K^+, Ca^{2+} and Mg^{2+}. These are present in the reactor either through the breakdown of organic matter or added as pH adjustment chemicals (Grady, 1996). The concentrations at which cations become toxic to the anaerobic environment vary, with 3000mg/l for Mg^{2+}, 8000mg/l for Na^+ and Ca^{2+}, and 12000mg/l for K^+. At lower concentrations these cations can be classified either as moderately inhibitory or stimulatory. It must also be emphasized that the toxic effect of one cation can be either decreased by another cation due to antagonism, or increased by another cation due to synergism (Table 2-11) (Mignone, 2005; Rittman & McCarty, 2001). Antagonism is defined as the reduction of the toxic effect of one cation by the presence of another, whereas synergism is the increase of the toxic effect of one cation by the presence of another (Mignone, 2005).

Table 2-11 Antagonistic and synergistic effect of cations (adopted from Rittman and McCarty, 2001 and Mignone, 2005)

Cation	Antagonistic cation	Synergistic cation
Na^+	K^+	Mg^{2+}, Ca^{2+}
K^+	Na^+, Ca^{2+}, Mg^{2+}	None

Ca^{2+}	Na^{+}, K^{+}	Mg^{2+}
Mg^{2+}	Na^{+}, K^{+}	Ca^{2+}

- **Heavy metals**

Heavy metals are the most common cause of failure in the anaerobic treatment (Table 2-12) (Chen et al., 2008; Parkin & Owen, 1986). They are divided into:

a) those that are consistently found like cadmium (Cd^{2+}), chromium (Cr^{6+}), copper (Cu^{2+}), lead (Pb^{2+}), nickel (Ni^{2+}), and zinc (Zn^{2+})
b) those that are frequently found like arsenic (As^{3+}), iron (Fe^{3+}), manganese (Mn^{2+}), mercury (Hg^{2+}), and silver (Ag^{+}), and
c) those that are occasionally found like aluminium (Al^{3+}), cobalt (Co^{2+}), molybdenum (Mo^{6+}), selenium (Se^{2+}), and tin (Sn^{3+}).

Their toxic effect is attributed to their ability to disrupt enzyme function and structure, by the binding of metals with thiol and other groups on protein molecules or by replacing naturally occurring metals in enzyme prosthetic groups (Chen et al., 2008). The best way to control heavy metals toxicity is by precipitation with H_2S (Mignone, 2005; Rittman & McCarty, 2001). Heavy metal sulphides are insoluble and as such, cannot cause any toxicity to the anaerobic microorganisms (Mignone, 2005).

Table 2-12 Soluble heavy metal concentrations inhibitory to the anaerobic treatment (adopted from Mignone, 2005)

Heavy metal	Soluble concentration (mg/l)
As^{3+}	0.5-1.0
Cd^{2+}	0.01-0.02
Cr^{6+}	1.0-1.5
Cu^{2+}	0.5-1.0
Ni^{2+}	1.0-2.0
Zn^{2+}	0.5-1.0

- **Ammonia-nitrogen**

Ammonium ion (NH_4^+) and unionised ammonia (NH_3) are the two principal forms of inorganic $NH_3 - N$ present in the anaerobic system (Chen et al., 2008; Parkin & Owen, 1986; Rittman & McCarty, 2001). Both forms are in equilibrium with each other, and whether the concentration of NH_3 or NH_4^+ is more inhibitory depends on the system pH (Equation 2-6) (Mignone, 2005; Rittman & McCarty, 2001).

$$\boldsymbol{NH_4^+ \leftrightarrow NH_3 + H^+} \qquad \textbf{Equation 2-6}$$

When the system pH is lower than 7.2, equation equilibrium shifts towards the NH_4^+, whereas when the pH is greater than 7.2 equilibrium shifts toward the NH_3 (Mignone, 2005). This is because NH_3 combines with CO_2 and H_2O to form NH_4HCO_3, which increases the pH level (Rittman & McCarty, 2001). High pH level (~ 8) and total $NH_3 - N$ concentrations can be inhibitory to the system due to the NH_3 levels. The response of the anaerobic treatment system to such inhibition is to reduce the consumption rate of $VFAs$, which in turn lowers the pH and inhibition is reduced (Chen et al., 2008; Rittman & McCarty, 2001). On the other hand, if enough NH_3 is released, the NH_4^+ concentration builds up and reaches its inhibitory level, which cannot be relieved by a lower pH, and thus causes toxicity in the system. NH_3 is inhibitory at lower concentrations than NH_4^+ (Chen et al., 2008; Mignone, 2005; Rittman & McCarty, 2001). The levels at which NH_3 becomes inhibitory and even toxic are unclear as there is conflicting information in the literature with reported concentrations of 650mg/l, 1500mg/l, 2000 mg/l, 3000 mg/l and even 5000mg/l (Angelidaki & Sanders, 2004; Bitton, 2005; Gerardi, 2003; Hartmann & Ahring, 2005; Parkin & Owen, 1986).

- **Sulphide**

At high sulphate (SO_4^{2-}) concentrations, S^{2-} toxicity is a common problem. This is mainly because of the affinity of SO_4^{2-} to electrons. However, it is the soluble, unionised form (H_2S) that is toxic to the anaerobic environment (Rittman & McCarty, 2001). Because, H_2S diffuses through the cell membrane more rapidly than ionised species, S^{2-} toxicity is highly dependent on the pH. At a neutral pH, 20-50% of the dissolved S^{2-} is in the form of H_2S (Bitton, 2005). The levels at which H_2S becomes toxic are between 150 and 200mg/l (Bitton, 2005; Mignone, 2005; Rittman & McCarty, 2001).

- **Organic compounds**

The organic compounds that can be toxic to the anaerobic environment are many and varied. Some of them can serve as a food source to the anaerobic microorganisms at low concentrations, whereas at high concentrations can cause process failure (Rittman & McCarty, 2001). Chlorinated hydrocarbons, hydrocarbons, phenols, alcohols, aldehydes, ketones, and aromatics, are some of the organic compounds that have been periodically reported in the literature over their toxicity in anaerobic treatment processes (Table 2-13).

Table 2-13 Toxicity levels of selected organic compounds to the anaerobic environment (adopted from Parkin and Owen, 1986)

Organic compound	Toxicity level (mg/l)
Formaldehyde	50-200
Chloroform	0.5
Ethyl Benzene	200-1000
Ethylene Dichloride	5
Kerosene	500

- ***SRT***

Another important factor controlling anaerobic treatment performance is the average time during which the sludge or solids, has remained in the reactor (Tchobanoglous & Burton, 2003). The residence time of solids (biomass) determines the type of microorganisms that will prevail in the

system and the extent of the different reaction rates (Grady, 1996). A decrease in the SRT decreases the extent of the reactions, whereas an increase in the SRT leads to increases in the reactions. Each time biomass (or sludge) is withdrawn, a fraction of the bacterial population is removed thus implying that the cell growth must at least compensate for the cell removal to ensure a steady state and avoid process failure (Appels et al., 2008). Therefore retention time must be long enough to allow metabolism by anaerobic microorganisms in the reactors. Generally, for an efficient anaerobic treatment at 30^{o}C, SRT should be more than 15 days, while decreasing temperatures require a higher SRT (Tchobanoglous & Burton, 2003).

- ***HRT***

It is the average time the wastewater or organic substrate remains in the reactor for treatment, and thus hydraulic retention time (Gerardi, 2003; Tchobanoglous & Burton, 2003). The HRT can be defined by the reactor volume (V_R), m^3, and the volumetric flow rate of the organic substrate (F_W), m^3 per day, as follows:

$$\boldsymbol{HRT = \frac{V_R}{F_W}} \qquad \textbf{Equation 2-7}$$

The HRT must be long enough to allow anaerobic bacteria to complete their metabolism and proliferation. HRTs vary depending on the type of reactor used (Bitton, 2005). Usually short HRTs are insufficient for a stable treatment due to a washout of methanogenic bacteria, while long HRTs result in a high operating cost (Appels et al., 2008; Bitton, 2005).

- ***OLR***

The loading rate of the organic substrate is important to the stability of an anaerobic system, as it is the determining factor of the appropriate HRT. OLR is usually expressed in volatile solids - VS (m^3/d) (Gerardi, 2003; Parkin & Owen, 1986). Although, the OLR is established empirically, it has a sound basis (Rittman & McCarty, 2001). Given that hydrolysis is the first step towards stabilization,

and thus the rate-limiting step of the anaerobic treatment process, the OLR must be kept at a range that the system can take. If the OLR is too high then hydrolysis and ultimately CH_4 stabilisation are incomplete, leading to a process failure (Rittman & McCarty, 2001).

- ***Mixing***

The reasons for mixing the contents of reactors are to provide efficient utilisation of the entire reactor volume, to prevent stratification and temperature gradients, to disperse metabolic end products and any toxic materials contained in the sludge, and to maintain intimate contact between the bacteria and the substrate (Parkin & Owen, 1986). Adequate mixing provides for a uniform environment, one of the keys to successful treatment. The effects of inefficient mixing on process kinetics are manifested in the decrease in effective system volume, which reduces the SRT and pushes the system towards failure. Additional concerns associated with inefficient mixing are foaming and scum formation, and excessive solids deposition (Parkin & Owen, 1986).

3 AIM AND OBJECTIVES

3.1 Aim

The aim of this thesis was to address the challenge of diverting food waste from landfill and its implications to the water industry, focusing on two food waste management options, the use of food waste disposal units (FWDs) and the anaerobic co-digestion with sewage sludge.

3.2 Objectives

The main objectives of this thesis had as follows:

- Critical assessment of the existing data on the potential impact of the use of FWDs on the wastewater system and the surrounding environment.
- Evaluation of the impact of the use of FWDs to water industry operations in the UK and comparison with the benefits and related savings from the reduction in cost of waste collection and disposal to local authorities (LAs).
- Assessment of the potential anaerobic-aerobic treatment of the increased strength municipal wastewater ensuing from the use of FWDs, as an approach to relieve the pressures associated with increased energy consumption and cost.
- Evaluation of the benefits and constraints of food waste co-digestion with sewage sludge and identification of the major hurdle to its implementation in the UK.
- Based on the outcomes of the above objectives, development of a sustainability framework to aid decision-making in the field of food waste management, with regards to the use of FWDs and the anaerobic co-digestion with sewage sludge, using multi-criteria approach.

3.3 Thesis structure

The rationale of the research herein, accompanied by a brief overview on the research topic, is presented in Chapter 1. Chapter 2 introduces the waste and wastewater management authorities in the UK and describes their major responsibilities and challenges. The importance of diverting food waste from landfill is addressed through the presentation of the impacts of landfilling, and the legislation that European Commission has put into force as a way to control these impacts is subsequently listed. An extensive analysis on the science of anaerobic treatment comes next. The aim and objectives of this thesis and its contribution to the science are clearly defined in Chapter 3.

Chapter 4 analyses the distribution of food waste generation around the world and critically assesses the potential benefits and impacts of the use of FWDs to the wastewater treatment facilities and the environment in general. In Chapter 5, an assessment of the use of FWDs in the UK, using a specific area as a case study, is presented, in order to identify the implications of implementing such an option, to both LAs and the water industry.

The aerobic treatment of the increased-strength wastewater, as a result of the use of FWDs, is associated with increased energy requirements and sludge production which represent an additional cost to the water industry. Anaerobic processes for the treatment of wastewater are considered an effective alternative to aerobic processes due to less energy requirement, less sludge production and increased potential for renewable energy generation. However, the performance of this option can possibly be affected by decreases in temperature. In the UK, where the average temperature is low, it is not clear how this process can be a sustainable solution over the conventional aerobic process. Chapter 6 aims to examine this potential, and show whether the anaerobic-aerobic treatment of the increased strength wastewater can be more beneficial to the water industry than the conventional aerobic treatment, especially when FWDs are implemented as a food waste management option.

Sewage sludge production from the wastewater treatment processes is anaerobically digested, as a way to utilise its putrescible organic matter into a stable residue by converting it into CH_4. Anaerobic digestion is a well-established process in many countries around the world, not only for the

stabilisation of sewage sludge but also for the treatment of other types of waste. The digestion of two substrates together, known as co-digestion, has recently been implemented as waste management option in many European countries due to its potential for CH_4 generation, that can be used for energy generation. Food waste has recently been introduced as a co-substrate in sewage sludge digestion. The anaerobic co-digestion of food waste with sewage sludge has been successfully implemented in some European countries, with its use continuously increased. While, in the UK, anaerobic digestion is widely used for the stabilisation of sewage sludge, when it comes to its co-digestion with food waste its use is still in its infancy. Chapter 7 aims to present the potential of food waste as a co-substrate in sewage sludge digestion, and address the barriers to its implementation in the UK.

Finally, the discussion of this thesis in Chapter 8, aims to inform the interested authorities on the decision-making process that can be followed to compare the sustainability of the use of FWDs with the anaerobic co-digestion of food waste with sewage sludge, based on their area-specific characteristics and practices.

Conclusions on the main findings of this research are presented in Chapter 9. The remainder of this thesis includes the list of references and the appendices, where supplementary data associated with the development of the findings presented in the Chapters, are provided.

3.4 Significance of the work

The significance of the thesis rests on that it critically assesses two food waste management options of great potential and relevance to LAs and water companies in the UK and worldwide. Both of these options have at least the theoretical potential to help with regulatory targets, new emerging drivers and sustainability in general, but their lack of large-scale implementation demonstrates the need for research into their practical and technical viability. This thesis makes a significant contribution to this by identifying the impacts of each process on water industry operations, and assessing their magnitude in long-term implementation. Each Chapter constitutes an individual, unique part of this assessment with Chapter 8 assembling the parts into a whole in order to assess the sustainability of each food waste management option.

In Chapter 4, a wide range of data on the impacts of FWDs, which is an area where reliable data are lacking, was consolidated in order to assist LAs in determining the viability of FWDs as a food waste management option, and to inform the water industry about the potential implications of their use. The limitations of existing data regarding these potential implications was highlighted and critically assessed, and the importance of further research in this area was underlined.

In Chapter 5 a model was developed that can be used by both LAs and the water industry to identify the impacts of small- and large-scale use of FWDs in their local area. Results of this model can be used to draw conclusions on the implications of using FWDs, thereby helping to inform the relevant authorities. It also establishes the importance of introducing new policy measures at both a national and regional level as a means to manage these implications.

The increased strength of wastewater resulting from the use of FWDs will impact wastewater treatment operations as a result of the increased energy consumption and sludge handling requirements. Chapter 6 recommends the incorporation of anaerobic treatment processes into municipal wastewater treatment as a way to relieve the pressures associated with the high cost of energy consumption and sludge management, as well as to produce additional renewable energy. However, low temperatures are considered to limit the performance of anaerobic treatment processes.

Through the use of modelling, this Chapter seeks to address the challenges of performing this process at low temperatures, which currently acts as an obstacle to the implementation of the anaerobic treatment of municipal wastewater in climates like that of the UK. This Chapter also makes suggestions for further research in the field, as the anaerobic treatment process could be a viable alternative to conventional aerobic treatment, especially in the future when increasing population and urbanisation and stringent regulations on energy consumption and carbon emissions are likely to create major problems to the water industry.

While many studies have focused on the benefits of anaerobic co-digestion of food waste with sewage sludge, only few have addressed its constraints as a waste management option. Chapter 7 assesses the benefits of food waste co-digestion with sewage sludge, addresses its constraints, demonstrates the main barriers to its implementation in the UK and makes recommendations on how to move anaerobic co-digestion of food waste with sewage sludge a step forward. This Chapter underpins the strong argument for the potential of anaerobic co-digestion of food waste with sewage sludge not only in the UK, but, also in countries that face the same concerns, and provides important information that can be used by anaerobic digestion operators to improve the benefits gained from the combined digestion of food waste with sewage sludge and understand the potential constraints associated with the process.

Finally, a sustainability decision-making framework is proposed in Chapter 8 which can be used by both LAs and the water industry to compare the use of FWDs and the anaerobic co-digestion of food waste with sewage sludge based on area-specific characteristics and practices. This framework was developed on the basis of a multi-criteria approach which offers the flexibility for stakeholders to determine the sustainability of food waste management options based on their own interests and needs. The methodology developed in the context of this framework was derived by assembling the benefits and impacts of the use of FWDs and the anaerobic co-digestion of food waste with sewage sludge to LAs and the water industry, as identified in the previous Chapters.

Most of the Chapters of this thesis have been published as peer reviewed articles in top journals, reflecting the originality of the research work. All chapters have either been published or submitted for consideration for publication to various international journals.

Publications

Iacovidou, E., Ohandja, D. G., Gronow, J.& Voulvoulis, N. (in press) The household use of food waste disposal units as a waste management option: A Review. *Critical Reviews in Environmental Science and Technology*.

Iacovidou, E., Ohandja, D. G. & Voulvoulis, N. (in press) Food waste disposal units in UK households: The need for policy intervention. *Science of the Total Environment*.

Iacovidou, E., Ohandja, D. G. & Voulvoulis, N. (submitted) Food waste co-digestion with sewage sludge – Realising its potential in the UK. *Journal of Environmental Management*.

Iacovidou, E., Ohandja, D. G. & Voulvoulis, N. (to be submitted) The potential anaerobic-aerobic treatment of increased strength wastewater as a result of the use of food waste disposal units (FWDs).

Iacovidou, E., Ohandja, D. G. & Voulvoulis, N. (to be submitted) Multi-criteria approach for the sustainability comparison of the use of food waste disposal units (FWDs) with the anaerobic co-digestion of food waste with sewage sludge.

Conference proceedings

Ohandja, D. G., Iacovidou, E., Bajzelj, B., Gronow, J. & Voulvoulis, N. (2010) Co-digestion of organic wastes in the UK: where are we?. *Proceedings of the Third International Symposium on Energy from Biomass and Waste, Venice, Italy, 8-11 November 2010*.

Vlachopoulou, M., Iacovidou, E., Ohandja, D. G., Gronow, J., Butwell, A. & Voulvoulis, N. (2011) Anaerobic digestion of food waste: Using the model ADM1 to simulate its performance and assess its potential as a waste management option, *Proceedings of the Thirteenth International Waste Management and Landfill Symposium, Sardinia, Italy, 3-7 October 2011*.

4 HOUSEHOLD USE OF FOOD WASTE DISPOSAL UNITS AS A WASTE MANAGEMENT OPTION

4.1 Introduction

As a component of municipal solid waste (MSW), most food waste ends up in landfill (El-Fadel et al., 1997a; Lou & Nair, 2009). Landfill has been the most convenient and economic form of waste disposal in many countries around the world. However, as a result of increased environmental regulations (e.g. the EU Landfill Directive) that aim to reduce the amount of biodegradable municipal waste and thus food waste going to landfill, alternative ways to food waste management have been introduced. These include kerbside collection and recycling, centralised incineration, home and centralised composting (De Koning & Van der Graaf, 1996; Diggelman & Ham, 2003; Wang et al., 2001), anaerobic digestion with and without sewage sludge (Edelmann, 2000b; GarcIa et al., 2005; Hamzawi, 1998; Hamzawi et al., 1998; Kim et al., 2004; la Cour Jansen, 2004; Mata-Alvarez et al., 2000; Sosnowski et al., 2003) and the use of food waste disposal units (FWDs).

Invented in 1927, the FWD is an electric device placed under the sink, which grinds food waste at source and discharges it through a water outlet to the sewer and from there to the local wastewater treatment plant (WWTP) (Evans, 2007; Galil & Shpiner, 2001). Cold tap water is used while grinding in order to attain efficient and uniform grinding (Rosenwinkel & Wendler, 2001). FWDs are used in many countries around the world, but their implementation as a waste management option has never been fully or appropriately considered (CECED, 2003). Although there is evidence that the use of FWDs enables the diversion of a considerable fraction of organic waste from the solid waste stream, their use is also associated with concerns about water consumption, the sewerage system, wastewater characteristics, treatment capacity and subsequent changes in the quantity and quality of sewage sludge and biogas.

The purpose of this Chapter is to critically assess the existing data on the potential impacts of the use of FWDs on the wastewater system and the environment. It reviews MSW and food waste generation rates and the applicability of FWDs in different countries around the world. It also analyses the

environmental impacts of the use of FWDs on energy usage, water consumption, the sewerage system, wastewater characteristics, sludge and biogas production and GHG emissions.

4.2 Food waste generation

A wide range of geo-demographic factors, including location, lifestyle, cooking-habits and socio-economic conditions, can substantially affect the amount of MSW, and thus food waste, generated (Knipe, 2005). However, economic conditions can be a critical factor when assessing the municipal and food waste generation rates as they can act as an indicator of the well-being of the people in a country. Therefore, municipal and food waste generation was examined as a function of the economic status of different countries, for which details can be found in Appendix A (World Bank, 2009).

In order to examine the relation between MSW/food waste generation and economic status, a list of countries categorized based on their income was drawn up and shown in Table 4-1. As can be seen, MSW generation appears to increase from the low- to the high-income economies (Defra, 2009c; IPCC, 2006; Magrinho et al., 2006; Metin et al., 2003; Ngoc & Schnitzer, 2009; ONS, 2010; Sakai, 1996; Sharholy et al., 2008; US EPA, 2009). Based on average MSW generation, which was estimated at 0.52, 0.67, 0.83 and 1.49 kg/cap/day for the low-, lower-middle-, upper-middle- and high-income economies, respectively, the difference between low- and high-income economies appears to be significant, as it may reach approximately 1 kg/cap/day. Food waste generation (Table 4-1), seems to follow the same pattern as MSW generation. Average food waste generation was estimated at 0.26, 0.31, 0.34 and 0.43 kg/cap/day, for the low-, lower-middle-, upper-middle- and high-income countries, respectively, while the difference between low- and high-income categories was found at 0.17 kg/cap/day, an approximately 40% increase. It is evident that municipal and food waste generation increases significantly with increasing income (Figure 4-1).

Table 4-1 MSW and food waste generation rates in several countries around the world

Classification based on economy	MSW generation (kg/cap/day)	Food waste fraction (%)	Food waste generation (kg/cap/day)	Reference
		Low-income economies		
Bangladesh	0.49	40.3	0.20	IPCC, 2006
Cambodia	0.52	55	0.29	Ngoc and Schnitzer, 2009
Lao PDR[1]	0.55	46	0.25	Ngoc and Schnitzer, 2009

Myanmar	0.45	54	0.24	Ngoc and Schnitzer, 2009
Nepal	0.49	40.3	0.20	IPCC, 2006
Vietnam	0.61	60	0.37	Ngoc and Schnitzer, 2009
Lower-middle-income economies				
Bolivia	0.44	44.9	0.20	IPCC, 2006
China	0.74	26.2	0.19	IPCC, 2006
Ecuador	0.60	44.9	0.27	IPCC, 2006
Guatemala	0.60	43.8	0.26	IPCC, 2006
Honduras	0.41	43.8	0.18	IPCC, 2006
India	0.38	40-60	0.15 – 0.23	Sharholy et al., 2008
Indonesia	0.76	62	0.47	Ngoc and Schnitzer, 2009
Nicaragua	0.77	43.8	0.34	IPCC, 2006
Paraguay	1.21	44.9	0.54	IPCC, 2006
Philippines, The	0.52	41	0.21	Ngoc and Schnitzer, 2009
Sri Lanka	0.88	40.3	0.35	IPCC, 2006
Sudan	0.79	52	0.41	IPCC, 2006
Thailand	0.64	48	0.31	Ngoc and Schnitzer, 2009
Upper-middle-income economies				
Argentina	0.77	44.9	0.35	IPCC, 2006
Brazil	0.49	44.9	0.22	IPCC, 2006
Bulgaria	1.42	30.1	0.43	IPCC, 2006
Colombia	0.71	44.9	0.32	IPCC, 2006
Costa Rica	0.47	43.8	0.21	IPCC, 2006
Cuba	0.58	46.9	0.27	IPCC, 2006
Dominican Rep.	0.69	46.9	0.32	IPCC, 2006
Latvia	0.74	30.1	0.22	IPCC, 2006
Lithuania	0.85	30.1	0.26	IPCC, 2006
Malaysia	0.81	62	0.50	Ngoc and Schnitzer, 2009
Mexico	0.85	33.9	0.29	IPCC, 2006
Peru	0.55	44.9	0.25	IPCC, 2006
Poland	0.88	30.1	0.26	IPCC, 2006
Romania	0.99	30.1	0.30	IPCC, 2006
Russian Federation	0.93	30.1	0.28	IPCC, 2006
St. Lucia	1.51	46.9	0.71	IPCC, 2006
Turkey	0.95	50-55	0.48 – 0.52	Metin et al., 2003
Uruguay	0.71	44.9	0.32	IPCC, 2006
Venezuela, RB[2]	0.90	44.9	0.40	IPCC, 2006
High-income economies				
Australia	1.89	36	0.68	IPCC, 2006
Austria	1.59	24.2	0.38	IPCC, 2006

Bahamas, The	2.60	46.9	1.22	IPCC, 2006
Belgium	1.29	24.2	0.31	IPCC, 2006
Canada	1.34	33.9	0.45	IPCC, 2006
Cyprus	1.86	36.9	0.69	IPCC, 2006
Czech Republic	0.90	30.1	0.27	IPCC, 2006
Denmark	1.84	23.8	0.44	IPCC, 2006
Estonia	1.21	30.1	0.36	IPCC, 2006
Finland	1.37	23.8	0.33	IPCC, 2006
France	1.29	24.2	0.31	IPCC, 2006
Germany	1.67	24.2	0.40	IPCC, 2006
Greece	1.12	36.9	0.41	IPCC, 2006
Hungary	1.23	30.1	0.37	IPCC, 2006
Iceland	2.74	23.8	0.65	IPCC, 2006
Ireland	1.64	24.2	0.40	IPCC, 2006
Italy	1.37	36.9	0.51	IPCC, 2006
Japan	1.10	25.5	0.28	Sakai, 1996
Korea, Rep.	1.04	26.2	0.27	IPCC, 2006
Luxembourg	1.80	24.2	0.44	IPCC, 2006
Malta	1.32	36.9	0.49	IPCC, 2006
Netherlands	1.70	24.2	0.41	IPCC, 2006
New Zealand	1.34	36	0.48	IPCC, 2006
Norway	1.70	23.8	0.40	IPCC, 2006
Portugal	1.32	33-41.5	0.44 – 0.55	Magrinho et al., 2006
Singapore	1.10	44	0.48	IPCC, 2006
Slovak, Rep.	0.89	30.1	0.27	IPCC, 2006
Slovenia	1.40	30.1	0.42	IPCC, 2006
Spain	1.64	36.9	0.61	IPCC, 2006
Sweden	1.18	23.8	0.28	IPCC, 2006
Switzerland	1.10	24.2	0.27	IPCC, 2006
UK (England)	1.53[3]	17.8	0.27	Defra, 2009c; ONS, 2010
USA	2.04	12.7	0.26	US EPA, 2009

[1]People's Democratic Republic of; [2]Bolivarian Rupublic of; [3]Figure calculated based on the total municipal solid waste produced in England in 2006/07(Defra, 2009c) and population growth in the same region and year (ONS, 2010).

The correlation between MSW and food waste generation rates according to the income is very strong as the average per capita MSW generation increases significantly with increasing income (Figure 4-1).

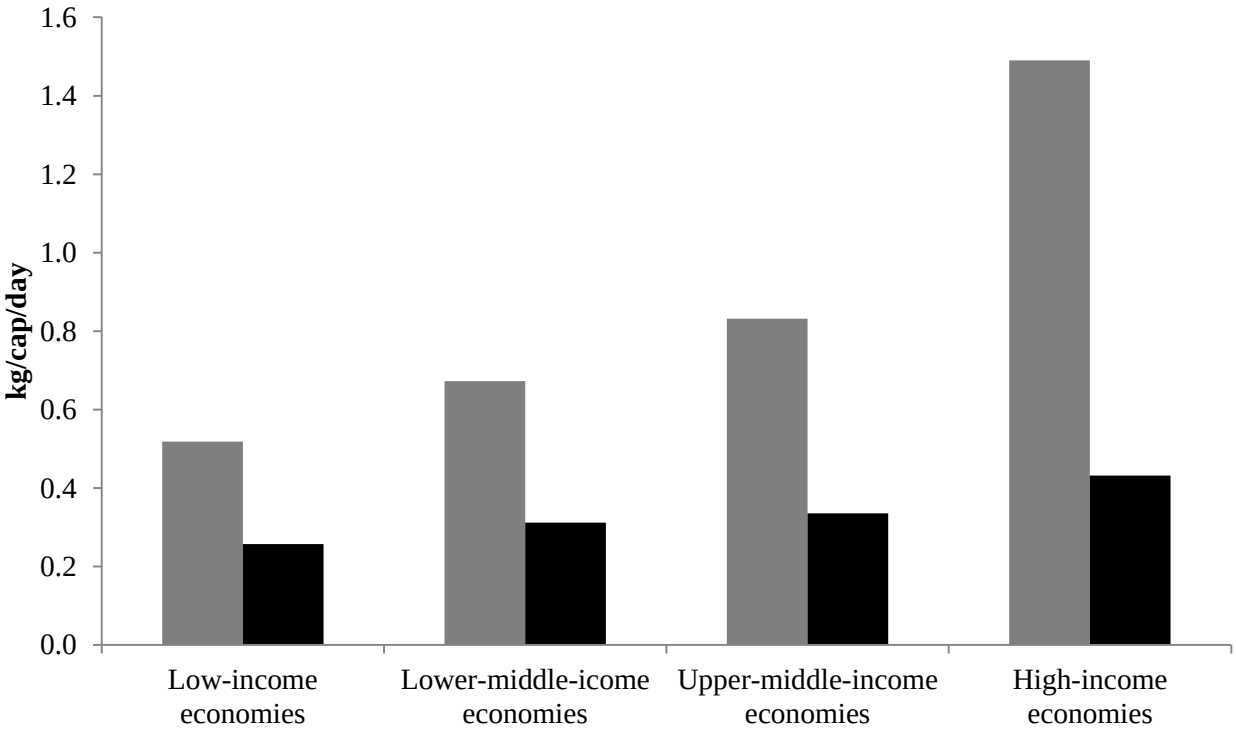


Figure 4-1 Average MSW (grey) and food waste (black) generation for each category

This increase can perhaps be attributed to the economic growth that characterizes high-income countries. The correlation between MSW and food waste generation can probably be attributed to the food waste fraction in the MSW stream. However, while the food waste fraction is higher in low-income (50%) than high-income (29%) countries (Table 4-1), the actual amount of food waste generated is higher in high-income countries (Figure 4-1). The higher food waste amount observed for high-income countries can perhaps be attributed to the fact that these countries waste more food within food supply chain whereas, in less developed countries, such as low-income countries, food is wasted during food preparation in households (Parfitt et al., 2010).

Figure 4-2 indicates the association between food waste fraction and income category. As can be seen the fraction of food waste appears to decrease as the income of countries increases.

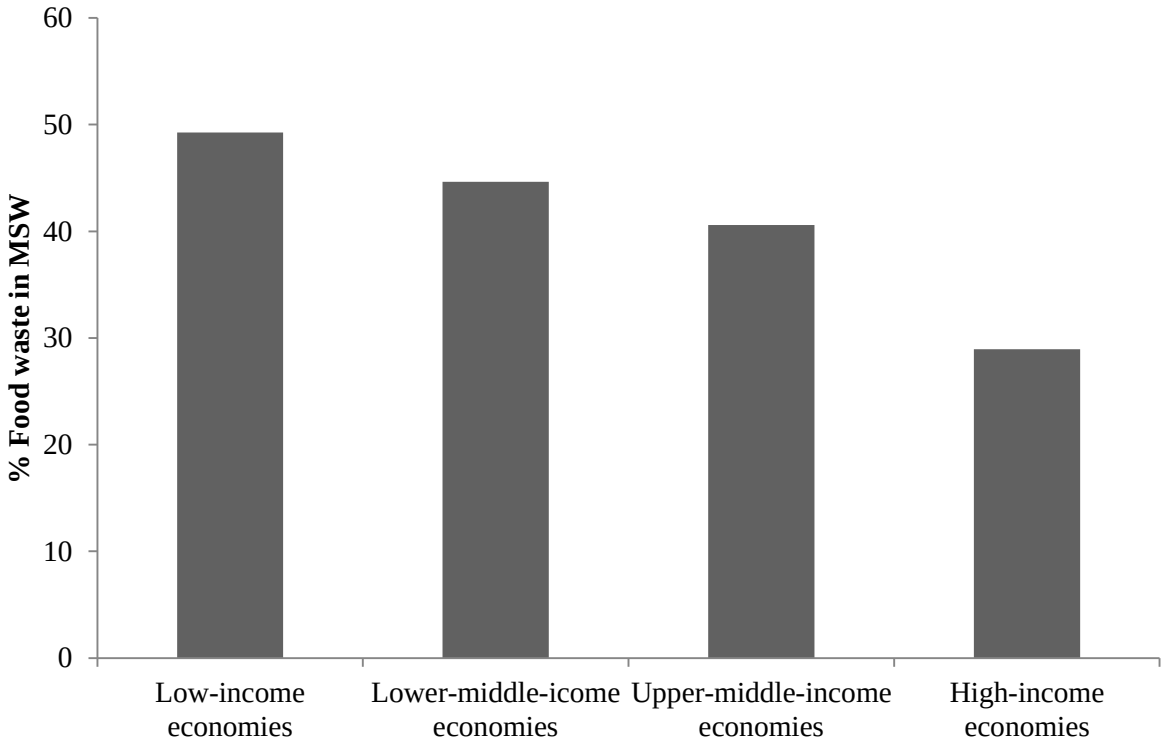


Figure 4-2 Fraction of food waste (%) in the MSW generated for each category

This can possibly be explained by the fact that MSW generation in low-income countries appears to be low and hence, food waste may contribute considerably in the total amount of waste generated. Additionally, in low-income countries, the lack of infrastructure and technical and managerial skills in food production and post-harvest processing have been the main reasons for the high generation of food waste (Parfitt et al., 2010). Another explanation to the high food waste generation could be that technologies which can keep the food fresh, such as refrigerators or freezers are not always available in these countries, resulting to the rotting and disposal of food to the waste stream (Sharholy et al., 2008).

4.3 The use of FWDs

FWDs were first introduced in the USA in the 1930s and since then their use had become widespread. In the beginning of 1970s, FWDs had been banned in New York City, based on the consensus that their use increased the loads in the wastewater treatment plants (WWTPs) and caused damage to the sewerage system (Department of Environmental Protection and The City of New York, 1997). In 1997 the ban on FWDs was rescinded, after a 21-month study conducted by the NYC Department of Environmental Protection (1997) which concluded that the use of FWDs did not present any significant impacts on water consumption, sewerage system, wastewater treatment and sludge management (Department of Environmental Protection and The City of New York, 1997). Nowadays, the use of FWDs is a usual practice in the USA and over 95% of the American cities allow their use (Environment Protection Agency, 2008). Their installation in USA households amounts to 50%, which is by far the highest penetration rate across the world. In Canada the installation rate of FWDs is 10% of households, whereas in Australia and New Zealand, it is about 12% and 30% of households, respectively. In these countries, the use of FWDs is not prohibited but each municipality has the authority to restrict their use. Decisions over restriction are usually based on sewer conditions and the ability of local WWTPs to accept the additional load (Environment Protection Agency, 2008). In Japan there are two types of disposers in use, direct-discharge disposers alone and those with an attached treatment tank. The use of each type of FWD is controlled by the local governments which can either impose limitations on their use or request restraint when it comes to the installation of direct-discharge disposers (Battistoni et al., 2007; CECED, 2003; EPA, 2008; NILIM - National Institute for Land and Infrastructure Management, 2005).

In Europe, the legislation on non-hazardous wastes leaves it to member states to decide which management process is more suitable for the treatment of their waste (Burnley, 2001). As such, legislation and policies over the use of FWDs vary amongst EU member states, with factors such as environmental awareness, capacity of the WWTPs, marketing opportunities for sludge and biogas and cultural attitudes towards technological developments being the main drivers (CECED, 2003). In the UK there is considered to be no regulation on the use of FWDs (CECED, 2003). The installation rate

of FWDs in the UK is about 5% which is the highest rate amongst member states. In Italy the use of FWDs was banned in 1997, but lifted in 2002, since it was considered that there was no objective basis for a ban and alternative ways of food waste management were needed in areas where the collection of waste was difficult (CECED, 2003). Since then, in Italy and also in Ireland, the municipalities can decide whether to restrict the use of FWDs or not. In Denmark, France, Norway, Finland and the Netherlands, permission is required prior to installation of a FWD. The permission depends on whether the local authorities (LAs) are ready to take the responsibility for any environmental impact or damage (CECED, 2003). In Germany, Austria, Belgium, Luxembourg and Sweden, the use of household FWDs is either not allowed or discouraged for a number of reasons, of which the most important is the potential impacts of FWDs on sewers and WWTPs (Environment Protection Agency, 2008). However, in Sweden the use of FWDs is allowed when the unit is connected to a tank, rather than the sewage system (Environment Protection Agency, 2008).

It must be highlighted that in the UK, the hospitality and food service (HaFS) sector uses widely FWDs, as a mean to divert food waste from landfill.

4.4 The environmental impacts of the use of FWDs

A number of studies, some theoretical (De Koning & Van der Graaf, 1996; Diggelman & Ham, 2003; Evans, 2007; Galil & Yaacov, 2001; Henze, 1997; Lundie & Peters, 2005; Marashlian & El-Fadel, 2005; MTP, 2008) and others experimental (Battistoni et al., 2007; Bolzonella et al., 2003b; Department of Environmental Protection and The City of New York, 1997; Galil & Shpiner, 2001; Karlberg & Norm, 1999; Magagni et al., 1996; Nilsson et al., 1990; Rosenwinkel & Wendler, 2001; Sankai et al., 1997; Wainberg et al., 2000), have been undertaken in order to examine the performance and potential implications of the use of FWDs. Some of these studies had been commissioned by FWDs manufacturers, while others were independent.

The benefits of using FWDs are many and varied. One of them is the diversion of food waste from landfills. A case study undertaken in Sweden was one of the first major studies on the use of FWDs (Nilsson et al., 1990). It was carried out for a period of two years and involved 100 apartments, housing 211 people. Even though not all the households used their FWD, the amount of food waste disposed of to landfill was decreased by 8 %. Food waste diversion reported in other studies ranged between 7 and 43% based on a 25 to 75% FWD installation rate (Galil & Yaacov, 2001; Marashlian & El-Fadel, 2005). In a desk study in the UK, it was estimated that by 2030, 0.38 to 6.25 million tonnes of waste could be diverted from landfill per year through the use of FWDs. However, based on the penetration rate of 5%, food waste diversion was found to be minimal (MTP, 2008).

Other benefits reported in the literature regarding the use of FWDs are the elimination of the negative impacts of waste disposal to landfill, the minimisation of energy consumption and GHG emissions from the handling of wastes by truck and the reduction of disease-causing vectors such as flies, rodents and rats that are attracted to food waste (Diggelman & Ham, 2003; Henze, 1997). It was reported that the use of such units can be a cost-effective, convenient and hygienic mean of separating food waste at source (Evans, 2007; MTP, 2008). In addition, the application of these units in areas where the collection of wastes is complicated and expensive, such as in small, decentralized towns and hilly or mountainous areas was considered of fundamental importance (Battistoni et al., 2007).

Beside the reported benefits that characterise FWDs, concerns over water consumption, damage to sewerage systems, changes in the wastewater characteristics and sludge production, are also associated with their use. These are further discussed and analysed in the following sections.

4.4.1 Impacts on energy consumption

For the operation of FWDs, electricity is used and hence, energy is consumed. The energy consumed by FWDs may depend on the model, type, frequency and duration of use (MTP, 2008). In the market, domestic FWDs models are typically rated between 0.5 to 0.75 horsepower (HP) (370 to 560 W), but commercial FWDs are rated up to 1.0 HP (746 W) (Waste Disposer Shop, 2010). The frequency and duration of use is difficult to define as this is based on homeowners' behaviour and habit. A frequency and duration of use of 2.2 times per day for 11.5 to 16.8 seconds per use, was reported by Ketzenberger (1995), whereas in other studies it ranged between 2.4 and 2 to 3 uses per day and lasted for about 38 and 36 seconds, respectively (Department of Environmental Protection and The City of New York, 1997; Nilsson et al., 1990). In a recent study, Evans (2007) reported that a use of 2.4 times per day for 16 seconds per use, corresponded to an annual energy consumption of 2 to 3 kW per household, whereas in the study of Karlberg and Norm (1999) annual energy consumption was reported at 3 to 4 kW per household. Other estimates ranged between 3.9 to 7.7 W per capita per day for a use of 36 to 75 seconds per day and 2 W per capita per day for a use of 24 seconds per day, for a 0.5 HP FWD (Bolzonella et al., 2003b; Magagni et al., 1996). Moreover, Lundie and Peters (2005) reported energy consumption of 0.02 kW per kg of waste, while an LCA by Diggelman and Ham (2003) demonstrated that the use of FWDs per 100 kg of waste required the lowest energy consumption. Although, reported as negligible in all studies, energy consumption by the use of FWDs was quoted as one of the disadvantages of the use of these units in the CECED (2003) report.

Finally, the cost of electricity consumed to run FWDs and its associated pollution was reported to be relatively insignificant (Marashlian & El-Fadel, 2005). For instance, in the study of Bolzonella et al (2003), it was estimated that an additional energy use of 4.3kWh and 8.5kWh per year, would result to an additional cost of €0.55 and €1.10 per year, respectively.

4.4.2 Implications for water consumption

Running water during the use of FWDs is essential for the production of a homogenized mixture that is able to be flushed into the sewer. The additional water consumption due to the use of FWDs and the percentage contribution to the total per capita water consumption reported varies substantially in different studies (Table 4-2). The increase in water consumption ranged between 1 and 6.6 litre/cap/day, while the percentage contribution varied between 0.3-3.5 per cent.

Table 4-2 Water consumption rates due to the use of FWDs and the percentage contribution to the total water consumption

Study	Water Consumption (litre/cap/day)	Increase in water consumption (%)
Ketzenberger et al., 1995[1]	0.3-3.5	n/a
De Koning and Van der Graaf, 1996	1.1-4.5	3.3
NYC, DEP, 1997	3.78	n/a
Wainberg, 2000	2.95	n/a
Rosenwinkel and Wendler, 2001	4.5	3.5
Galil and Sphiner, 2001	0.8-6.6	0.3-2.8
Bolzonella et al., 2003	1-1.9	0.4-0.8
Diggelman and Ham, 2003	1.03	n/a
Marashlian and El-Fadel, 2005	n/a	0.72-2.35
Sinclair Knight[2]	4.5	n/a

[1]Adapted from Defra's Market Transformation Programme; [2] Adapted from Wainberg (2000)

In all studies, it was reported that water consumption due to the use of FWDs (Table 4-2), is negligible compared to the total per capita water consumption. Based on the average per capita water consumption, which ranges between 150-575 litre per day (Data360, 2008; Waterwise, 2010), it can indeed be suggested that the additional water consumption of 1-7 litre per day due to the use of FWDs would be insignificant. However, in countries where the risk of water scarcity is high, either due to low availability or high demand, even low rates of growth in water consumption might be of great

concern, especially were there to be long-term and large-scale use of FWDs (Feitelson & Chenoweth, 2002; World Resources Institute, 2008). In the studies of De Koning and Van der Graaf (1996) and Diggelman and Ham (2003), while it was reported that water consumption rates were insignificant (Table 4-2), it was added that it is difficult to evaluate the actual volume of water consumed by these units. This is because water consumption can depend on different household practices (Diggelman & Ham, 2003). The amount of food waste that can be generated per capita per day, the actual amount of food waste ground, and the frequency and duration of use of the unit are some of the factors that can cause significant variations in water consumption rates (MTP, 2008). In the UK it was estimated that FWDs use water in greater quantities than the volume of food waste ground (Water UK, 2009). In the UK, it was also reported that while water utilities are trying to reduce the per capita water consumption per day by promoting water-efficient products and technologies, such as water-saving devices for flushed water, an equivalent of 1 toilet flushing (6-4 litre) additional water is being consumed by FWDs (Defra, 2008; Water UK, 2009; Waterwise, 2010).

Finally, it was reported that FWDs not only can increase water consumption but can also contribute to water pollution (European Parliament and Council, 2000). In the UK, it was reported that significant volumes of high quality water can be wasted due to the use of FWDs (Water UK, 2009).

4.4.3 Impacts on the sewerage system

It was suggested that suspended solids (SS) deposition, fat, oils and grease (FOG) deposition, hydrogen sulphide (H_2S) generation and overloads are some of the potential impacts that ground food waste disposal can cause to the sewerage system, but there is little evidence to support this.

It has been reported that incidents of sewer blockages due to the increased SS loading rate were not detected or expected to occur in the sewers due to the use of FWDs (De Koning & Van der Graaf, 1996; Marashlian & El-Fadel, 2005; Nilsson et al., 1990). A direct investigation with video and photo inspections in the sewerage line, undertaken in a number of studies, showed no significant SS deposition before, during or after the use of FWDs (Battistoni et al., 2007; Department of Environmental Protection and The City of New York, 1997; Karlberg & Norm, 1999; Nilsson et al.,

1990). However, in a few studies it was reported that SS deposition is possible (Bolzonella et al., 2003b; Galil & Shpiner, 2001). For instance, Bolzonella et al., (2003) demonstrated that the different fractions of organic wastes disposed in the sewer, the size of the ground particles and their settling velocities are all important parameters of their susceptibility to settlement, and explained that some deposition can be expected in the sewer. These findings were in line with the results reported in the study of Galil and Shpiner (2001).

As food waste contains FOG, this can solidify and stick to pipes, thus causing blockages to the sewers (Brooksbank et al., 2007; Rinzema et al., 1994). However, in some studies where FOG blockages were undetected it was proposed that food waste is discharged with cold water and therefore, any grease or fat will coalesce onto other particles, rendering them unable to attach in the sewer surface (Battistoni et al., 2007; De Koning & Van der Graaf, 1996; Evans, 2007). In contrast, in the New York City study it was suggested that a large uptake of FWDs could increase FOG blockages, leading to increases in cost for sewer maintenance. In the UK, it has been suggested that food waste could be one of the major causes of FOG blockages (Perriam, 2007; Water UK, 2009). It has also been proposed that in areas where blockages are already a usual phenomenon, the additional load due to the use of FWDs could be important. Blockages are not only be caused by FOG deposition, but also due to items flushed down the toilet, the build-up of washing powder and SS deposition, among other factors (Cleansing Service Group, 2010).

The National Clay Pipe Institute (1995) reported that food waste might cause damage in the sewage system due to the potential increase in the production of H_2S (The National Clay Pipe Institute, 1995). H_2S can be formed when sulphide is produced through the anaerobic degradation of organic matter in the sewers. It is a known corrosive agent able to attack clay pipes, concrete and metal surfaces, causing deterioration of the sewage system and subsequent leakages (Boon, 1992; Nielsen et al., 2006; The National Clay Pipe Institute, 1995). It can also cause malodours at very low concentrations and can be toxic to human health (Æsøy et al., 1998; Æsøy et al., 1997; Boon, 1992; Nielsen et al., 2006). In the majority of the studies reviewed, sulphide generation was not reported to occur due to the use of FWDs. However, in the study of Wainberg et al., (2000) it was reported that the disposal of food

waste to the sewers can cause significant increases in the H_2S concentration, especially with increasing FWDs uptake.

Overloads to storm drains, due to changes in climatic conditions and the lack of a dual system, can occur and the use of FWDs can add to this problem. When overloads occur, polluting sewage can escape from the sewer and contaminate the surrounding environment. In countries where the sewerage system is well monitored and/or the size of the pipes and sewer lines is big, overloads are less likely to occur and the system is capable of accepting the additional load due to food waste disposal (Karlberg & Norm, 1999; Marashlian & El-Fadel, 2005). However, in countries where the system is ageing and poorly monitored, this is not the case, as the increasing urbanization and more frequent and intense rainfall events can affect the sewerage system's capacity and the additional load due to the use of FWDs would add significantly to the problem (Water UK, 2009).

4.4.4 Impacts in wastewater characteristics

The increases in the parameters that make up wastewater (WW) characteristics due to the use of FWDs have been widely documented. Total suspended solids (TSS), the biochemical oxygen demand (BOD), the chemical oxygen demand (COD), nitrogen (N) and phosphorus (P) were examined in most of the studies.

The reported per capita increase in the wastewater parameters and the specific contribution to the typical per capita rates vary widely between the studies (Table 4-3). The increase in TSS, BOD and COD concentrations ranged between 7-91 g/cap/day, 6-77 g/cap/day and 18-165 g/cap/day, respectively, while for N and P, it ranged between 0.6-10 and 0.1-3 g/cap/day, respectively. The specific contribution, reported in the studies, varied between 2-60%, 7.5-62%, 15-44%, 1.4-19% and 1.2-14%, for TSS, BOD, COD, N and P, respectively. However, these rates are not indicative for all studies.

Table 4-3 Additional loading in wastewater characteristics due to the use of FWDs

Study	TSS		BOD		COD		N		P	
	g/cap/day	***% increase***	***g/cap/day***	***% increase***	***g/cap/day***	***% increase***	***g/cap/day***	***% increase***	***g/cap/day***	***% increas***
Nilsson et al., 1990	34	48	31	48	88	n/a	10.2	12	3.1	n/a
De Koning and Van der Graaf, 1996[1]	48	n/a	52	n/a	76	n/a	1.6	n/a	n/a	n/a
Rosenwinkel and Wendler, 2001[1]	28-40	40-60	6-15	10-25	18-36	15-30	1.5	5-10	0.13-0.25	7-14
Bolzonella et al., 2003	50	n/a	n/a	n/a	75	n/a	2.5	n/a	0.25	n/a
Battistoni et al., 2007	11	30	n/a	n/a	55	44	2	19	n/a	n/a
Galil and Sphiner, 2001[1]	7-34	n/a	10-31	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Karlberg and Norm, 1999	n/a	n/a	n.d.	n/a	n/a	n/a	n.d.	n/a	n.d.	n/a
DEP, NYC, 1997 (Queens)	21.6	n/a	23.6	n/a	37.6	n/a	4.5	n/a	0.45	n/a
DEP, NYC, 1997 (Brooklyn)	117	n/a	77.2	n/a	164.6	n/a	8.5	n/a	1.2	n/a
DEP, NYC, 1997 (Manhattan)	20.9	n/a	42.2	n/a	54	n/a	5.8	n/a	0.54	n/a
Gatto, 2000 [2]	50	n/a	n/a	n/a	106	n/a	12	n/a	0.9	n/a
Magagni et al., 1996 [2]	20.8-90.6	n/a	10.4-36	n/a	n/a	n/a	0.6-2	n/a	0.1	n/a
Marashlian and El-Fadel, 2005	n/a	1.9-7.1	n/a	17-62	n/a	n/a	n/a	n/a	n/a	n/a
Diggelman and Ham, 2003	n/a	7.9	n/a	7.5	n/a	n/a	n/a	1.4	n/a	1.2
Jones et al., 1994	n/a	n/a	n/a	16.5	n/a	n/a	n/a	3	n/a	4.6

[1] Adopted from Bolzonella et al., 2003; [2] Based on literature

While the increase in TSS, BOD and COD concentrations shows that FWDs can substantially affect the wastewater strength, it was reported that the additional loadings to WWTPs would not cause any significant modifications to the wastewater treatment. It was proposed, however, that increases in cost would be incurred due to the need for additional aeration and nutrient control measures (Battistoni et al., 2007; Bolzonella et al., 2003b; De Koning & Van der Graaf, 1996; Karlberg & Norm, 1999; Marashlian & El-Fadel, 2005; Nilsson et al., 1990). In a few studies, it was suggested that the increases in the organic content of wastewater can be beneficial for the wastewater treatment processes, as it can improve the biological nutrient removal and hence reduce the cost of chemicals used to remove nutrients from the wastewater (Battistoni et al., 2007; De Koning & Van der Graaf, 1996).

Despite the benefits, in some studies it was reported that the use of FWDs at a small scale would not lead to an uncontrolled change at the WWTP, but that the long term and large scale use of FWDs could incur an additional capital and operational cost. For instance in the New York City study (1997), it was reported that the additional load in the WWTP resulting from the long term and large scale use of FWDs would require increased capacity and more intense treatment processes in order to remove the extra organic and nitrogen content. It was also suggested that the increased flow rate would incur additional costs due to the increased pumping and chlorination requirements (Department of Environmental Protection and The City of New York, 1997).

4.4.5 Impacts in sludge and biogas production

The increase in TSS and BOD/COD rates due to the use FWDs can lead to a subsequent increase in sludge and biogas generation rates. However, the treatment processes used at WWTPs can cause significant variations in both of these. In the study of Galil and Yaacov (2001), three wastewater treatment processes, namely biological treatment (BT), primary sedimentation followed by biological treatment (PS-BT) and primary chemically enhanced sedimentation followed by biological treatment (PCS-BT) were examined. It was reported that raw sludge production in the BT treatment was increased by 24-38 g/cap/day, but when the primary settler was present, sludge was increased by 50-

80 g/cap/day. In the case of PCS-BT, the extra sludge generated was 67–100 g/cap/day. The additional energy potential from biogas production ranged between 50-73%, with the highest (73%) being observed for the PS-BT treatment. Similarly, Bolzonella et al., (2003) supported that sludge and biogas production were increased significantly when the primary settler was present compared to when it was not.

The increase in sludge generation ranged between 4-70%, while the extra biogas generation ranged between 22-100% (Table 4-4). These increases were regarded as beneficial in all studies, since CH_4 produced during the anaerobic digestion (AD) of sludge could be recovered and used as a renewable source of energy.

Table 4-4 Extra sludge and biogas generation rates due to the use of FWDs

Study	**Sludge**		**Biogas**	
	g/cap/day	***% increase***	***l/cap/day***	***% increase***
Nilsson et al., 1990	n/a	70	n/a	50
De Koning and Van der Graaf, 1996	n/a	4	n/a	n/a
Jones et al., 1994	n/a	18.1	n/a	n/a
Rosenwinkel and Wendler, 2001	50	62	16-25	90-100
Karlberg and Norm, 1999	n/a	n/a	n/a	22
Galil and Yaacov, 2001	50-80	n/a	n/a	73
Bolzonella et al., 2003	n/a	37	n/a	30

It was suggested that CH_4 produced can replace the electricity generated from fossil fuels and, hence, offset the cost of energy consumption associated with the increased oxygen requirements and sludge treatment (Battistoni et al., 2007; Bolzonella et al., 2003b; Diggelman & Ham, 2003; Galil & Yaacov, 2001; Karlberg & Norm, 1999; Marashlian & El-Fadel, 2005; Nilsson et al., 1990; Rosenwinkel & Wendler, 2001). However, in a few studies it was demonstrated that the increases in sludge production can potentially increase the operating cost due to the additional sludge digesters, dewatering facilities and biosolids handling (Department of Environmental Protection and The City of

New York, 1997; Rosenwinkel & Wendler, 2001). It was also suggested that if a WWTP does not have anaerobic digestion facilities or if it is unable to utilize the CH_4 produced, then there is no way to offset the additional treatment cost (CIWEM, 2003).

Together with the change in volume, sludge composition may also change. This change necessitates that regulations over the treatment and disposal of sludge to land will have to be reviewed and possibly restricted. Depending on the composition, sludge should perhaps be subjected to a treatment process that is sufficient enough to remove organic pollutants and eliminate pathogens. However, in the study of Diggelman and Ham (2003), it was reported that sludge of good characteristics can be obtained and used as a soil conditioner.

4.4.6 Impacts on greenhouse gas (GHG) emissions

The impact of FWDs on GHG emissions was neglected in the majority of the studies. Increases in BOD, SS and TKN can certainly increase GHG emissions during the wastewater treatment process, while increases in energy consumption and sludge haulage, treatment and disposal can also contribute to GHG emissions. Evans (2007) suggested that the net GWP of the use of FWDs is better than the solid waste collection followed by route with centralised composting, and landfill. According to this author, FWDs generate -16.8kg CO_2 eq. per 100kg of food waste, whereas centralised composting and landfilling generate -1.4kg and +74.3kg CO_2 eq. per 100kg of food waste respectively. However, it should be noted that the net GWP estimated in this study included not only the GHG emissions from energy consumption but also the energy recovery from biogas production, carbon sequestration and fertilizer offset. In the life cycle assessment (LCA) study by Wainberg et al. (2000) the use of FWDs was ranked second best on net GHG potential after home composting with 42.2kg CO_2 eq. per 100kg food waste, when compared to home composting, centralised composting and co-disposal of food waste with MSW. Similarly, in the LCA study of Diggelman and Ham (2003), it was demonstrated that the use of FWDs followed by a publicly owned treatment plant had the lowest GHG emissions when compared to the use of FWD followed by a rural on-site treatment system, MSW collection

followed by landfill, MSW collection followed by composting and MSW collection followed by a waste-to-energy options, which was 44kg CO_2 eq. per 100kg of food waste.

However, it is important to note that in the LCA studies there might be differences in the direct and indirect emissions, due to the different area characteristics and practices, resulting in a different net GHG associated with the use of FWDs. Therefore, a LCA of FWDs with other food waste management options based on specific area characteristics and practices (with regards to collection, transportation and pre-treatment), is necessary to assess GHG emissions.

4.5 Conclusions

The use of FWDs for diverting food waste from landfill could be a useful waste management option, particularly in situations where regular separate food waste collection from households is difficult. FWDs are practical and relatively inexpensive and their use can effectively reduce the need of separate bins for collection and disposal, and can also offer a more convenient way of dealing with food waste. However, the use of such units is associated with potential effects that make their sustainability uncertain. Water consumption, potential impairment to sewerage systems and impacts on wastewater treatment due to changes in wastewater characteristics or increased sludge production are all key issues that need to be considered before their application as a sustainable waste management option.

The experimental and theoretical studies undertaken so far have shown that the use of FWDs has no significant impact on water consumption, sewerage systems and the wastewater treatment plants, however, the likelihood of potential effects and the associated economic implications, especially at a long term and large scale use of FWDs, has been highlighted. The different area-specific characteristics such as water resources, the condition of the sewerage system and the various treatment processes practiced in the WWTPs, as well as the associated GHG emissions and other environmental impacts, are important factors that must be taken into account before the implementation of FWDs as a wide scale waste management option. Further research and development on FWDs could provide opportunities for some of the above parameters to be better addressed and managed for this option to become a sustainable alternative to landfill in the future.

5 FOOD WASTE DISPOSAL UNITS IN UK HOUSEHOLDS: THE NEED FOR POLICY INTERVENTION

5.1 Introduction

The disposal of biodegradable waste to landfill can lead to the formation of landfill gas and leachate which can result in adverse environmental impacts. To control these impacts, the European Union (EU) Landfill Directive (1999/31/EC) came into force and requires all EU member states to reduce the amount of biodegradable waste going to landfill to 75% by 2006, 50% by 2009 and 35% by 2016, based on the waste production levels of 1995 (European Council, 1999). Countries which have been heavily reliant on landfill, such as the UK, have an additional four years to comply with the targets set in this Directive. Therefore, in order to reduce the amount of biodegradable waste going to landfill to 75% by 2010, 50% by 2013 and 35% by 2020, alternatives to landfill are increasingly being considered across the UK (European Council, 1999).

In the UK, local authorities (LAs) have the responsibility for managing municipal solid waste, of which a high proportion is biodegradable. To deliver the required landfill diversion targets they have promoted and implemented a wide range of waste management options including kerbside collection and recycling, home and centralised composting, centralised incineration, anaerobic digestion, and the use of food waste disposal units (FWDs) (Defra, 2007b).

The use of FWDs foresees the diversion of food waste from the solid waste stream through grinding with the addition of water for direct discharge as wastewater in the sewers (Evans, 2007; Galil & Shpiner, 2001). Although most LAs would treat household food waste collected as part of the biodegradable fraction or either separately, some (LAs) have been encouraging the installation and use of FWDs. As a result, in order to tie up with the recommendations set by LAs, FWDs have recently been installed in new build housing developments.

In 2008, in the UK, 5% of households had FWDs (MTP, 2008), an installation rate considered to be the highest amongst the EU member states (Environment Protection Agency, 2008). Although their

use is controlled in European countries, being banned in Austria, Belgium and Germany, and regulated locally by municipalities in Italy, France and Sweden, the story is very different outside the EU. The highest installation rate is found in the USA, where 50% of households have FWDs, whereas in Canada, Australia and New Zealand the installation rate of FWDs is about 10%, 12% and 30% of households, respectively (Environment Protection Agency, 2008).

This uneven FWDs installation rate is mainly attributed to the belief that the use of these units can be associated with a number of technical and environmental limitations. For that purpose many studies have been undertaken to examine the feasibility of their use (Chapter 4). In these studies, recommendations on the use of FWDs as a waste management option differ widely and there is widespread uncertainty regarding their potential benefits and impacts to the wastewater treatment works. This is mainly because the different area-specific characteristics such as water resources, household practices, the condition of the sewerage system and different wastewater treatment processes can affect the viability of FWDs as a waste management option. These characteristics are important factors that must be taken into account before the adoption of FWDs as a wide scale waste management option (Chapter 4).

In the UK, waste collection and disposal is not under the same authority as water industry operations and thus the use of FWDs becomes more complicated, as it would transfer the responsibility for food waste management from LAs to the water industry. As such, whilst LAs could benefit by reductions in the amount of waste collected, the water industry would be left to deal with additional costs related to water treatment and distribution, wastewater treatment, and sludge treatment and disposal, without the transfer of the associated revenues.

Based on this supposition and with limited research undertaken in the UK on the potential impacts of the use of FWDs, this Chapter aims to evaluate the additional burden to water industry operations in the UK, compared with the benefits and related savings from the reduction in cost of waste collection and disposal to LAs, using the Anglian Region as a case study. Although, the proposed methodology can be applied to any region, the Anglian Region was selected because of the availability of data.

5.2 Methods

The Anglian Region, one of the areas with the fastest growing population in the UK, was chosen as a case study. This region was identified by the Environment Agency as the driest region in the UK, with an average of 600 mm of rainfall each year, in contrast to the average 900 mm for the rest of England and Wales (Anglian Water, 2008). Anglian Water is the main provider for water and/or wastewater treatment services in this area (East of England) (Anglian Water, 2008). Details on services provided in the Anglian region can be found in Appendix B. Based on the latest available data provided by the LAs and Anglian Water, year 2008 was chosen as base year for calculations and 2035 as the projection year (Appendix B).

To evaluate the additional burden of the use of FWDs from year 2008 to year 2035, three scenarios were investigated. In these scenarios a current, future and hypothetical market installation rate were used to project the FWD penetration in 2035, based on the 2008 average FWD installation rate of 5% (MTP, 2008). The purpose of having these three scenarios was to show the magnitude of large-scale use of FWDs over the current penetration rates (Table 5-1).

Table 5-1 Scenarios of household FWD penetration rate

Scenario	FWD penetration rate		Assumption
	Increase per year (%)	Projected in 2035 (%)	
Low	0.3	10	Market penetration remains stable at current rates
Medium	1	24	Market penetration increases as predicted by MTP (2008)
High	15	96	Hypothetical case where FWDs are promoted as the desirable waste management option

In order to evaluate the benefits resulting from the use of FWDs, a simple economic analysis was undertaken. Cost elements borne by LAs included the collection and disposal of household residual

waste. Household residual waste includes the waste that has not been separated for reuse or recycling. Food waste is a fraction of household residual waste and accounts for approximately 36.7% of it. Therefore this type of waste was considered in the analysis. For the water industry, cost elements such as the water and wastewater treatment, sewer clean and sludge management were considered (Figure 5-1).

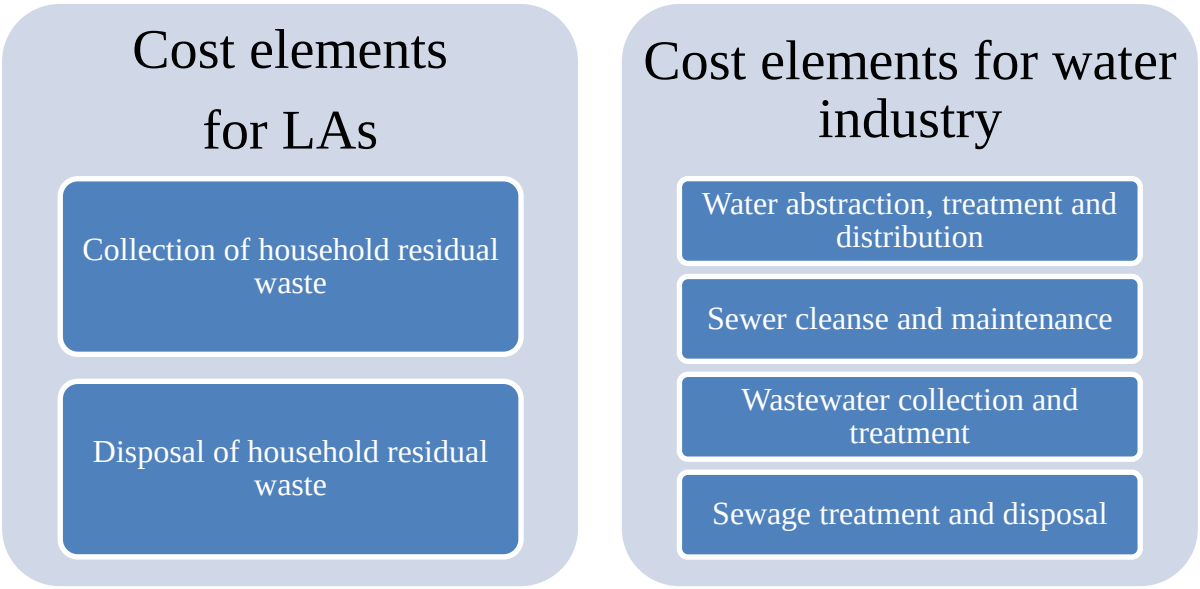


Figure 5-1 Cost elements used for comparing the LAs savings to the water industry costs

The cost of household residual waste collection is largely dependent on the number of households in the Anglian region, whereas the cost of household waste disposal depends on the amount of household residual waste collected. According to that, changes in the fraction of food waste were assumed not to create any changes in the cost of waste collection. A reduction in the amount of food waste due to the use of FWD would reduce the amount of household residual waste for disposal, and therefore the associated cost. To calculate this cost, both the amount of household residual waste generated in the Anglian region and the fraction of food waste within it were first calculated. For each FWD penetration rate scenario, the food waste fraction in residual waste was changed and so, as a result, was the amount of household residual waste. The calculated amount of household residual waste collected and the cost value of household residual waste disposal (Table 5-2) were used to

project the cost of household residual waste disposal to LAs in 2035. To calculate the cost of household residual waste collection, the number of households in the Anglian region was first calculated. This number, together with the cost value of household residual waste collection (Table 5-2), was used to calculate the cost of waste collection in the Anglian region (Audit Commission, 2011). The cost values of household residual waste collection and disposal were based on the 2008 average costs, as estimated on the basis of data collected from LAs in the Anglian region. Inflation rates in these costs by 2035 were not included in the calculations not only to exclude the inherent uncertainties in these, but also to ensure that all costs are directly comparable. However, it is worth mentioning that the government, in a way to reduce the UKs dependence on landfill, has put landfill tax on an escalator from 2008-09 until 2014 (Defra, 2009b). The annual increase in landfill tax is set at £8 per tonne of waste, which means that by 2014 landfill tax will be £80 per tonne of waste (Defra, 2009b).

The costs of water treatment and distribution, wastewater collection and treatment, and sludge treatment and disposal, were calculated based on water consumption, tonnage of biochemical oxygen demand (BOD), and dry solids (DS) treated in the whole area, respectively (Table 5-2) (Ofwat, 2008). Thus, increases in these parameters due to FWDs were used to estimate the additional costs. However, the marginal cost to treat an additional tonne of BOD or DS can be very different, depending on the method used. That is because some wastewater treatment plants can incur relatively small additional costs from BOD increase, whereas others have BOD, suspended solids (SS) or even heavy metal limits that disproportionally affect the marginal costs of an additional volume. In addition, expenditures associated with sewer cleansing and maintenance, although they constitute a cost to the water industry, were not calculated in the analysis mainly because there are no direct costs associated with the disposal of food waste in the sewer. Therefore it was difficult to accurately estimate how the disposal of food waste would increase these costs in the future. Although treatment of sewage sludge is costly, there are inherent benefits such as the increased generation of biogas, through anaerobic digestion. Various treatment processes are currently used to stabilise sludge, ranging from simple mesophilic anaerobic digestion to advanced treatment methods such enzymic or thermal hydrolysis.

Biogas production and energy consumption by these processes are highly variable and as such, these benefits could not be reliably estimated in this study (Caldwell, 2009; Jolly & Gillard, 2009).

In addition, inflation rates by 2035 for costs associated with water treatment and distribution, and wastewater collection and treatment, were as before, not included in the calculations due to the uncertainties related to them and to ensure that costs are directly comparable.

Table 5-2 Average costs to LAs and water and wastewater services in the East Anglia region

Parameters	Description	Value	Reference
Average cost of water consumption[1]	Provided	£191.22/Ml	Ofwat, 2008
Average cost of wastewater treatment[1]	Provided	£486.21/tonneBOD	Ofwat, 2008
Average cost of sludge treatment and disposal[1]	Provided	£217.43/tonneDS	Ofwat, 2008
Average cost of household residual waste collection[2]	Adopted	£50.20/household	Audit Commission, 2011
Average cost of residual waste disposal[2]	Adopted	£53.70/tonne residual waste	Audit Commission, 2011

[1]Details can be found in Appendix B.2; [2]Details can be found in Appendix B.4

As previously mentioned, for calculating the costs to LAs and the water industry, certain parameters such as the number of households, the amount of household residual waste generated, the fraction of food waste diverted from residual waste, the amount of water consumed, the amount of wastewater, BOD, SS and DS generated in the whole region due to the use of FWDs, had first to be calculated. For calculating these parameters a number of equations were formulated. These equations were formulated based on a number of other parameters that have been either provided by the appropriate bodies, or adopted from the literature (Table 5-3) (Anglian Water, 2007; De Koning & Van der Graaf, 1996; Defra, 2009c; Essex & Southend waste LDF, 2010; Marashlian & El-Fadel, 2005; MTP, 2008; Ofwat, 2008; Thomas, 2011; WRAP, 2008a). The methodology followed to calculate the aforementioned parameters, is analytically described below.

The first step was to calculate the amount of total household food waste generated and the quantity diverted from landfill to the sewer. Total household food waste generation (FW_{gnr}) was estimated based on the amount of food waste generated per household ($FW_{gnr/hh}$), and the number of households in any given year (HH_{n+1}) from the base year (when, $n = 0$) in the Anglian Region as follows:

$$\boldsymbol{FW_{gnr} = FW_{gnr/hh} * HH_{n+1}} \qquad \textbf{Equation 5-1}$$

Knowing the amount of total household food waste generated in the region, it was then possible to estimate the fraction diverted from landfill, through the use of FWDs, based on food waste penetration rate ($FWD_{p.r}$) and the number of households that have a FWD (HH_{FWD}). This was calculated in each year, as follows:

$$\boldsymbol{HH_{FWD} = HH_{nFWD} + \left[(HH_{n+1} - HH_{nFWD}) * FWD_{p.r}\right]} \qquad \textbf{Equation 5-2}$$

Because not all food waste can be ground by a FWD, three main parameters were used to quantify the amount of food waste disposed of to the sewer: the amount of food waste generated per household, the number of households that have a FWD, and the fraction of food waste that could be ground by a FWD (FW_{grd}) as shown in equation (5-3).

$$\boldsymbol{FW_{FWD} = FW_{gnr/hh} * HH_{FWD} * FW_{grd}} \qquad \textbf{Equation 5-3}$$

Then, the following equation was used to calculate the percentage reduction of food waste landfilled:

$$\%\boldsymbol{FW_{DL}} = \left(\frac{\boldsymbol{FW_{FWD}}}{\boldsymbol{FW_{gnr}}}\right) * \boldsymbol{100} \qquad \textbf{Equation 5-4}$$

The percentage increase in water consumption was calculated based on the amount of household food waste disposed to the sewer through the use of FWDs (FW_{FWD}), the volume of water needed to grind

food waste (W_{FW}), and the daily amount of water consumed (W_{cap}) due to population growth (P_{n+1}), as follows:

$$\%\boldsymbol{W_{inc}} = \left[\frac{\boldsymbol{FW_{FWD} * W_{FW}}}{\boldsymbol{W_{cap} * P_{n+1}}}\right] * \mathbf{100} \qquad \textbf{Equation 5-5}$$

Similarly, the percentage increase in wastewater flow generation rate was calculated as shown in the following equation:

$$\%\boldsymbol{WW_{inc}} = \left[\frac{\boldsymbol{FW_{FWD} * (W_{FW} * FW_{m.c})}}{\boldsymbol{WW_{cap} * P_{n+1}}}\right] * \mathbf{100} \qquad \textbf{Equation 5-6}$$

Besides the amount of food waste that is disposed to the sewer through the use of FWDs and the water needed to grind it, the moisture content of food waste ($FW_{m.c}$), was also used for calculating the additional wastewater generation rate due to the use of FWDs (Equation 5-6).

To calculate the increases in BOD and SS, the concentration of BOD (X^1_{FW}) and SS (X^2_{FW}) in food waste and the average quantity of BOD (X^1_{cap}) and SS (X^2_{cap}) generated per capita were used, as follows:

$$\%\boldsymbol{X^{1,2}_{inc}} = \left[\frac{\boldsymbol{FW_{FWD} * W_{FW} * X^{1,2}_{FW}}}{\boldsymbol{X^{1,2}_{cap} * P_{n+1}}}\right] * \mathbf{100} \qquad \textbf{Equation 5-7}$$

Based on SS concentration the increase in dry solids (DS) generated due both to the use of FWDs and population growth was calculated as follows:

$$\%\boldsymbol{Y_{inc}} = \left[\frac{(\boldsymbol{X^2_{FW}/X^2_{WW}}) * \boldsymbol{Y_{WW}}}{[(\boldsymbol{X^2_{cap} * P_{n+1}})/\boldsymbol{X^2_{WW}}] * \boldsymbol{Y_{WW}}}\right] * \mathbf{100} \qquad \textbf{Equation 5-8}$$

Many of the parameters used in the above equations were provided from waste collection and disposal authorities and water industry operations in the Anglian Region (Appendix B.2), whereas others were adopted from literature or calculated as described above (Table 5-3).

Table 5-3 Parameters used in the equations

Parameters	Description	Value	Reference
Food waste generation ($\boldsymbol{FW_{gnr}}$)	Calculated	-	This Chapter
Kerbside household residual waste generation	Adopted	653kg/household/year (12.6kg/household/week)	Defra, 2009c
Food waste generation per household ($\boldsymbol{FW_{gnr/hh}}$)	Adopted	4.6kg/week (0.66kg/d)	WRAP, 2008
Number of households in the area and household growth per year ($\boldsymbol{HH_{n+1}}$)	Provided	2,347,335 and 1.35%	Anglian Water, 2007; Ofwat, 2008
Increase in number of households that have a FWD ($\boldsymbol{HH_{FWD}}$)	Calculated	-	This Chapter
Number of households in previous year that have a FWD ($\boldsymbol{HH_{nFWD}}$)	Calculated	-	This Chapter
FWD penetration rate ($\boldsymbol{FWD_{p.r}}$)	Adopted	Scenario based	MTP, 2008; this Chapter
Food waste diversion from landfill though the use of FWDs ($\boldsymbol{FW_{FWD}}$)	Calculated	-	This Chapter
Food waste ground by a FWD ($\boldsymbol{FW_{grd}}$)	Adopted	95%	Marashlian and El-Fadel, 2005
Food waste diverted from landfill ($\boldsymbol{FW_{DL}}$)	Calculated	-	This Chapter
Increase in water consumption due to the use of FWDs ($\boldsymbol{W_{inc}}$)	Calculated	-	This Chapter
Water consumption per kg food waste ground ($\boldsymbol{W_{FW}}$)	Adopted	11.7l	Thomas, 2011
Daily water consumption per capita ($\boldsymbol{W_{cap}}$)	Provided	150l	Ofwat, 2008
Population in the area and population growth per year ($\boldsymbol{P_{n+1}}$)	Provided	5,484,400 and 0.55%	Anglian Water, 2007; Ofwat, 2008

Increase in wastewater flow due to the use of FWDs ($\boldsymbol{WW_{inc}}$)	Calculated	-	This Chapter
Daily wastewater generation per capita ($\boldsymbol{WW_{cap}}$)	Provided	167.5l	Ofwat, 2008
Moisture content of food waste ($\boldsymbol{FW_{m.c}}$)	Adopted	70%	De Koning and Van der Graaf, 1996
BOD/SS increase due to the use of FWDs ($\boldsymbol{X_{inc}^{1,2}}$)	Calculated	-	This Chapter
Daily BOD generation per capita ($\boldsymbol{X_{cap}^{1}}$)	Provided	60g	Thomas, 2011
BOD load in food waste ($\boldsymbol{X_{FW}^{1}}$)	Adopted	8.37g/l	Thomas, 2011
Daily SS generation per capita ($\boldsymbol{X_{cap}^{2}}$)	Provided	80g	Thomas, 2011
SS load in food waste ($\boldsymbol{X_{FW}^{2}}$)	Adopted	7.71g/l	Thomas, 2011
DS increase due to the use of FWDs ($\boldsymbol{Y_{inc}}$)	Calculated	-	This Chapter
DS generation per cubic meter of wastewater ($\boldsymbol{Y_{WW}}$)	Provided	0.4kg	Essex and Southend Waste, 2010
SS content per cubic meter of wastewater ($\boldsymbol{X_{WW}^{2}}$)	Calculated	0.418kg	This Chapter

5.3 Results

This analysis shows that in 2008 approximately 563 kilo tonnes of food waste were generated in the Anglian Region, of which 27 kilo tonnes were diverted from landfill through the use of FWDs. By 2035, it is predicted that 809 kilo tonnes of food waste will be generated. The projected low, medium and high FWD uptake scenarios correspond to the diversion of 50, 155 and 708 kilo tonnes, respectively, of food waste from landfill to the sewer, leaving 759, 654 and 101 kilo tonnes to be landfilled. This corresponds to a significant reduction of 12%, 81% and 94% in landfilled food waste, for low, medium and high scenarios, respectively (Figure 5-2).

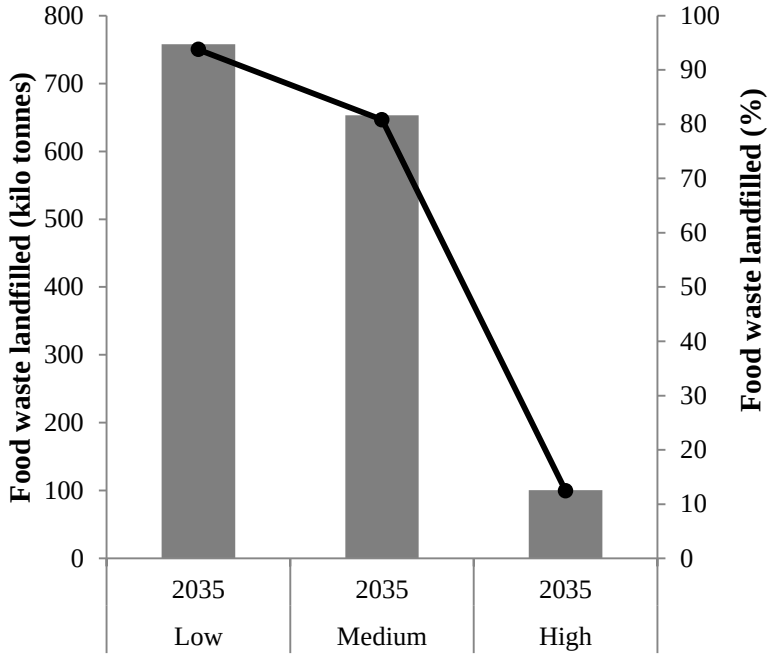


Figure 5-2 Food waste disposed of to landfill in the Anglian Region based on the three scenarios at year 2035

This diversion is the main cause of increases in water consumption, wastewater flow and characteristics and sludge generation rates, and corresponding decreases in household residual waste collected and disposed, each with associated cost increases and decreases.

In 2008, a diversion of 27 kilo tonnes of food waste from landfill to the sewer through the use of FWDs resulted in an increase of 0.86Ml/d out of the total 621Ml/d in water consumption with a 0.14% increase in the total cost. In terms of wastewater flow and characteristics, 0.91Ml/d out of 920Ml/d of the wastewater generated was attributed to the use of FWDs, with BOD and SS increases being 7 and 6.6 tonnes/d out of the total of 329 and 439 tonnes/d, respectively. These increases were responsible for a cost increase of £1.3million. Sludge generation and disposal accounted for a cost increase of approximately 1.51%, attributable to the additional volume of 5.5 tonnes/d from the total of 368 tonnes/d. The total cost of those increases to the water industry was estimated at £1.86 million out of the total £131 million.

In the projection year 2035, food waste diversion to the sewer would result in higher increases in water consumption, wastewater flow and characteristics (Table 5-4).

Table 5-4 Summary of the potential increases in water industry due to the use of FWDs, for all scenarios examined in year 2035

Parameter	Low		Medium		High	
	Additional load per day (year)	***Increase (%)***	***Additional load per day (year)***	***Increase (%)***	***Additional load per day (year)***	***Increase (%)***
Increase in water consumption in Ml ($\boldsymbol{W_{inc}}$)	1.6 (592)	0.2	4.9 (1,816)	0.7	22.7 (8,283)	3.2
Increase in waste-water flow in Ml ($\boldsymbol{WW_{inc}}$)	1.7 (627)	0.1	5.3 (1,925)	0.4	24 (8,778)	1.8
BOD increase in tonnes ($\boldsymbol{X^1_{inc}}$)	13.6 (4,955)	2.9	41.2 (15,201)	8.8	190 (69,326)	40.2
SS increase in tonnes ($\boldsymbol{X^2_{inc}}$)	12.5 (4,564)	2	38.4 (14,002)	6	175 (63,860)	27.8
DS increase in tonnes ($\boldsymbol{Y_{inc}}$)	10.5 tonnes (3,823)	2	32 tonnes (11,729)	6	146.5tonnes (53,492)	27.8

In 2035, the projected increases in water consumption, wastewater flow and characteristics would account for a substantial increase in their associated costs (Table 5-5).

Table 5-5 Expected cost increases to the water industry due to the use of FWDs in year 2035 for all scenarios

Description	Low		Medium		High	
	Extra Cost (£ million)	***Increase (%)***	***Extra Cost (£ million)***	***Increase (%)***	***Extra Cost (£ million)***	***Increase (%)***
Water treatment and distribution	0.11	0.2	0.35	0.7	1.58	3
Wastewater treatment	2.41	3	7.39	9	33.71	40
Sludge treatment and disposal	0.83	2	2.55	6	11.63	28

The increase in cost becomes more pronounced as we move from low to the high scenario. In all scenarios, the highest contribution to cost comes from the wastewater treatment, followed by sludge treatment and disposal, and finally water treatment and distribution (Table 5-5). The total cost that the water industry would incur in 2035 for the low to high scenario was estimated at between £179 and £223 million (Figure 5-3). The £223 million is the cost the water industry would incur in the case of a high uptake of FWDs, of which the total extra cost due to the use of FWDs was estimated at £47 million. This extra cost corresponded to an increase of 27% over the total cost of £223 million.

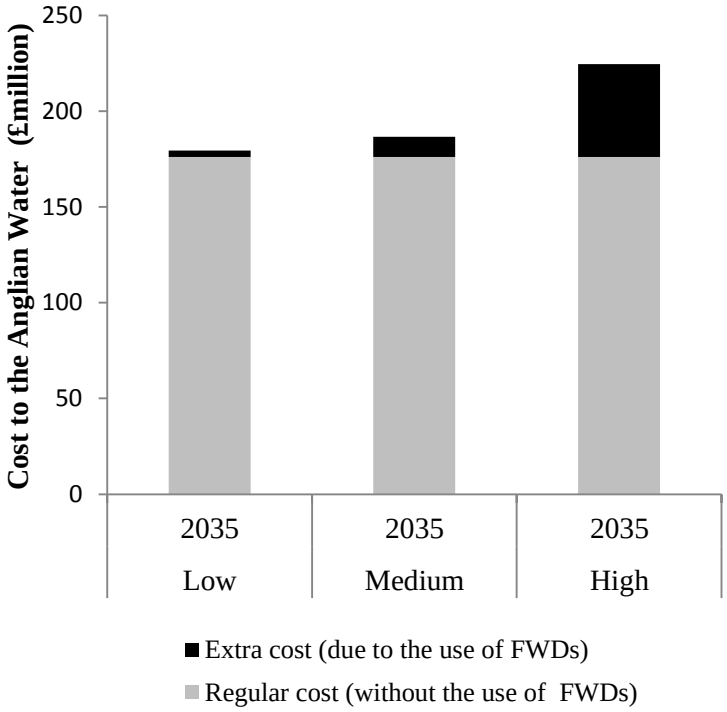


Figure 5-3 Total cost to the water industry in year 2035 for all scenarios

Accordingly, in the low and medium scenarios the total cost was estimated at £179 million and £186 million, of which £3.4 million and £10.3 million, respectively, were credited to the use of FWDs. These extra costs would have corresponded to an increase of 2% and 6%, for low and medium scenarios, respectively. Results are more analytically presented in Appendix B.3.

For LAs, the cost of collection and disposal of household residual waste in 2008 was estimated at approximately £118 million and £81 million, respectively, adding up to a total of £199 million. In that year, the amount of residual waste generated in the Anglian Region was estimated at 1533 kilo tonnes, but only 1506 kilo tonnes were collected by LAs. The remaining 27 kilo tonne was the amount of food waste diverted to the sewer through the use of FWDs. The total amount of food waste generated in the same year was approximately 563 kilo tonnes.

In 2035, the projected amount of household residual waste that would have been generated in the entire region was estimated at 2202 kilo tonnes, of which 809 kilo tonnes was food waste. Of the

36.7% of food waste in household residual waste, approximately 2%, 7% and 32% would have been diverted from kerbside collection to the sewer through the use of FWDs in low, medium and high scenarios, respectively (Figure 5-4).

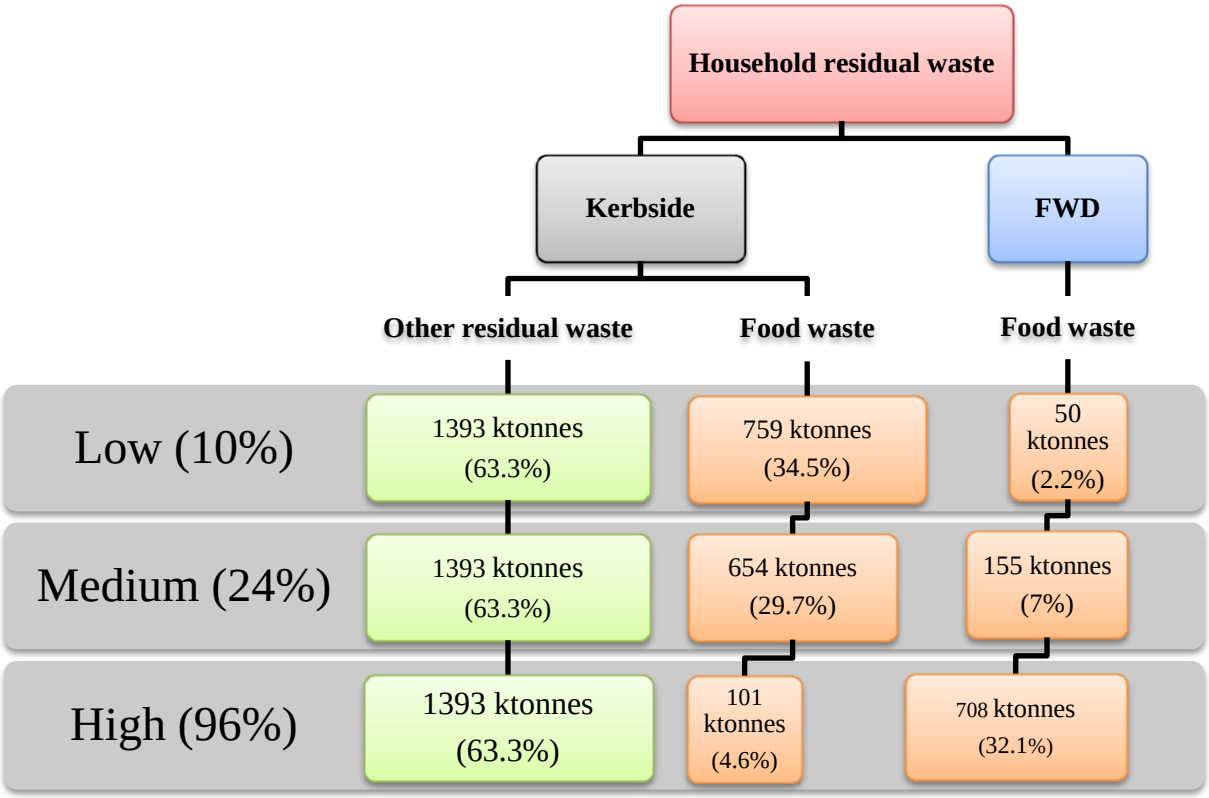


Figure 5-4 Household residual waste distribution in 2035

The cost of household waste collection in 2035 was estimated to be £169 million for all scenarios, whereas the cost of disposal varied between £80 and £116 million (Figure 5-5(a)). Savings to LAs resulting from the reduction of food waste fraction in household residual waste were estimated at £3 million, £8 million and £38 million, for low, medium and high scenarios, respectively, corresponding to an increase in savings of 1%, 3% and 13% for low, medium and high scenarios (Figure 5-5(a)).

As the food waste fraction in household residual waste decreases, a less frequent collection of household waste would be required. In such case, LAs could adopt a biweekly collection scheme in order to maximise their savings. The biweekly scheme could introduce flexibility in residual waste

collection frequency. Based on calculations, biweekly collection could be implemented by year 2023, potentially saving £71.8million to LAs. For every year since 2023 until 2035, savings to LAs from the biweekly collection increases by approximately £1million per year. In this analysis, the assumption that a biweekly scheme could only be applied to the high scenario, when food waste fraction is considerably reduced, was made. Based on this assumption, LAs could obtain significant additional savings in waste collection under the high scenario (Figure 5-5(a), (b)).

The cost of collection and disposal under the low and medium scenarios remains the same in both schemes with a noticeable decrease only in the high scenario. In that scenario the cost of waste collection was reduced to approximately £85 million, due to the adaptation of the biweekly scheme, presenting to LAs additional savings of approximately £84 million. These savings, added to the savings from household waste disposal, result in a total reduction in cost of £122 million in year 2035, which corresponded to a 43% increase in LAs savings. Results on these calculations are more analytically presented in Appendix B.5

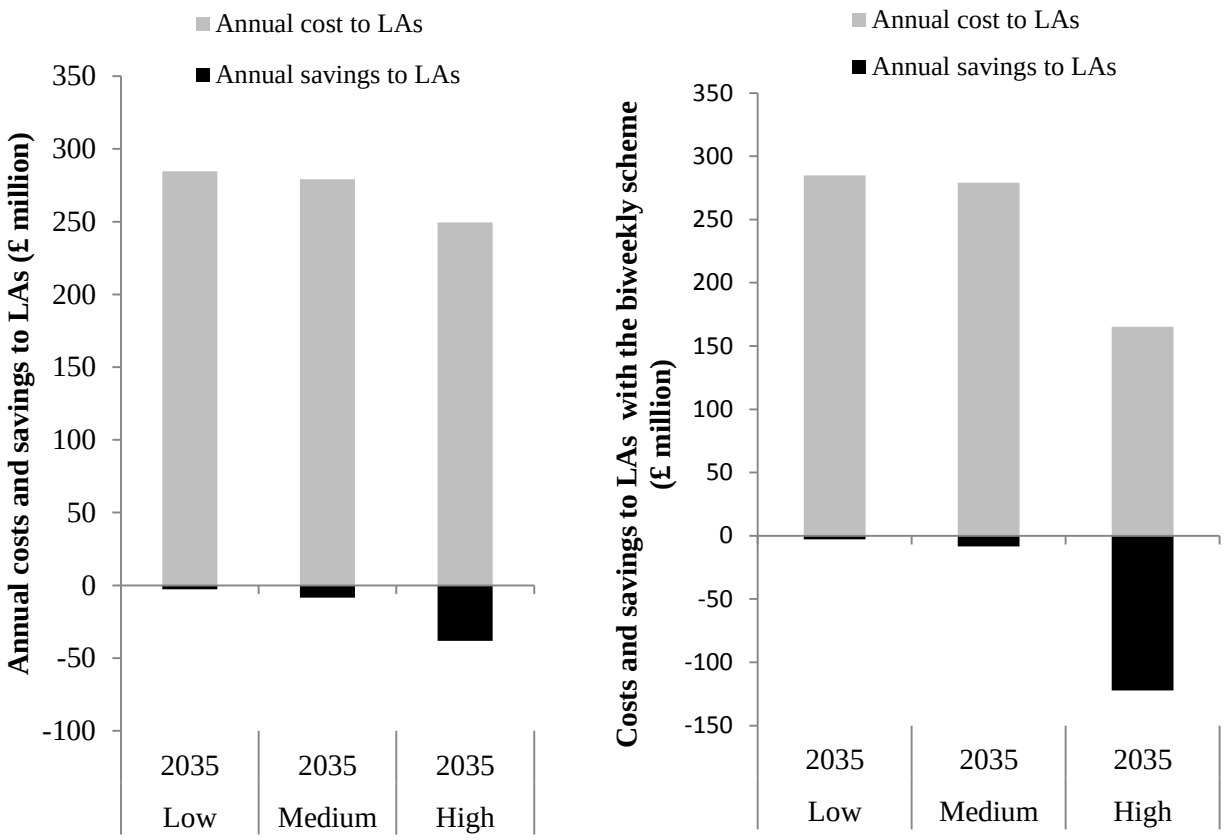


Figure 5-5 Cost and savings to LAs for household waste collection and disposal due to the use of FWDs based on (a) the regular collection regime and (b) the biweekly scheme

5.4 Discussion

The simple economic analysis undertaken in this Chapter demonstrates that the use of FWDs can be an effective alternative waste management option that diverts food waste from landfill. This diversion of food waste from the solid waste stream to the sewer could be beneficial for LAs. Results demonstrate that there are two options by which LAs could benefit from this diversion. The first option is based on the current waste collection regime in which, as the fraction of food waste in the kerbside household residual waste decreases, the overall cost of household waste disposal also decreases. This decrease in cost becomes more significant in the high scenario in which the FWD penetration rate reaches 96% in the entire region. However, in the high scenario, the fraction of food waste in household residual waste is so small that the kerbside collection of household residual waste could become less frequent, potentially leading to the adoption of a biweekly waste collection scheme. Adoption of such a scheme could maximise the savings on waste collection and disposal for LAs.

The cost to the water industry could be manifested by the increases in water consumption, and in tonnes of BOD and DS treated. Although increases in water consumption due to the use of FWDs were found to be insignificant, the associated cost was found to be substantial both directly and environmentally. Environmentally, because even minor additional water consumption can be detrimental in the Anglian Region - which has been classified as an area of 'serious' water stress - and it is not clear whether the existing water resources would be enough to deliver the additional water demand in the future (Anglian Water, 2008). Directly, because of the need for the increased water treatment and distribution, due to the use of FWDs, in the Anglian Region would be associated with an extra cost. This cost was found to increase recognisably, but not substantially, as the FWD penetration rate was increased. However, increases in cost, even if small, can be of significant importance when it comes to the aggregate cost. Increases in water consumption, not only incur a considerable cost to the water industry, but also to the consumer. Nevertheless, it should be acknowledged that FWD engineering and product design could have a positive impact on this, with new generation of FWDs taking less time to grind food waste and thus using less water, or even using

alternative sources of water such as recycled water after dish cleaning or other uses in a household to reduce water demand.

Increases in cost due to the additional BOD and the resulting DS treated accounted for the major part of the extra cost associated with the use of FWDs. Although not included in this analysis, one of the main concerns of the water industry is also the increase in cost associated with sewer blockages (Perriam, 2007; Water UK, 2009). The use of FWDs increases SS and fat, oil and grease (FOG) concentrations in the resulting wastewater, which are liable to settle in the sewer and cause sewer blockages. Nevertheless, the costs the use of FWDs could impose on the water industry are many and may become more pronounced as the FWD penetration rate increases. This is evidenced by the fact that in this Chapter, as the FWD penetration rate increases, the total cost to the water industry increases notably.

Besides cost implications, there is also the cost to the environment and to social welfare. Food waste disposal into the residual waste stream drives collection frequency and contributes to hygiene-related issues associated with rotting food waste (Chapter 4). Waste handling by trucks can be a major cause of carbon emissions, noise and nuisance and as thus, a reduction in the frequency of collection can reduce these negative environmental and social impacts. Thus, the use of FWDs can reduce the frequency of collection and hygienic impacts, offering a cleaner environment to household occupants. This has also been highlighted in an LCA study by Diggelman and Ham (2003), who evaluated FWDs to be an effective tool for food waste management in comparison to other options, both for wastewater and waste managers (Diggelman & Ham, 2003). Taking into account some evidence of the unwillingness of people to participate on food waste collection schemes, FWDs can be an attractive waste management option for both LAs and the water industry with regards to the environmental cost (WRAP, 2009). However, the economic pressures associated with their use can be an important barrier to the water industry.

Therefore, for a sustainable use of FWDs, policy measures will have to be taken that enable the establishment of a platform within which savings from waste collection and disposal of food waste

could be transferred to subsidise the water industry. However, based on the current waste collection scheme, savings made by LAs are not enough to compensate for the increased costs to the water industry. Consequently, while LAs would benefit from the diversion of food waste from the solid waste stream, the water industry would have to deal with the potential impacts of this diversion and its associated cost. In that case it is essential to either regulate the use of FWDs or to ban them completely in order to prevent their negative impacts to the water industry.

The adoption of the biweekly collection scheme would result in significant savings to LAs that could cover the increased cost of water and wastewater treatment. Switching to biweekly waste collection would depend on several factors including the environmental and health hazards that remaining food waste could pose to the surrounding environment.

5.5 Conclusions

Whereas the uptake of FWDs would lead to an additional cost to water industry operations, LAs could benefit from savings from the reduction in cost of waste collection and disposal. However, based on the current waste collection scheme, the simple economic analysis undertaken for the Anglian Region clearly demonstrates that savings to LAs are not significant enough to cover the increased cost to water industry operations. It also demonstrates that only a change in waste collection frequency would generate significant savings for LAs which could compensate for the increased costs to the water industry. This could be achieved by the adoption of a biweekly collection scheme in the region. The adoption of such a scheme would not only benefit the LAs economically but also socially and environmentally. The disposal of food waste into the sewer would minimise the hygiene-related issues as well as the environmental impacts associated with frequent collection and transportation. The main conclusion of this Chapter is that policy intervention is needed in the region, to regulate the use of FWDs either promoting them as an alternative to landfill to increase savings from waste management or banning them as a threat to wastewater treatment provision to reduce costs to the water industry.

6 THE POTENTIAL ANAEROBIC-AEROBIC TREATMENT OF INCREASED STRENGTH WASTEWATER AS A RESULT OF THE USE OF FOOD WASTE DISPOSAL UNITS (FWDs)

6.1 Introduction

Food waste disposal units (FWDs) are currently promoted as a food waste management option that grinds food waste with the addition of tap water and subsequently diverts it into the sewer for treatment with wastewater. This diversion can cause increases in water consumption, wastewater flow and characteristics and sludge generation rates (Chapter 5). Conventional aerobic processes for the treatment of this increased strength wastewater resulting from the use of FWDs, although effective in meeting the effluent discharge standards, are associated with a rising cost due to the increased energy consumption and sludge handling requirements (Chapter 5). This can be a major problem to wastewater treatment operations, especially in Europe, where regulations require energy consumption to be reduced and renewable energy generation to be increased, as a way to tackle climate change (Environment Agency, 2010b). As a result, new wastewater treatment alternatives are increasingly being investigated in an attempt to reduce energy requirements, carbon emissions and costs associated with aerobic treatment in addition to renewable energy generation.

The implementation of anaerobic technology for the treatment of wastewater is considered a mechanically and economically viable solution due to its ability to generate methane (CH_4), a renewable source of energy, and produce less sludge (Elmitwalli et al., 2002; Lettinga, 2006; Uemura & Harada, 2000). At the same time, this process offers many other benefits such as low construction, operational and maintenance costs, high organic removal efficiency and less land requirements (Lettinga, 1995; Lettinga, 2006; Mahmoud et al., 2004). In addition, it can bring about biodegradation of dangerous organic pollutants such as organotins and endocrine disrupting compounds that may be present in wastewater (Barnabé et al., 2009; Fent, 1996; Voulvoulis & Lester, 2006).

Anaerobic treatment of wastewater is extensively used in warmer climates, yet its application in colder areas is considered to be inadequate, especially at temperatures lower than 20^{o}C (Lettinga et

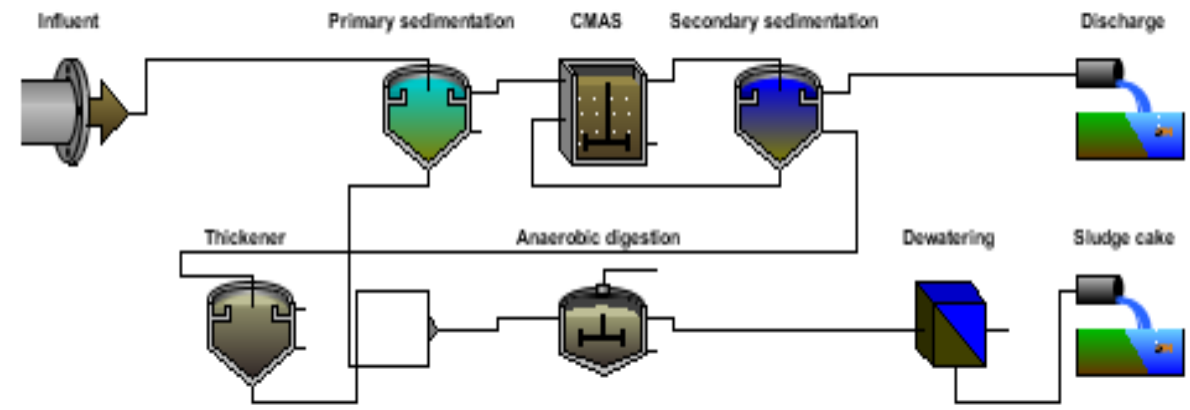
al., 1983). At low temperatures the hydrolysis rate becomes slow, which results in a decrease in biochemical oxygen demand (BOD) removal efficiency and promotes the accumulation of suspended solids (SS) (Bodik et al., 2000; Elmitwalli, 2001; Langenhoff & Stuckey, 2000; Lew et al., 2009; Nachaiyasit & Stuckey, 1995). However, the application of high rate anaerobic treatment in bioreactors, such as the upflow anaerobic sludge bed (UASB) reactor, the expanded granular sludge bed (EGSB), fluidised bed bioreactor, filter bioreactors and membrane bioreactor, has proved suitable for the treatment of wastewater at low temperatures (Lettinga et al., 1983; Singh & Viraraghavan, 2004). The concept of high rate anaerobic treatment lies on the capability of the bioreactor to retain a high concentration of biologically active aggregated biomass in the form of granules (Lew et al., 2004; Singh & Viraraghavan, 2004). This gives the bioreactors the ability to operate at high organic loading rates (OLRs) and solid retention times (SRTs), and low hydraulic retention times (HRTs) (Lew et al., 2004; Singh & Viraraghavan, 2004). However, the effluents of high rate treatment fail to comply with the discharge standards and, as such, an additional aerobic biological step to remove the residual carbon and nutrients is required (Chernicharo, 2006; Elmitwalli, 2003).

This Chapter aims to assess the potential of anaerobic-aerobic treatment of increased strength wastewater at low temperatures, as compared to the conventional aerobic treatment, using the UASB reactor as an example of a high rate bioreactor. To achieve that, modelling was incorporated to simulate the behaviour of both processes, and their inputs and outputs were compared.

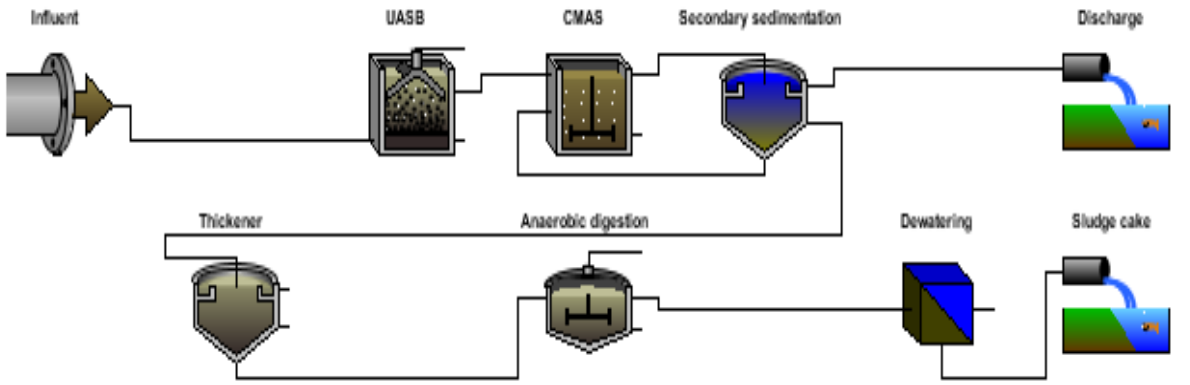
6.2 Methodology

The conventional aerobic treatment and the anaerobic-aerobic treatment flow sheets were prepared to simulate their behaviour (Figure 6-1(a), (b)). Wastewater in the conventional aerobic treatment flow sheet (Figure 6-1(a)) was assumed to be first treated in a primary sedimentation tank followed by an activated sludge process in the completely-mixed activated sludge (CMAS) reactor attached with a secondary sedimentation tank (Figure 6-1(a)). Wastewater from the secondary sedimentation tank was assumed to be discharged, whereas the waste activated sludge (WAS) produced was assumed to be condensed in a sludge thickener and then fed together with primary sludge produced in the primary sedimentation step into the anaerobic digester (Figure 6-1(a)). The digested sludge was then assumed to be drained in a dewatering unit and finally disposed of as a sludge cake (Figure 6-1(a)). The incorporation of the anaerobic treatment process, by replacing the primary sedimentation tank in the conventional aerobic treatment flow sheet with a UASB reactor as an example of high rate anaerobic bioreactor, led to the development of the anaerobic-aerobic treatment flow sheet (Figure 6-1(b)). In this flow sheet, WAS was the only sludge fed into the anaerobic digester on a daily basis, as sludge produced in the anaerobic treatment process remains within the reactor for approximately 40 days (Figure 6-1(b)).

Both flow-sheets (Figure 6-1(a), (b)) were built into the GPS-X software, developed by Hydromantis Inc., and the input data were modified in accordance to the wastewater characteristics assumed later on, in this Chapter. The carbon, nitrogen, phosphorus library (mantis2) was used for simulating both flow sheets (Hydromantis, 2012). Simulations were run at steady-state for 30 days at a temperature of $20^{o}C$, which is the lowest temperature at which GPS-X can give results on the anaerobic treatment performance with a relatively low uncertainty, without changing the process kinetics. To change the process kinetics and thus be in a position to run the simulations at temperatures lower than $20^{o}C$ would require a number of parameters that are currently not available in the literature.



(a)



(b)

Figure 6-1 Flow sheets of the (a) conventional aerobic treatment process, (b) anaerobic-aerobic treatment as developed in the GPS-X software

Modifications in the wastewater characteristics due to the use of FWDs were calculated based on the methodology developed in Chapter 5. The calculations on the wastewater characteristics in this Chapter were based on the typical UK design standards and on a population equivalent (PE) of 10,000 (Table 6-1).

Table 6-1 Wastewater characteristics as modified by the use of FWDs

	Units	Value	Reference
Flow rate	m^3/d	1335	Ofwat, 2011; Chapter 5
Biochemical oxygen demand (BOD) concentration	mg/l	650	Thomas, 2011; Chapter 5
Total kjeldahl nitrogen (TKN) concentration	mg/l	46.5	Thomas, 2011; Chapter 5
Total phosphorus (TP) concentration	mg/l	11.3	Thomas, 2011; Chapter 5

The per capita wastewater generation was estimated at 130 litres per day (Ofwat, 2011), with the use of FWDs contributing to an additional 3.51 litres per day (Chapter 5). BOD concentration per litre food waste ground was reported to be 8.37g, whereas nitrogen and phosphorus concentrations were reported to be 0.27g and 0.054g per litre food waste ground, respectively (Thomas, 2011). Based on these data, and using the typical UK wastewater characteristics, the concentrations of BOD, TKN and TP were calculated and adjusted in the influent characterisation of both flow sheets (Table 6-1). A more detailed description of the influent characterisation is presented in Appendix C.

Along with the changes in the influent characteristics, the design characteristics of the wastewater treatment objects were also adjusted for calibrating the model. For the UASB reactor, the volume was set up to achieve an HRT of 8 hours, which is the average HRT used for UASB reactors working at a temperature of 20^{o}C (Elmitwalli et al., 2002; Singh & Viraraghavan, 2003). The dimensions of the primary sedimentation tank, CMAS reactor, secondary sedimentation tank and anaerobic digester were set up achieving an HRT of 4h, 3.2h, 3.2h and 4d, respectively. In anaerobic digestion solids retention time (SRT) and HRT are the same. Details on the dimensions of the different objects can be found in Appendix C.

The mass flow in terms of BOD and TSS was used in both wastewater treatment flow sheets to identify the benefits and impacts of each. Energy consumption for the aeration process was estimated through the GPS-X software based on the amount of BOD (COD) treated. Other energy requirements, such as for pumping, mixing and heating the anaerobic digester were not included because of the inherent uncertainties related with these. However, an approach to compare the energy requirements

for anaerobic digestion is the amount of sludge which, among other factors, affects the energy required for heating-up the digester (Zupančič & Roš, 2003). The higher the amount of sludge digested is, the more energy needed to heat up the digester (Zupančič & Roš, 2003). The UASB reactor examined in this Chapter was operated at 20^{o}C, without heating. This was taken as a prerequisite for the whole assessment, mainly because a significant amount of energy would be consumed to heat up the reactor to a temperature in the optimal mesophilic range that would make its use less favourable than the conventional aerobic treatment (Lettinga et al., 2001). CH_4's density was calculated at the operational temperature (T^{o}C) using the following equation,

$$d_{CH_4,\, T^oC} = d_{CH_4,\, 0^oC} \times \left(\frac{273.15 + 0^oC}{273.15 + T^oC}\right)$$

where $d_{CH_4,\, 0^oC}$ is 0.7167g/l (Tchobanoglous & Burton, 2003). CH_4 has a calorific value of 50.1kJ/g and, using the conversion factor of 1kJ to 0.00278 kWh, the potential renewable energy generation was calculated (Tchobanoglous & Burton, 2003).

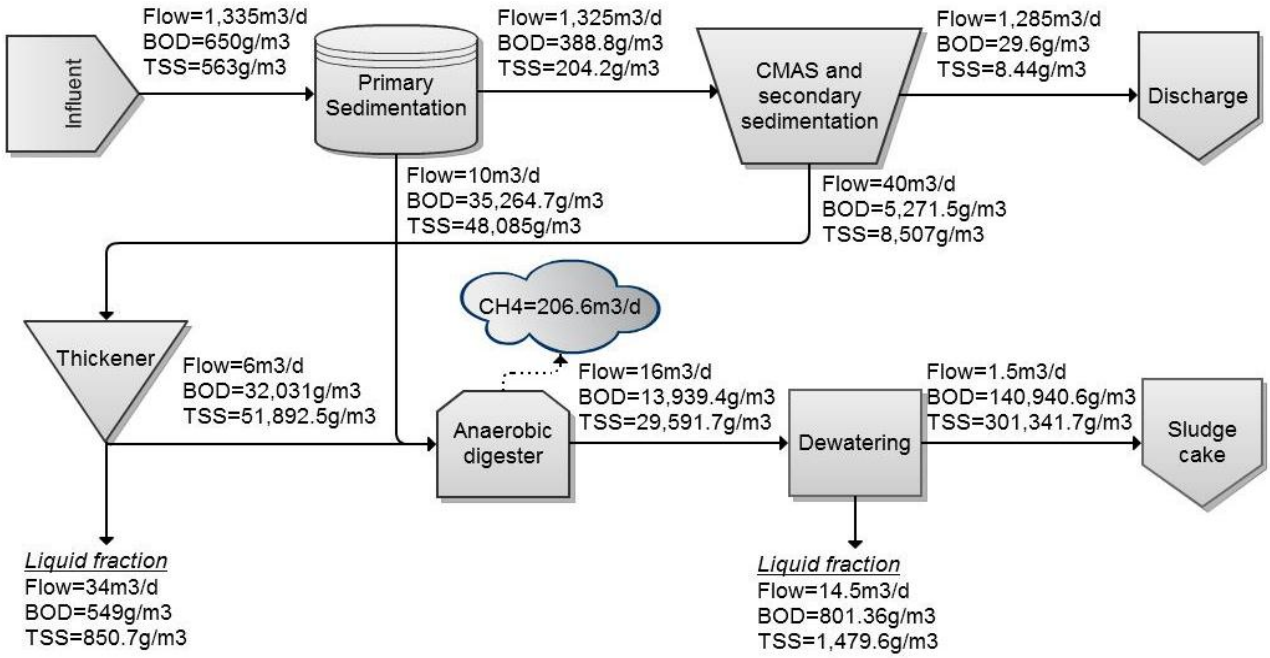
An additional component of this study was to examine the effect of temperature on the anaerobic treatment performance. To achieve this, the wastewater characteristics and the anaerobic reactor configuration (UASB) described in the previous part of this Chapter were used. The simulations were limited to the assessment of the anaerobic treatment performance only, and were run at temperatures between 35^{o}C and 20^{o}C.

6.3 Results

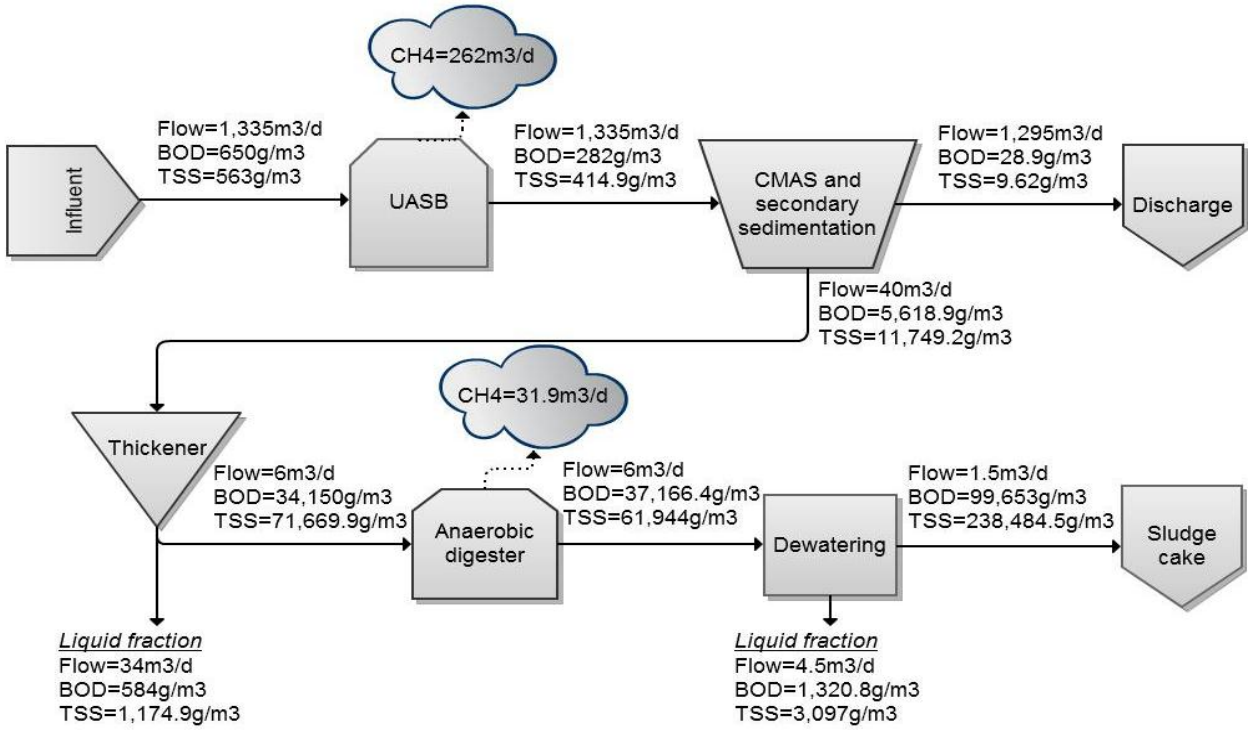
In the conventional aerobic treatment flow sheet (Figure 6-2(a)), wastewater first underwent primary sedimentation, where 63.7% and 38.2% (40%) of the influent TSS and BOD (COD) was removed, respectively. This led to the production of 480kg/d primary sludge that required further treatment. Effluent from primary sedimentation flowed into the CMAS for secondary treatment and secondary sedimentation for further BOD and TSS removal, resulting in the generation of 340.3kg/d (weight) of waste activated sludge (WAS). Thickened WAS was mixed with primary sludge and fed into an anaerobic digester. Anaerobic digestion of sludge produced up to 333.23m^3/d of biogas with a composition of 61.8% CH_4 and 37.8% CO_2. The stabilised sludge produced was dewatered and disposed of as a sludge cake of 452kg/d weight (Figure 6-2(a)).

In the anaerobic-aerobic treatment flow sheet, wastewater was first treated in a UASB reactor where 56% of BOD (58% COD) and 23% of TSS were removed (Figure 6-2(b)). These figures agree with those reported in the literature, where studies on the performance of UASB reactors for the treatment of municipal wastewater, demonstrated efficiencies of 50-70 (40-60%) or even 88% (84% COD) BOD (COD) removal at temperatures in the range of 12-18^{o}C and 20^{o}C, respectively (Alvarez & Lidén, 2009; Castillo et al., 1997; Elmitwalli et al., 1999; Seghezzo et al., 1998; Singh & Viraraghavan, 2003; Singh & Viraraghavan, 2004). In terms of TSS removal, some studies reported efficiencies higher than the figure of 23% as observed in this study. Nevertheless, it is known that UASB reactors may result in limited TSS removal as a result of a fraction of solids that flows through the reactor un-treated. In practice this can be improved by controlling the sludge bed height (Hydromantis, 2012). Biogas production in the UASB reactor was found to be 262m^3/d, of which 77% was CH_4 and 18% was CO_2. The remaining 5% was comprised of other gases, such as dinitrogen (N_2) and hydrogen (H_2). Effluent from the UASB reactor flowed into the CMAS and secondary sedimentation for further treatment, where 470kg/d of WAS was produced. Thickening of the WAS was followed by anaerobic digestion, where 32m^3/d of biogas were produced. This low biogas production can be explained by the low BOD and TSS content of WAS. The digested sludge was

dewatered and disposed of as a sludge cake of 358kg/d weight (Figure 6-2(b)). Details on the mass flow and operational variables of both treatment flow sheets can be found in Appendix C.



(a)



(b)

Figure 6-2 Mass-flow representations of the (a) conventional aerobic treatment, and (b) anaerobic-aerobic treatment flow sheets

The sludge cake produced and disposed in the anaerobic-aerobic treatment flow sheet was found to be 22% less than that of the conventional aerobic treatment flow sheet (Table 6-2). Also, in the anaerobic –aerobic treatment flow sheet total CH_4 captured was found to be approximately 33% more than in the conventional aerobic treatment flow sheet, leading to an analogous increase in the renewable energy generation (Table 6-2). Nonetheless, the conventional aerobic treatment flow sheet seems to hold the lead on energy consumption, which is 32.6% more than the energy required in the anaerobic-aerobic treatment flow sheet (Table 6-2). Also, approximately 46% more sludge would need to be digested in the conventional aerobic treatment flow sheet compared to that of the anaerobic-aerobic treatment (Table 6-2).

Table 6-2 Comparison for the main outputs of both wastewater treatment flow sheets

	Unit	**F1(a)**	**F1(b)**	**Difference**
Sludge cake	kg/d	452	352	100
Total CH_4 captured	m^3/d	206.6	294	87.4
Renewable energy generation	kWh/d	1,826.05	2,720	893.95
Blower energy consumption for CMAS	kWh/d	163.3	110	53.3
Sewage sludge digested	kg/d	792.2	430	362.2

F1(a): conventional aerobic treatment; F1(b): anaerobic-aerobic treatment

Based on these findings, the incorporation of anaerobic treatment processes in the current wastewater treatment operations and a temperature of 20^{o}C present several advantages to the water industry compared to the conventional aerobic treatment.

Notwithstanding these advantages, a decrease in temperature may cause a decrease in biogas production as a result of deterioration of the methanogenic activity in anaerobic reactors (Alvarez & Lidén, 2009; Elmitwalli, 2000; Elmitwalli, 2003; Lettinga et al., 2001; Mahmoud et al., 2004; Singh & Viraraghavan, 2003). This is demonstrated herein, where the simulation of anaerobic treatment at temperatures between 35^{o}C and 20^{o}C using the UASB reactor as an example of a high rate bioreactor showed a decrease in the removal efficiency, and, as a result, in biogas production (Table 6-3).

Additionally, while biogas production decreased with temperature, the CH_4 content of the biogas appeared to increase (Table 6-3).

Table 6-3 Temperature effect on the UASB reactor performance (HRT of 8h)

Temperature	**35°C**	**30°C**	**25°C**	**20°C**
BOD removal (%)	71.8	71.2	70.3	69.2
TSS removal (%)	29	28.4	26.9	26.3
Biogas production (m^3/d)	413.7	390.5	364.6	340.1
CH_4 (%)	72.4	73.8	75.7	77.1
CO_2 (%)	23.9	22.3	20	18.2

Previous studies explained that as temperature decreases, acidogenesis also decreases as a result of the reduced hydrolysis of complex organics, which lowers the proportion of CO_2 in the biogas, while increasing the proportion of CH_4. The production of acetate from CO_2 and H_2, and its subsequent reduction, was reported as another pathway to an increased proportion of CH_4 in the biogas (Massé et al., 2003). Additionally, the increased solubility of various gases found in the biogas in the wastewater at low temperatures also affects biogas composition. At low temperatures, CO_2 becomes more soluble in water than CH_4 (Singh & Viraraghavan, 2003). Even though many studies agree on CH_4 solubility during the anaerobic treatment of wastewater at low temperatures, the exact proportion of dissolved CH_4 has not as yet been verified (Agrawal et al., 1997; Lettinga et al., 1983; Singh & Viraraghavan, 2003). For example, Uemura and Harada (2000) reported that 40 and 65% of CH_4 produced was dissolved in the anaerobically-treated effluent at temperatures ranging from 25 to 13°C, whereas Yoda et al. (1985) (as reported in the study of Sanz and Polanco) reported that CH_4 collected at low temperatures represented 10-30% of the total CH_4 produced, suggesting that 70-90% of the total CH_4 produced was in the dissolved form (Sanz & Polanco-Fdz, 1990; Uemura & Harada, 2000). In this study, it was not possible to observe dissolved CH_4.

6.4 Discussion

Anaerobic treatment of municipal wastewater is changing from a developing to a well-established process, with full-scale applications in warm climates and in many developing countries. This is because anaerobic bioreactors are relatively simple to construct and operate, and offer important benefits such as low capital and operating costs, low energy consumption, low sludge production and high renewable energy generation. Notwithstanding these benefits, the application of anaerobic treatment processes in cold climates is as of yet limited. Models that have been developed to predict the performance of anaerobic treatment at low temperatures over the past years, show findings that vary widely, mainly because of the variation in the design, operational and kinetic parameters used in these models (Cakir & Stenstrom, 2005; Castillo et al., 1997; El-Fadel & Massoud, 2001; Mahmoud et al., 2004; Rebac et al., 1999; Zakkour et al., 2001). The use of UASB in this study is an example of a high rate bioreactor which may have particular weaknesses, such as TSS removal. Also, a limitation of this study is the incapability of the modelling software to simulate the anaerobic treatment of wastewater at low temperatures with certainty, and also its inadequacy to project the amount of gases dissolved in the wastewater.

The fact that low temperatures affect BOD removal and subsequently the overall performance of the anaerobic treatment is true. However, the extent to which BOD removal becomes the determining factor of the efficiency of anaerobic treatment of municipal wastewater at low temperatures is largely unknown. Also important for the efficiency of anaerobic treatment at low temperatures is the strength of the influent (Singh & Viraraghavan, 2004). Increased strength wastewater may lead to an increased reactor performance that would result in higher efficiency, even at low temperatures (Castillo et al., 1999; Lettinga et al., 2001). The incorporation of anaerobic technology for the treatment of increased strength wastewater as a result of the use of FWDs for diverting food waste from landfill to the sewer might therefore be a favourable solution to the water industry. This incorporation would be able to capture most of the energy potential in food waste which would otherwise be lost in the conventional aerobic treatment process. Findings of this study demonstrate that the anaerobic treatment of

increased strength wastewater as a result of the use of FWDs presents relative benefits over the conventional aerobic treatment.

In the future, where increased population and urbanisation are likely to increase the pressures associated with energy consumption and costs for the treatment of wastewater and sludge, the incorporation of anaerobic treatment processes in municipal wastewater treatment operations might be a justified solution. Alongside the increased energy and cost, the problem of increased sludge production and the imposition of stringent regulations on its disposal to agricultural land would be another hurdle to wastewater treatment operations, which is likely to intensify the preference for anaerobic-aerobic treatment processes for the treatment of increased strength wastewater. Furthermore, the increased production of biogas by the implementation of the anaerobic-aerobic treatment of the increased strength wastewater would help the water industry to increase its renewable energy generation.

Nevertheless, the greatest challenge to the implementation of anaerobic wastewater treatment at low temperatures is the potential solubility of CH_4 in the wastewater. CH_4 solubility even at the lowest proportions can be of a major importance when it comes to greenhouse gas (GHG) emissions. Therefore, while anaerobic treatment could assist the water industry to mitigate increased energy consumption, it could also create problems associated with carbon emissions, thereby possibly outweighing the benefits of increased renewable energy generation. Ultimately, this would require the capture and utilisation of CH_4 using technologies, such as air-stripping or CH_4 capturing systems, in order to ensure the implementation of anaerobic treatment as a sustainable solution to the water industry. The economics of such a solution need to be assessed to ensure that the balance of benefits is not shifted from anaerobic treatment to conventional aerobic treatment.

The existing prejudice against the performance of anaerobic technology at low temperatures is an obstacle to the implementation of anaerobic treatment of municipal wastewater under climates like that in the UK. The recent advancements in anaerobic technology that are currently available in the market, such as membrane bioreactors and fixed-film reactors, have been reported to achieve higher

efficiencies and be less sensitive to environmental variations (Chan et al., 2009). The concept of integrated treatment which combines the anaerobic and aerobic treatment in a single bioreactor is a breakthrough in the field of anaerobic technology, and it has been suggested that this will be a viable alternative to both the anaerobic-aerobic treatment and conventional aerobic treatment (Chan et al., 2009). As such, research and development should be directed towards new applications of anaerobic technology for the treatment of wastewater that would ensure both environmental and economic advantages to water industry operations, independent of the prevailing climatic conditions.

6.5 Conclusions

The application of anaerobic-aerobic treatment of increased strength wastewater as a result of FWD use presents relative benefits over conventional aerobic treatment, as it offers higher renewable energy generation potential and less sludge, and has a lower operating cost due to the lower energy consumption. While temperatures can lower the performance of anaerobic treatment, the magnitude at which this effect would prevent the incorporation of anaerobic treatment processes in municipal wastewater treatment operations is as of yet undefined. A limitation that could make anaerobic treatment unfavourable in cold climates is the potential solubility of CH_4 in wastewater, which bears the risk of creating pressures associated with carbon emissions. Collaboration between researchers and operators is essential for the selection of the best available anaerobic technology for the treatment of wastewater based on the prevailing environmental conditions of each area.

7 FOOD WASTE CO-DIGESTION WITH SEWAGE SLUDGE – REALISING ITS POTENTIAL IN THE UK

7.1 Introduction

Anaerobic digestion (AD) converts organic matter into biogas a renewable source of energy, and digestate, a valuable fertiliser and soil conditioner. AD is widely used for the stabilisation of sewage sludge that is a mixture of primary and biological sludge. This sludge is the major by-product of the wastewater treatment processes and as such AD is a well-established process in many wastewater treatment plants worldwide (Arnaiz et al., 2006; Bougrier et al., 2006; Parkin & Owen, 1986).

In recent years, the need for renewable energy generation and the need to divert biodegradable waste from landfill have pushed AD into the market for the treatment of biodegradable municipal waste. Out of the biodegradable fraction of municipal waste, food waste is the most challenging because of its high moisture content and readily biodegradable nature (Davidsson et al., 2008; Stroot et al., 2001). Although many options have been suggested for the management of food waste, including incineration, composting, the use of food waste disposal units (FWDs) and AD, AD has attracted more interest, particularly in policy making, due to its potential for energy generation (Banks et al., 2008; Gallert et al., 2003; Hartmann & Ahring, 2006; Komemoto et al., 2009; Zhang et al., 2005; R. Zhang et al., 2007). However, food waste can be highly variable depending on its source, and its composition might differ significantly from one place to another, which can be a hindrance to its successful and efficient digestion (R. Zhang et al., 2007). One approach aiming to overcome this limitation is the use of food waste as a co-substrate in sewage sludge digestion, a process known as co-digestion.

Co-digestion involves the digestion of two substrates together as a way to improve digestion efficiency and increase the energy output. It is a well-established process in Europe, with Germany and Scandinavia being the pioneers, having over 20 years of experience (Braun & Wellinger, 2009; Environment Agency, 2010b). The number of co-digestion plants is continuously increasing in many European countries and has become a standard practice in the agricultural sector (Braun & Wellinger,

2009). New plants for sewage sludge and food waste co-digestion have recently been built in Denmark, Germany and Switzerland (Braun & Wellinger, 2009). In the UK, the first co-digestion plant has been installed at Holsworthy, Devon for the treatment of animal manure and food waste (Environment Agency, 2010b). Nevertheless, the co-digestion of sewage sludge with food waste does not as of yet exist in the UK.

To change this, decisions have recently been taken in the UK to promote the use of co-digestion of food waste with sewage sludge, with the water industry being designated as a major potential implementer (Defra, 2009a; Defra, 2010; Defra, 2011a). This could be a major opportunity for the water industry to increase its renewable energy generation, thereby reducing its environmental footprint (Defra, 2011a; Kiely et al., 1997). At the same time, the application of co-digestion of food waste with sewage sludge will help the UK to meet the EU Landfill Directive targets (1999/31/EC), which require the reduction of biodegradable waste disposed of to landfill to 35% by 2020, based on the 1995 levels (European Council, 1999). Additionally, this process could contribute to the achievement of targets set by the Climate Change Act and the EU Renewable Energy Directive. The Climate Change Act requires the reduction of carbon emissions by 34% by 2020, based on the 1990 levels, whereas the EU Renewable Energy Directive (2009/28/EC) requires 15% of the UK's total energy from renewable sources by 2020 (Defra, 2011a; European Parliament and Council, 2009).

Although a number of studies have focused on examining the benefits of food waste co-digestion with sewage sludge, only a few have addressed its constraints as a waste management option. Therefore, this Chapter aims to assess the benefits and address the constraints that hinder its application for successful implementation in the UK. Industrial food waste was not considered for co-digestion with sewage sludge in this Chapter as it is rich in organic content, and can present additional challenges associated with compositional characteristics (Murto et al., 2004).

7.2 Characterisation of food waste

Food waste comprises the food wasted from residences, commercial establishments, institutions and other sources like factory cafeterias, canteens or lunchrooms, including unavoidable food waste from food preparation, food from poor preparation (under and over cooked) and food from refrigerator or freezer accidents (Chapter 4). It is highly variable in nature and its characteristics in terms of carbohydrates, lipids and proteins differ considerably based on its components (Belitz et al., 2009; Carucci et al., 2005; Christ et al., 2000; Lu et al., 2008; Straka et al., 2007). Meat, fish, eggs and cheese are rich in proteins and lipids, bread, potatoes and flour are rich in carbohydrates and fresh vegetables present an even composition of carbohydrate and protein content (Table 7-1).

Table 7-1 Composition of different components of food waste reported in the literature (on % dry weight/matter)

Waste	Carbohydrates	Proteins	Lipids	Reference
Meat and bone	<1	70-75	23-30	Straka et al., 2007
Egg (inc. shell)	2	35	32	Belitz et al., 2009
Fish meat	-	75.6	20.2	Lu et al., 2008
Whole milk powder	37.7	26.5	27.4	Belitz et al., 2009
Skim milk	53	36	-	Belitz et al., 2009
Cheese	-	25-35	20-45	Belitz et al., 2009
Legumes[1]	46-74	23-29	1.3-5	Belitz et al., 2009
Fresh vegetables[2]	27.1	26.9	1.4	Carucci et al, 2005
Fruit	83	4	2	Christ et al., 2000
Bread	84	14	2	Straka et al., 2007
Potato	99	1	<1	Straka et al., 2007
Rice	76.3	7.4	0.8-2.4	Belitz et al., 2009
Flour	70	12	<1	Straka et al., 2007

[1]excluding soybeans which are composed by 31.6% carbohydrates, 41% proteins and 19.6% lipids; [2]mainly spinach

Furthermore composition of these components can vary widely depending on their cultivar and origin. Generally, vegetables are rich in carbohydrates, but some families such as mushrooms, leafy (spinach) and inflorescence (broccoli) vegetables are rich in protein (Belitz et al., 2009). The same applies to fruits, the composition of which can be strongly influenced by the variety and ripeness (Belitz et al., 2009). Generally, all fruits are rich in carbohydrates, whereas protein and lipids are present in small amounts. The exception is nuts, in which protein content is higher than in fruits, and lipids represent approximately 50% of their composition (Belitz et al., 2009). Milk powder, skim milk and legumes present a wide distribution of the different characteristics, with slightly higher carbohydrate content. Overall, legumes have higher carbohydrate content relative to milk, which has higher lipid content (Table 7-1). The impact of the differing compositions of various food waste components is evidenced by numerous studies that have analysed the composition of food waste from different origins (Table 7-2) (Ferris et al., 1995; Hansen et al., 2007; la Cour Jansen, 2004; Redondas et al., 2012; Yan et al., 2011).

Table 7-2 Composition of food waste from different sources (on % dry weight/matter)

Food waste origin	Carbohydrates	Proteins	Lipids	Reference
Household	55[1]	17	13	La Cour Jansen, 2004
Household	60.7	14.4	14.4	Hansen et al., 2007
Urban (households, markets, restaurants)	78.2	16.9	4.9	Redondas et al., 2011
University dining hall	64.5	14.9	17.5	Ferris et al., 1995
Military facilities	56.79	17.5	22	Ferris et al., 1995
Institution restaurant	63.9	21.3	12.4	Yan et al., 2011

[1]calculated as the difference between lipids and protein plus the starch and sugars fraction

Whilst the proportion of carbohydrates, proteins and lipids gives an overall assessment of food waste, a comparison of different food wastes and their characteristics is commonly expressed in terms of available carbon (C), hydrogen (H), oxygen (O), nitrogen (N) and sulphur (S) (Table 7-3). The

elemental composition of carbohydrates, proteins and lipids indicates that an important quantity of carbon is available in food waste, which makes it highly degradable (Table 7-3) (Straka et al., 2007).

Table 7-3 Chemical composition of carbohydrates, proteins and lipids (on % weight) (adopted from Straka et al. 2007)

Name	C	H	O	N	S
Carbohydrates	40	7	53	-	-
Proteins	46	5	30	18.5	0.5
Lipids	76	12	12	-	-

Food waste degradability through AD has been extensively investigated over the previous years (Banks et al., 2011; Banks et al., 2008; Cho et al., 1995; Chynoweth et al., 1993; Climenhaga & Banks, 2008; Del Borghi, 1999; GarcIa et al., 2005; Gunaseelan, 1997; Hamzawi et al., 1998; Kim et al., 2003; Mata-Alvarez et al., 1992; Mata-Alvarez et al., 2000; Zhang et al., 2008; R. Zhang et al., 2007). In many studies, it was reported that the digestion of food waste was limited either due to the lack of sufficient nutrients such as nitrogen, phosphorus and to a smaller extent metals, or to volatile fatty acids ($VFAs$) accumulation resulting from its extremely high degradability (Climenhaga & Banks, 2008; Converti et al., 1997; Kim et al., 2004; Mata-Alvarez et al., 2000). In others, food waste was successfully digested but only after modifications were applied to the digester, either in the form of chemicals, the introduction of a pre-treatment step, the recirculation of nutrients through leachate or increases in temperature (Banks et al., 2008; Kim et al., 2006; Zhang et al., 2005; R. Zhang et al., 2007). However, most of these studies have been undertaken on an experimental basis, with the implementation of AD of food waste at full scale reported to be limited and problematic (Chen et al., 2010; Gallert et al., 2003; Zhang et al., 2005; Zhang et al., 2008).

In face of such limitations associated mainly with technical challenges and increased costs from the addition of chemicals, increases in temperature and/or proper digester management, increasing concerns regarding the reliability of food waste digestion have led to the investigation of its co-digestion with sewage sludge as an alternative. On the other hand, it is widely recognised that water

treatment and distribution and wastewater treatment processes are very energy intensive, and could only benefit from any initiatives that help in this area (Postnote, 2007; Shannon et al., 2008).

Sustainability requires that we do not address single issues in isolation, but to adopt a systems approach with integrated solutions. If the whole system is considered, a wastewater treatment plant is not only an "end-of-pipe" solution to clean wastewater but can serve as an energy producer. The interest in biogas is increasing and there is a large unused potential in maximizing the production of biogas, used for both heating and as a fuel in transport applications (AEBIOM, 2009; NSCA, 2006). The water and wastewater sector could benefit from any technological developments and breakthroughs in these areas. Co-digestion is one such breakthrough, and its implementation through an interdisciplinary, integrated and holistic approach could be developed to a real sustainable solution (Voulvoulis, in press).

7.3 Outcomes of food waste co-digestion with sewage sludge

7.3.1 Benefits

The co-digestion of food waste and sewage sludge has been extensively applied over the last years, with the benefits of improved CH_4 generation and dilution of toxic compounds being highlighted by many researchers (Callaghan et al., 1999; Edelmann, 2000a; Forster-Carneiro et al., 2008; Gómez et al., 2006; Hartmann et al., 2002; Heo et al., 2003; Kiely et al., 1997; Mata-Alvarez et al., 1992; Sosnowski et al., 2003). However, the parameters that will determine the extent of these benefits are of equal importance when it comes to increasing co-digestion performance.

The improvement in CH_4 yields from the addition of food waste to sewage sludge digestion has been widely reported in the literature (Table 7-4) (Edelmann, 2000a; Hamzawi, 1998; Hamzawi et al., 1998; Hartmann et al., 2002; Heo et al., 2003; Heo et al., 2004; Kim et al., 2003; Krupp et al., 2005; Kubler et al., 2000; la Cour Jansen, 2004; Lebrato et al., 1995; Mata-Alvarez et al., 2000).

Table 7-4 Comparison of CH_4 yields during the mesophilic (~35°C) digestion of sewage sludge, food waste and their mixture

Substrate	CH_4 yield ($l CH_4$ /gVS_{deg})	Reference
Sewage sludge	0.116	Kim et al., 2003
Sewage sludge (50%VS) + Food waste (50%VS)	0.215	Kim et al., 2003
Sewage sludge	0.318	Sosnowski et al., 2008
Food waste	0.234	Sosnowski et al., 2008
Sewage sludge (75%DS) + Food waste (25%DS)	0.439	Sosnowski et al., 2008
Sewage sludge	0.269	La Cour Jansen et al., 2004
Food waste	0.395	La Cour Jansen et al., 2004
Sewage sludge (80%VS) + Food waste (20%VS)	0.326	La Cour Jansen et al., 2004
WAS (90%VS) + Food waste (10%VS)	0.186	Heo et al., 2004
WAS (50%VS) + Food waste (50%VS)	0.321	Heo et al., 2004
WAS (10%VS) + Food waste (90%VS)	0.346	Heo et al., 2004

[VS] volatile solids; [was] waste activated sludge

CH_4 yields were found to increase notably by increasing the fraction of food waste in sewage sludge, making clear the value of the co-digestion process (Cecchi et al., 1988; Heo et al., 2003; Heo et al., 2004; Kim et al., 2003; la Cour Jansen, 2004; Lebrato et al., 1995; Purcell & Stentiford, 2006; Rintala & Järvinen, 1996; Zupancic et al., 2008). However, there is a limit to which the addition of food waste to sewage sludge can lead to an improvement in CH_4 yields. Altering the food waste fraction to either above or below the optimum mixing ratio can either limit the digestion performance in terms of overall CH_4 yields or result in a complete failure due to instability in the system (Heo et al., 2003; Heo et al., 2004; Kim et al., 2004; Lebrato et al., 1995; Purcell & Stentiford, 2006; R. Zhang et al., 2007).

It is also worth noting that the improved carbon to nitrogen (C:N) ratio and kinetic reactions achieved by the addition of food waste to sewage sludge were reported to be the prerequisites for an improving CH_4 yield (Dohányos et al., 2004; Mottet et al., 2009).

- ***Improved C:N ratio***

The balance between carbon and nutrients, expressed as the C:N ratio, is an important parameter for a stable digestion process. While some researchers have suggested that the C:N ratio for optimum digestion performance is in the range of 20 to 30, many have demonstrated that digestion can be successfully performed under a wider range of C:N ratios (Demirekler & Anderson, 1998; Hartmann et al., 2002; Heo et al., 2003; Heo et al., 2004; Kim et al., 2003; Sosnowski et al., 2003; Zhang et al., 2008). However, not all C:N ratios are suitable for digestion. A C:N ratio over 30 is usually a cause for instability, mainly due to nutrient deficiency which severely affects microorganism activity resulting in lower substrate removal and CH_4 production (Kayhanian & Hardy, 1994). On the other hand, C:N ratios lower than 6 negatively affect the digestion process, mostly due to low carbon availability in combination with high ammonia (NH_3) generation which at relatively high concentrations is toxic to anaerobic microorganisms (Kayhanian & Hardy, 1994).

In general, sewage sludge is characterised by a low C:N ratio ranging from 6 to 9. As such, the addition of food waste which is characterised by a high C:N ratio can improve the C:N ratio of sewage sludge (Heo et al., 2003; Kim et al., 2003; Stroot et al., 2001). Noteworthy improvements in the C:N ratio arising from the addition of food waste in sewage sludge digestion have been widely documented (Table 7-5). These improvements were reported in the range of 6-15.4 (Table 7-5).

Table 7-5 The C:N ratio of sewage sludge and food waste and its improvement through their mixing

C:N ratio			Reference
Sewage sludge	**Food waste**	**Mixture[1] (sewage sludge:food waste)**	
6.0[2]	16.1	7.0 (90:10)	Heo et al., 2003
6.0[2]	16.1	7.8 (70:30)	Heo et al., 2003
6.0[2]	16.1	8.8 (50:50)	Heo et al., 2003
7.2	20.8	9.9 (80:20)	Kim et al., 2003
7.2	20.8	12.7 (60:40)	Kim et al., 2003
7.2	20.8	15.4 (40:60)	Kim et al., 2003
9.3	24.5	14.2 (75:25)[3]	Sosnowski et al., 2003
5.6[2]	16.2	6.0 (90:10)	Heo et al., 2004
5.6[2]	16.2	7.0 (70:30)	Heo et al., 2004
5.6[2]	16.2	8.9 (50:50)	Heo et al., 2004
5.6[2]	16.2	11.0 (30:70	Heo et al., 2004
5.6[2]	16.2	14.7 (10:90)	Heo et al., 2004
7.2	13.8	8.5 (60:40)	Kim et al., 2004

[1]on VS basis; [2]waste activated sludge; [3]on dry solids basis

Improvements in the C:N of sewage sludge digestion with food waste were found to be proportional to the fraction of food waste added. Hence increases in the C:N of the sewage sludge and food waste mixture were attributed to increases in the food waste fraction (Table 7-5) (Heo et al., 2003; Heo et al., 2004; Kim et al., 2003; Sosnowski et al., 2003; R. Zhang et al., 2007). It must be noted, that

improved C:N ratio can also be observed by changes in the food waste characteristics. Food waste rich in proteins is likely to improve the C:N ratio as these can provide important nutrients in the substrate, compared to food waste that is rich in carbohydrates (Table 7-3). Nevertheless, the improved C:N ratio indicates that the addition of food waste to sewage sludge digestion plays an important role in providing essential carbon to the system (Kim et al., 2003). Carbon availability is essential for the improvement of digestion performance, mainly because of its influence on the kinetics of the process (Fernández et al., 2010).

- ***Improved kinetic reactions***

Hydrolysis has been reported as the rate-limiting step in the AD of sewage sludge (Bolzonella et al., 2003a; Dohányos et al., 2004; Mottet et al., 2010). This is mainly because of the composition of sewage sludge, which is rich in proteins (30%DM) (European Commission, 2001a), which have a lower hydrolysis potential than carbohydrates and lipids (Table 7-6) (Arnaiz et al., 2006; Bougrier et al., 2005; Bougrier et al., 2007; Eskicioglu et al., 2008; Lafitte-Trouqué & Forster, 2002; Lin et al., 1997; P. Zhang et al., 2007). The low hydrolytic potential of proteins has been reported in many studies (Christ et al., 2000; Gavala et al., 2003a; Pavlostathis & Giraldo-Gomez, 1991). In some of these studies it has also been noted that the hydrolysis rate of proteins depends on their solubility potential, pH and the origin of the anaerobic culture (Gavala et al., 2003b). Therefore, the addition of easily degradable matter in the form of food waste has been reported to improve the degradation efficiency and accelerate the hydrolysis of sewage sludge (Bolzonella et al., 2003a; Dohányos et al., 2004; Kim et al., 2003; Mottet et al., 2010). The higher the amount of food waste added to sewage sludge, the faster the growth of anaerobic microorganisms. An acceleration of the microorganism growth rate results in a higher hydrolytic rate (k) and, as a result, in a higher acidification and methanogenesis potential (Table 7-6) (Pavlostathis & Giraldo-Gomez, 1991; Ponsá et al., 2008).

Table 7-6 Improvement of the kinetic constants of the first-order rate of hydrolysis, by the addition of food waste in sewage sludge and the increase of temperature

Substrate	k_h (d^{-1})	Temperature (^oC)	Reference

Sewage sludge	0.15	35	Kim et al., 2003
Sewage sludge (80%VS[1]) + Food waste(20%VS)	0.18	35	Kim et al., 2003
Sewage sludge (50%VS) + Food waste (50%VS)	0.22	35	Kim et al., 2003
Sewage sludge	0.21	55	Kim et al., 2003
Sewage sludge (80%VS) + Food waste(20%VS)	0.22	55	Kim et al., 2003
Sewage sludge (50%VS) + Food waste (50%VS)	0.35	55	Kim et al., 2003
Sewage sludge	0.11	35	Sosnowski et al., 2008
Food waste	0.3	35	Sosnowski et al., 2008
Sewage sludge (75%DS[2]) + Food waste (25%DS)	0.17	35	Sosnowski et al., 2008

[1]volatile solids; [2]dry solids

As such, the addition of food waste to sewage sludge digestion improves the kinetic coefficients of hydrolysis and therefore the overall digestion performance (Table 7-6). In addition, an increase in temperature can result in even better kinetics (Kim et al., 2003; Sosnowski et al., 2008). The effect of temperature on the hydrolysis kinetics originates from the combined effect of hydrolytic enzymes, bacterial growth and solubility of the substrate on improved activity (Angelidaki & Sanders, 2004).

Additional benefits of food waste digestion with sewage sludge include the revenue from the increased renewable energy generated and the dilution of some potentially toxic compounds such as heavy and light metal ions and organic compounds. The latter is particularly important where sewage sludge contains high amount of metals.

Light metal ions, heavy metals and organics are some other toxicants commonly found in sewage sludge that may affect the digestion process (Chen et al., 2008). Co-digestion of food waste with sewage sludge can improve the digestion performance by reducing the negative effect of these toxic compounds (Murto et al., 2004; Purcell & Stentiford, 2006; Stroot et al., 2001). The efficient dilution of inhibitory substances would also increase the degradation of sewage sludge and increase CH_4 yields (Bolzonella et al., 2003a; Chen et al., 2008; Dohányos et al., 2004).

Increased renewable energy generation in the form of CH_4 offers the advantage of increased income. By generating additional renewable energy, the water industry would be able to increase its revenue by cutting the electricity bills and also by selling the surplus energy.

7.3.2 Constraints

Food waste co-digestion with sewage sludge can be associated with a number of constraints, with the variability of food waste composition often reported here, because of the disturbances it can cause to the anaerobic system.

Co-digestion of food waste with sewage sludge can be limited due to the high variability of food waste. A stable digestion performance depends on the composition of food waste added to sewage sludge, which if changed can cause instability in the anaerobic population and consequently in the digestion process. This is because microorganisms are acclimatised in a specific mixture and changes in this mixture may result in changes in the process reactions. In addition, CH_4 yields may also vary due to seasonal variations in food waste (Buffiere et al., 2006; Saint-Joly et al., 2000).

The concentration of light metal ions and biodegradation intermediates plays an important role in smooth process performance, as they can be the most potent causes of toxicity in AD. A compound can be described as toxic or inhibitory when it causes an adverse change in the microbial population or halts bacterial growth (Chen et al., 2008). By increasing the fraction of food waste, the risk of increasing the concentration of light metal ions and/or biodegradation intermediates at levels that can be toxic to the anaerobic population becomes greater.

Light metals ions, also known as cations, are present in many types of food waste and can be a cause of toxicity in the digestion process if present at high levels. High concentration of cations, such as potassium (K^+), sodium (Na^+), calcium (Ca^{2+}) and magnesium (Mg^{2+}), in the digester can cause bacterial cells to dehydrate due to osmotic pressure (Carucci et al., 2005; Chen et al., 2008). Co-digestion of sewage sludge with food waste rich in vegetables is likely to show an increase in the K^+ content that may inhibit the digestion process (Edelmann, 2000a). Sodium inhibition is also likely to

occur since food waste is a source of Na^+ (Carucci et al., 2005; Edelmann, 2000a). However, McCarty (1964) has reported that Na^+ concentrations in the range of 100-200mg/l are essential for the growth of microorganisms (Carucci et al., 2005; Edelmann, 2000a; MacFarlane et al., 1983; McCarty, 1964). Rittman and McCarty (2001) reported that light metals ions can be present in the digester at levels that are moderately inhibitory or strongly inhibitory (Table 7-7) (Rittman & McCarty, 2001).

Table 7-7 Inhibitory concentrations of cations to the AD process (adopted from Rittman and McCarty, 2001)

Cations	Moderately inhibitory (mg/l)	Strongly inhibitory (mg/l)
Na^+	3500-5500	8000
K^+	2500-4500	12000
Ca^{2+}	2500-4500	8000
Mg^{2+}	1000-1500	3000

Moderately inhibitory: prevents the right performance of the digestion process, but can be restored by the addition of antagonistic cations; S*trongly inhibitory*: causes toxicity in the system that leads to process failure

$VFAs$, long chain fatty acids ($LCFAs$) and ammonia (NH_3) are the main biodegradation intermediates of AD. The accumulation of these intermediates, beyond certain levels, in the anaerobic digester can be a cause of toxicity in the co-digestion process (Angelidaki & Ahring, 1992).

The addition of food waste to sewage sludge results in an initial increase in $VFAs$ concentration due to the rapid acidification of soluble organic compounds found in food waste (Heo et al., 2003; Kim et al., 2003; Sosnowski et al., 2008). $VFAs$ are subsequently decreased as a result of their uptake by the anaerobic microorganisms (Carucci et al., 2005; Converti et al., 1997; Demirekler & Anderson, 1998; Habiba et al., 2009; Kim et al., 2003; Sosnowski et al., 2008). However, when the production rate surpasses the uptake rate the excessive levels of $VFAs$ produced are accumulated in the digester. This accumulation can cause acidity in the digester which if not restored can lead the pH to drop to such a level that stops the digestion process from occurring (Cho et al., 1995; Gerardi, 2003; Hartmann et al.,

2004; Heo et al., 2003; Kim et al., 2003). Levels at which $VFAs$ become toxic have not been documented, however, it was reported that $VFAs$ can be present in the digester at concentrations up to 6000mg/l without being toxic, provided that the pH is maintained in the optimal range (Parkin & Owen, 1986).

A high lipidic content in food waste may also affect the co-digestion process with sewage sludge, due to the excessive production of $LCFAs$. $LCFAs$ have been shown to be toxic to the anaerobic population, and the higher their concentration in the digester, the more toxic their effect is (Alves et al., 2001; Carucci et al., 2005; Chen et al., 2008; Cirne et al., 2007; Hanaki et al., 1981). The levels at which $LCFAs$ can be toxic vary widely depending on the predominant form of acid present in the digester (Carucci et al., 2005; Chen et al., 2008).

The degradation of protein-rich mixtures of food waste and sewage sludge is the primary cause of NH_3 production and accumulation (Gallert et al., 1998; Gallert & Winter, 1997; Kayhanian, 1994). NH_3 and ammonium ion (NH_4^+) concentrations both exist in anaerobic digesters. NH_4^+ can inhibit the activity of methanogens and hence, CH_4 production. However, NH_3 was reported to be more inhibitory than NH_4^+ because of its capability to penetrate through cell membranes (Chen et al., 2008; El Hadj et al., 2009; Heo et al., 2004; Rittman & McCarty, 2001). However, there is an uncertainty as to the range at which NH_3 concentrations can be inhibitive (Chen et al., 2008). This uncertainty is mainly because of the differences in operational conditions, such as alkalinity, temperature, substrate composition and acclimation period (Chen et al., 2008; Gerardi, 2003). A number of studies reported that ammonia-nitrogen ($NH_3 - N$) levels of 1500mg/l can decrease methanogenic activity, but further increases in those levels can result in the toxicity of the anaerobic environment (Chen et al., 2008; Gerardi, 2003; Heo et al., 2003). In a study of McCarty, it was reported that concentrations between 1500 and 3000mg/l were inhibitory at pH levels above 7.4, and any excess beyond 3000mg/l was toxic regardless of the pH (Parkin & Owen, 1986). However, Hartmann and Ahring (2005) showed that $NH_3 - N$ inhibition may begin at concentrations above 650mg/l, and that the likelihood of toxicity depends on the magnitude of NH_3 increase, whereas Angelidaki et al., (2004) showed that $NH_3 - N$ inhibition may begin at concentrations above 4000mg/l. On the other hand, if the NH_4^+

concentration builds up in the digester at levels of about 3000mg/l, it causes inhibition which cannot be relieved and causes toxicity in the digester (Heo et al., 2004; Rittman & McCarty, 2001). Finally, mixtures with a higher proportion of food waste than sewage sludge have a limited possibility for NH_3 inhibition, mainly because of the higher carbon availability (Heo et al., 2004).

Operational constraints associated with food waste handling are extremely important. Impurities found in food waste such as plastics, metals, glass and other packaging parts are likely to cause tremendous technical problems in the wastewater treatment line and co-digestion performance (Braun, 2002; De Baere, 2000; Hartmann et al., 2000; Hartmann et al., 2004; Kubler et al., 2000; Lebrato et al., 1995). Plastics in the form of plastic bags can be wrapped around the stirring equipment in storage and reactor tanks, wear out the pumps and form a top layer in the reactors. Furthermore, plastic contamination in the form of phthalates can change the quality of the digestate and make it unacceptable for application to agricultural land (Hartmann et al., 2000; Hartmann et al., 2004). Metals are toxic for the bacterial biomass whereas, lignocellulosic material such as wood and paper are digested slowly and are troublesome (Lebrato et al., 1995). Mechanical problems such as clogging in the conveyor line may also arise from metal contamination (CADDET, 1998). These can lead to increases in operational costs, and also to the loss of a relatively important fraction of food waste during the pre-treatment process. This loss can be associated with a reduction to the benefits for the water industry, as less biogas will be produced per tonne of waste delivered. Therefore, it is of critical importance that food waste is free of impurities and exogenous material prior to co-digestion with sewage sludge.

Overloading the digesters must also be avoided in order for the process to operate successfully. Overloading episodes can cause a decrease in CH_4 yields, and even in a failure of the whole process, while blocking of the pipes and foaming incidents in the digester are also possible (Ganidi et al., 2009; Heo et al., 2004; Purcell & Stentiford, 2006; Zhang et al., 2008). Increased foaming, blocking of pipes and insufficient mixing of the substrates resulting from digester overloading can ultimately lead to a loss in biogas production (Stroot et al., 2001).

In the UK, the current operational and regulatory framework makes anaerobic co-digestion of food waste with sewage sludge a rather complicated matter. Sewage sludge and food waste are both covered by different regulatory regimes. When food waste is co-digested with sewage sludge, regulation becomes more complex and unclear, with the process standing between two sets of regulations (Defra, 2011a). Waste management license requirements, the quality of the co-digestate and renewable energy generation credits are some of the regulatory constraints and uncertainties that currently prevent the adaptation of co-digestion of sewage sludge with food waste by the water industry. Anaerobic digestion schemes were originally under the EU Pollution Prevention and Control (PPC) regulations, also known as IPPC permitting regulations. This IPPC permit aimed to ensure that energy is used efficiently, emissions minimised, accidents prevented and improvements made. To operate an anaerobic digestion facility, where sewage sludge was digested with wastes of household, commercial or industrial origin, a Waste Management Licence was required under the Environmental Protection Act 1990 (CIWEM, 2011; Monnet, 2003). This licence was required for the delivery, treatment and disposal of this waste. Subsequently, in order to simplify regulations, both the IPPC permit and Waste Management Licence were merged under the term Environmental Permitting Regulations. The scope of the Environmental Permitting Regulations is to control the water industry in terms of the use of sewage sludge and food waste, the use of the resulting biogas, and the storage and spreading of the co-digestate in a way that is safe to the environment and human health (Defra, 2011a; Environment Agency, 2010a). It also regulates the size of the combined heat and power (CHP) plants when wastes other than sewage sludge would be added to water companies' anaerobic digesters (Ohandja et al., 2010). The water industry has to apply for this permit each year, thus incurring an extra cost.

The processing of food waste that contains or has been contaminated by meat and any other animal materials falls into the Animal By-Products Regulations (ABPR). The ABPR defines three categories of ABP, with category 3 being the least harmful. This category includes food waste originating from households, restaurants and catering facilities (European Parliament and Council, 2002; Legislation, 2011). As such, the water industry must have an ABP permit to be able to process food waste. To

meet the requirements set by the ABPR, food waste pre-treatment is required firstly to ensure the removal of packaging material and other impurities that may be present in food waste and secondly to pasteurise the food waste before being added to the digester (Defra, 2009a). The aim of the ABPR is to sanitise food waste and prevent pathogen transfer. The use of co-digestate may also be covered by the ABPR although this is not yet clearly defined (Defra, 2010).

The co-digestate produced from the digestion of mechanical-segregated food waste with sewage sludge is considered a waste digestate in the revised Waste Framework Directive (rWFD) and thus, it cannot be applied directly to land (European Commission, 2011a). A potentially expensive environmental permit might be required for its disposal to land, though this is currently uncertain. Regulations that govern the production and use of such co-digestate are currently unclear and yet to be defined (CIWEM, 2011; Defra, 2011a). That would leave the water industry to deal with a huge amount of co-digestate if not otherwise managed.

Economic complications related to the planning and operation of the co-digestion process can be a significant barrier to its adaptation. The building of new digesters, the upgrade of existing ones, and the installation of facilities for the delivery, pre-treatment and storage of food waste (Edelmann, 2000a; Edelmann, 2000b; Hartmann et al., 2004; Krupp et al., 2005), required for co-digestion, cannot currently be easily provided due to economic barriers and policy restrictions. These regulations impose a level of complexity which exceeds the potential of co-digestion, and initiatives to implement this process have been severely hindered. This is not surprising as individual approaches in waste management are challenging and bound to become unsustainable.

7.4 Discussion

Taking into account the relationship between water provision, energy security and resource efficiency, and implications both in terms of availability and demand, water and environmental challenges often prove complex to address. Nonetheless these links also provide opportunities for synergies that could deliver real benefits when cross-sectorial solutions are applied (UNEP, 2009; Voulvoulis, in press). Water management is the realm of complex, dynamic systems, interconnected and interdependent, and therefore it is important to understand them as a 'whole' and in relation to other sectors (Balasubramaniam & Voulvoulis, 2005). By-products from sewage treatment in combination with organic solid waste, such as food waste, can provide a valuable source of energy if managed properly and utilised effectively. In this way, waste can be seen as a raw material that can be turned into a resource rather than simply be discarded. As such, the co-digestion of food waste with sewage sludge could become a strategic and cross-sectorial solution, if carefully applied, with the potential to convey beneficial synergies for the water industry and LAs or contractors responsible for food waste management in the UK.

Such synergies could deliver economic benefits from the additional renewable energy generated and its associated incentives, and savings on costs for the infrastructure required for the exclusive digestion of food waste. The necessary infrastructure and resources could be provided through synergies and cooperation between water and waste authorities. The amortisation of the required investment, given the revenue from the increased renewable energy generation and the incentives for compliance with the Climate Change Act and the EU Renewable Energy Generation, might occur sooner than expected (Mata-Alvarez et al., 2011).

The need to simplify regulations, and make the quality criteria of both co-digestate and renewable energy more clear, is almost a prerequisite for moving things forward. Currently, the digestate produced from sewage sludge digestion is covered by the Sludge Regulations and the Safe Sludge Matrix which govern its application to land if appropriately managed, whereas the digestate produced from source-segregated food waste is covered by the Anaerobic Digestate quality protocol and the

Publicly Available Specification (PAS) 110, which can determine if the digestate is no longer classified as a waste and therefore, can be applied to agricultural land. However, the co-digestate produced from sewage sludge digestion with food waste no longer falls into the sludge regulations, the anaerobic digestate quality protocol or the PAS, but is still considered a waste according to the rWFD, and the existing regulations concerning its production do not as of yet clearly define which quality criteria should be considered for its proper management. That leaves the full marketing opportunities of the co-digestate unexploited, with the water industry left to deal with it. Similarly, regulations that govern renewable energy generation depend on the process selected for biogas exploitation, and financial incentives differ accordingly. However, when it comes to the renewable energy generated from the co-digestion of food waste with sewage sludge regulations on the allocated incentives become problematic (Defra, 2011a). An example is the Renewable Obligation Certificates (ROCs) which are provided to the water industry for its renewable energy generation potential. While, renewable energy generation from sewage sludge digestion is eligible to receive 0.5 ROCs per MWh of electricity produced, the renewable energy generation from the digestion of biodegradable waste such as food waste is eligible for 2ROCs per MWh. When it comes to the renewable energy generation from the co-digestion of food waste with sewage sludge, the allocation of ROCs thus becomes a problem.

Such simplifications would also assist the UK to meet its targets for the reduction of landfilled waste, greenhouse gas (GHG) emissions and electricity generation from fossil fuels. It would also provide the basis for the continued growth of anaerobic digestion as a waste management option and a renewable energy generation source. Additionally, it would open the doors to a competitive market for digestate and energy supply within and outside the country.

The agricultural sector could play a key role in providing a market for the co-digestate, but it is essential to develop a coherent policy that could help stimulate the development of that market. Land owners and farmers would pay for the co-digestate either for land application or as a result of diversification into farm energy production through the co-digestion of bio-waste. Furthermore, the development of a pricing structure for renewable energy would lead to the growth of a biogas market

that would promote its injection into the national grid and its use as a transport fuel. This in turn would increase the biogas demand as the need for renewable energy generation would increase in the future.

Finally regulations that govern the delivery and pre-treatment of food waste will need to be revised. The potentially expensive permit that will be required for the delivery and pre-treatment of food waste becomes an unnecessary barrier to the implementation of the co-digestion process within the water industry. This barrier must be removed and financial transparency between the relevant stakeholders must be applied in order to provide incentives to the water industry as a way to promote sustainability through the use of the co-digestion process.

Additionally, LAs or contractors responsible for food waste management could facilitate the establishment of source-segregating food waste collection systems or the selective removal of impurities from the collected food waste prior to co-digestion. Source-segregated food waste requires less pre-treatment which results in less food waste losses. The high recovery of food waste would bring about additional benefits due to the increased biogas production. Nonetheless, source-segregation by the use of plastic bags bears the risk of food waste contamination that may lead to a potential contamination of the by-products, and also to mechanical problems associated with the separation of food waste from plastic bags during pre-treatment, incurring both operational and economic problems (Hartmann et al., 2004). As such, the use of paper bags, instead of plastic, can improve the quality of source-segregated food waste (in terms of contamination), leading to an efficient pre-treatment and an improved quality of the co-digestion by-products. This would provide the opportunity to create new markets for the digestate and biogas, and promote the utilisation of food waste as a resource. To achieve source-segregation of food waste the participation and engagement of the public is important. Poor food waste separation at source can hinder the efficiency of pre-treatment and affect the anaerobic co-digestion process efficiency, leading to a reduction in the revenue from renewable energy generation and the potential utilisation of the digestate. While the use of FWDs can be an easier and reliable way to manage food waste at source, its impacts to the water industry in terms of changes in the wastewater characteristics, increased sludge production and

potential implications to the sewerage system are important aspects that must be taken into account for their sustainable implementation (Chapters 4, 5).

The collaboration of all relevant stakeholders and regulators is required to support changes in regulatory frameworks necessary for the application of co-digestion of food waste with sewage sludge. Such a development would prove profitable for both the water industry and the UK, as part of an interdisciplinary, integrated and holistic strategy, with a potential to deliver benefits across the two sectors, taking us a bit closer to sustainability.

7.5 Conclusions

The application of anaerobic co-digestion of food waste with sewage sludge, although well established in many European countries, is still in its infancy in the UK. This process has many benefits to offer, with a successful application associated with an increased renewable energy potential, outweighing constraints associated with the variability of food waste and its handling requirements prior to co-digestion. With both regulations and water infrastructures designed and constructed on the basis of linear views and sectorial requirements and conditions and technologies from the past in many parts of the world, in the UK, sewage sludge and food waste digestion operations are also under very different regulatory and management regimes. With sustainability requiring that we do not address single issues in isolation, but through a systems approach that delivers integrated solutions, co-digestion of food waste with sewage sludge could become such a solution. However, the key factor for the implementation of the co-digestion process is regulations that govern the planning and operation of the co-digestion process. If carefully applied, co-digestion can deliver beneficial synergies for the water industry and authorities responsible for food waste management. The collaboration of all relevant stakeholders and regulators to support changes to current regulatory frameworks to enable this, is proposed as the way forward, particularly as their complexity has been identified as the major hurdle to the implementation of co-digestion in the UK.

8 OVERALL DISCUSSION

Environmental protection has been a key driver to changes in solid waste management. The establishment of the revised EU Waste Framework Directive (rWFD) in 2008, which emphasised the need for choosing appropriate technologies for improving the protection of human health and the environment by promoting waste prevention, reuse and recycling (European Parliament and Council, 2008), in conjunction with the targets set by the earlier EU Landfill Directive in 1999 for diverting biodegradable waste from landfill, have started to make a positive impact in the field of food waste management (Chapter 5).

This thesis was confined to the examination of the use of FWDs and the anaerobic co-digestion of food waste with sewage sludge that are both directly related to local authority (LA) practices and water industry operations. These options were investigated as two options for food waste management alternative to landfill.

Findings from the research undertaken in the previous Chapters aimed to inform decision-making, particularly in terms of area specific characteristics and practices (Chapter 4, 5 and 7). Therefore, a methodology has been developed to assist LAs and the water industry to identify the impacts of food waste management options to their operations, using multiple criteria aiming to inform and facilitate the decision-making process.

Multi-criteria analysis is frequently applied to compare and assess the impacts of different policies and waste management options, and has been selected for food waste management in the context of sustainability (Balasubramaniam & Voulvoulis, 2005; Bazzani, 1998; Boggia & Cortina, 2010; Cheng et al., 2002; Ferrarini et al., 2001; Garfì et al., 2009; Hokkanen & Salminen, 1997).

8.1 A sustainability decision-making framework to evaluate food waste management options

A proposed multi-criteria framework was developed (Figure 8-1), involving a number of basic steps beginning from defining the problem, which in this case is choosing between the two municipal food waste management options (the use of FWDs and anaerobic co-digestion with sewage sludge), in terms of sustainability (Figure 8-1) (Cheng et al., 2002; Ferrarini et al., 2001; Hermann et al., 2007; Hokkanen & Salminen, 1997).

Environmental, social and economic criteria were selected for the comparative assessment of the two alternatives (Figure 8-1). Reflecting sustainability, these criteria were given the same weight so as to indicate their relative importance to decision-making, demonstrating the requirement that the alternatives perform equally well in all three areas (Balasubramaniam & Voulvoulis, 2005; Hokkanen & Salminen, 1997).

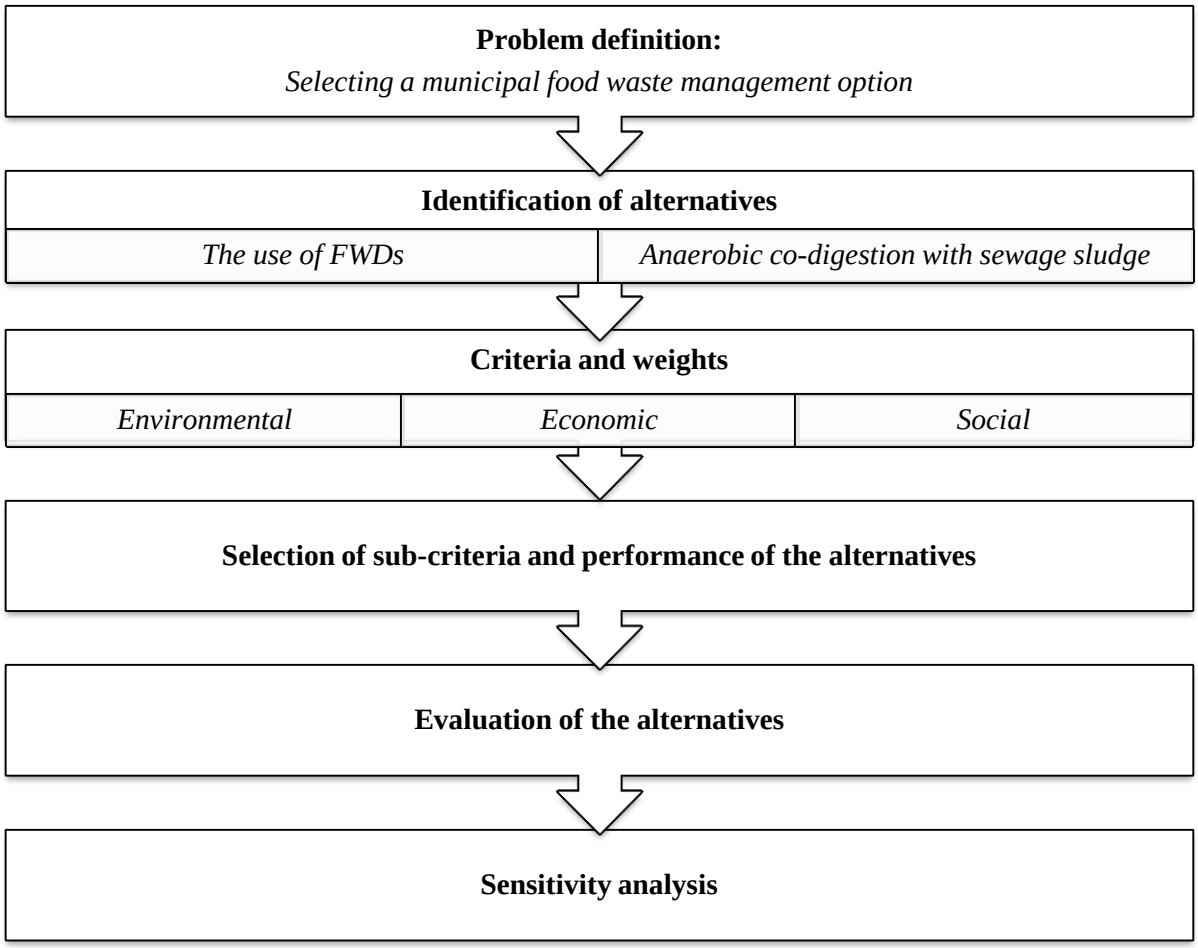
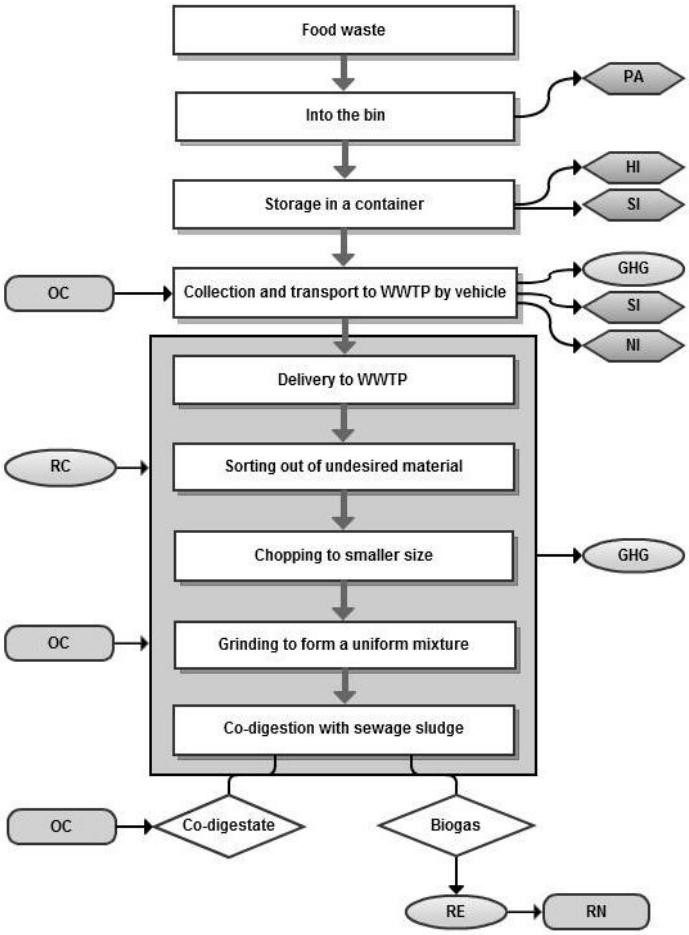
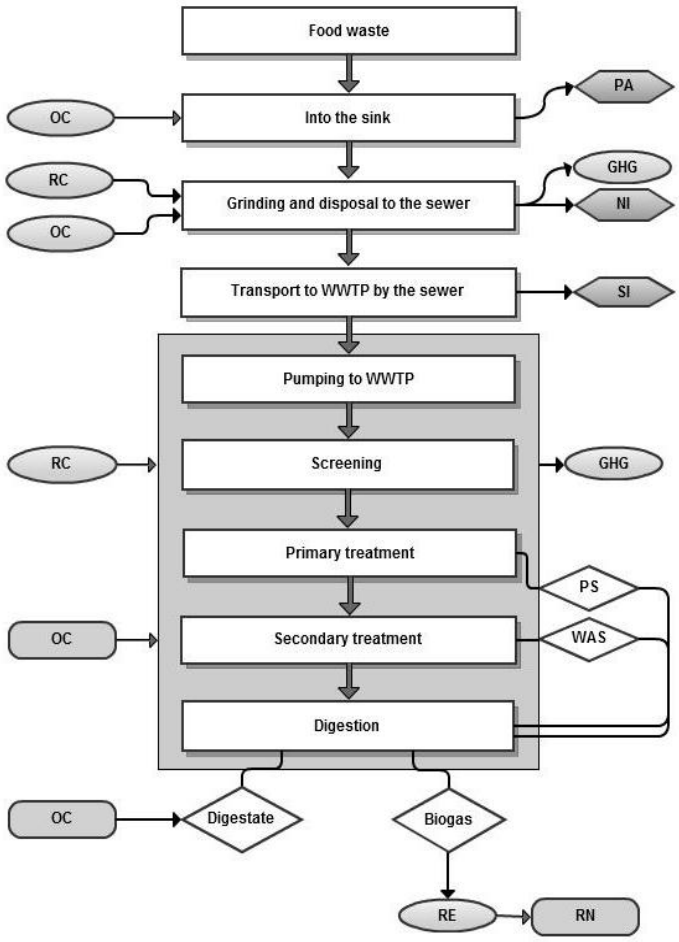


Figure 8-1 Sustainability decision-making framework using multi-criteria approach to evaluation of the problem

The three sustainability criteria were further evaluated using a number of sub-criteria, based on the process of each food waste management option. A mass balance approach (Figure 8-2) was used to identify the appropriate sub-criteria for this evaluation with environmental, economic and social sub-criteria included as inputs and outputs in terms of money and resources, emissions, benefits, and other impacts (Figure 8-2) (Bolzonella et al., 2003b; Edelmann, 2000a; Hartmann et al., 2002; Nilsson et al., 1990; Rosenwinkel & Wendler, 2001).



Anaerobic co-digestion of food waste with sewage sludge



Use of Food Waste Disposal units (FWDs)

Environmental sub-criteria

RC: Resources Consumption
GHG: Greenhouse gas emissions
RE: Renewable Energy

Economic sub-criteria

OC: Operational Cost
RN: Revenue

Social sub-criteria

PA: Public Acceptance
HI: Health Impacts
SI: Smell Impacts
NI: Noise Implications

Figure 8-2 A mass balance approach to identifying the sub-criteria of the two food waste management alternatives

Three, two and four sub-criteria were used to evaluate the performance of the food waste management option alternatives with regards to environmental, economic and social criteria, respectively (Figure 8-2), as follows:

RC: refers to resources consumption including land requirements and water and energy consumption in both food waste management options (Balkema et al., 2002; den Boer et al., 2007). Energy consumption can be further divided into energy from the burning of fossil fuels or electricity and heat consumption in households and in the industry (den Boer et al., 2007).

RE: is the energy generated by the use of biogas (or methane, CH_4) produced during the anaerobic digestion (AD) process, and is considered the resource that can outweigh the impact of energy consumption. In this study the AD process is assumed to be the most dominant sewage sludge management option in water industry operations.

GHG: these emissions can be either *direct,* through the combustion of fuels, fugitive losses from the anaerobic reactor and the utilisation of biogas, or *indirect* through energy consumption in the form of fuel, electricity and heat, and the production of materials - used for the manufacturing of FWDs or the construction of the facility - (upstream), and from the digestate and its use (downstream) (Møller et al., 2009).

OC: includes the costs of investment, operation and maintenance which are the most commonly used in assessing waste management options (Balkema et al., 2002). These costs may depend on area characteristics (architectural and geographical), and the practices followed in the water industry operations and LA practices.

RN: is the benefit generated from recovered energy (renewable energy) or materials (digestate), and from the reduction in cost of municipal waste collection and transport (den Boer et al., 2007).

PA: when householders are directly involved in a food waste management option in terms of adopting new handling concepts, including the use of technology, or in terms of waste separation and collection, the potential acceptance of each option is an important socio-cultural consideration (Balkema et al., 2002).

HI: are impacts associated with the food waste management process and are considered very important to the implementation of each option, due to implications for human health (Defra, 2007a)

SI: is the annoyance caused by unpleasant odours (Bottero et al., 2011). This is difficult to measure because people respond to odours in a different way depending on factors such as previous exposure, personal health, sensitivity and social background (Defra, 2007a).

NI: is the annoyance caused to individuals by noise when implementing each food waste management option (Bottero et al., 2011). However, the annoyance caused by these implications depends on each individual's response to the noise.

The performance of the alternatives with regards to the different sub-criteria is a complex process that differs between regions and depends on data availability and regional characteristics and practices. To facilitate the application of this process, examples and mechanisms to calculate their performance are summarised in Table 8-1.

Table 8-1 Method to calculate the performance of the alternatives to the identified sub-criteria

		FWDs	Anaerobic co-digestion
Environmental	Resources consumption (RC)	Calculated based on: - Water consumption for grinding (household) - Energy consumption for food waste grinding (household) - Energy consumption for water treatment and distribution, wastewater pumping and treatment and sludge management and disposal (water industry)	Calculated based on: - Fuel consumption for collection and transport (LAs) - Land requirements for waste processing facilities (water industry) - Energy consumption for food waste processing (water industry) - Additional energy consumption for heating-up the digester (water industry)
	Greenhouse gas emissions (GHG)	Calculated based on energy consumption for: - The grinding of food waste (household) - The treatment of the additional water used for the grinding of food waste (water industry) - The pumping and treatment of additional wastewater generated (water industry) - The management of the additional sludge produced (water industry) - The burning of fossil fuel for the disposal of the produced digestate (water industry)	Calculated based on: - The burning of fossil fuels for the collection and transport of food waste to the pre-treatment facilities (LAs) - Energy consumption at food waste processing facilities (water industry) - Additional digester heating for treatment of food waste together with sewage sludge (water industry) - The burning of fossil fuels for the disposal of the produced digestate (water industry)
	Renewable		

	energy (RE)	Calculated based on the biogas/CH_4 generation as a result of the additional sewage sludge digested, produced at the WWTPs[1] due the use of FWDs (water industry)	Calculated based on the biogas/CH_4 generation from the digestion of food waste together with sewage sludge (water industry)
Economic	Operational Cost (OC)	Calculated using the cost of: - Purchase and installation of a FWD (householder) - Maintenance of the FWD (householder) - Energy consumption for using the FWD (householder) - Water treatment and distribution (water industry) - Wastewater treatment (water industry) - Sludge management (water industry) - Digestate disposal (water industry)	Calculated using the cost of: - Collection of food waste (LAs) - Transport of food waste to the waste management facilities (LAs) - Gate fee for food waste delivery at WWTPs (LAs) - Investment cost for the waste management facilities (water industry) - Operational cost of food waste processing (water industry) - Additional heating of the digester due to the addition of food waste (water industry) - Digestate disposal (water industry)
	Revenue (RN)	Benefit of reduction in cost of collection and transport (LAs) Benefit from renewable energy generation from the additional sludge digestion (water industry)	Benefit from additional renewable energy generation (water industry)
Social	Public acceptance (PA)	These criteria are usually measured by conducting a survey that involves the public's participation. Generally: - PA is critical due to technology involvement. - HI depend on people sensitivity and area-specific characteristics. - The response of the public to SI and NI differs due to sensitivity, culture, habits and previous exposure.	These criteria are usually measured by conducting a survey that involves the public's participation. Generally: - PA is relied on people willingness to separate their food waste from other residual waste. - HI depends on people sensitivity and area-specific characteristics. - The response of the public to SI and NI differs due to sensitivity, culture, habits and previous exposure.
	Health impacts (HI)		
	Smell impacts (SI)		
	Noise implications (NI)		

[1]Wastewater Treatment Plants

The relative contribution of the sub-criteria to the criteria is presented in the Table 8-2.

Table 8-2 Sub-criterion weighting for the two food waste management options

Criteria	Sub-criteria	Weighting	Individual weighting (%)
Environmental	RC	⅓	11.11
	GHG	⅓	11.12
	RE	⅓	11.11
Economic	OC	½	16.67
	RN	½	16.67
Social	PA	¼	8.33
	HI	¼	8.33
	SI	¼	8.33
	NI	¼	8.33
Total:			100

The individual weighting for each sub-criterion is calculated by multiplying the weight of the criterion by the weight of each individual sub-criterion within each group. The sum of the individual weighting of all sub-criteria must then be equal to 100%, to ensure the consistency of the weighting assignment (Table 8-2).

8.2 Application of the sustainability decision-making framework in the Anglian region

In order to assess the sub-criteria, data from Anglian Water services and from the LAs in the Anglian region were collected and used. When these were not available, data based on UK figures were adopted from the literature. For the sub-criteria to be comparable, the overall evaluation was done per tonne of food waste in both processes. Additionally, it was assumed that FWDs can grind everything and that no regulatory constraints limit any stage of the two processes. The performance of the alternatives according to the different sub-criteria is presented in an impact matrix (Table 8-3).

To ensure that that the results were comparable, water consumption in the use of FWDs was omitted from the analysis, and so was land requirement for the anaerobic co-digestion of food waste with sewage sludge. As such, RC was limited to energy consumption estimates expressed in kWh per tonne food waste in both processes (Table 8-3). In the use of FWDs, the energy consumption was calculated based on the electricity required for grinding food waste and its subsequent treatment in the wastewater treatment plants (WWTPs). Because energy consumption rates from Anglian Water Ltd., were not available, these were calculated based on the cost of electricity in the WWTPs and the UK industrial electricity price of 7.89p per kWh, for which details can be found in Appendix D (DECC, 2011; Monson et al., 2007; Ofwat, 2011). For anaerobic co-digestion of food waste with sewage sludge, RC was calculated based on the amount of fuel consumption during the collection and transport of food waste, which was expressed into kWh per tonne food waste using a conversion factor of 10.89 kWh per litre diesel burnt, and also from the processing of food waste at the waste management facilities (Banks et al., 2011; Burnley et al., 2011; Carbon Trust, 2012). Details on these calculations can be also found in Appendix D.

GHG emissions were expressed in carbon emission equivalents (kgCO_2) (Table 8-3). Carbon emissions from the burning of fossil fuels (diesel) and electricity consumption were evaluated using the latest conversion factors of 2.67kgCO_2 per litre diesel and 0.525kgCO_2 per kWh of electricity consumed, respectively (Carbon Trust, 2012).

Diggleman and Ham (2003) reported that when using FWDs, 4.8kg of CH_4 could be generated from 100kg of food waste (Diggelman & Ham, 2003). A tonne of food waste digested was reported to give 260m^3 of biogas, of which 60% was CH_4 (European Commission, 2001b). Based on these, and using the CH_4 calorific value of 13.93kWh/kg, the RE was calculated for both processes (Table 8-3) (Tchobanoglous & Burton, 2003).

The cost of electricity for grinding food waste was estimated based on the UK domestic electricity price conversion factor of 12.89p per kWh. This cost, together with the cost of sewage collection and treatment and sludge treatment and disposal, calculated based on the methodology developed in Chapter 5 and based on the latest data provided by the Anglian Water (Ofwat, 2011), was used to calculate the OC for the use of FWDs (Table 8-3). Because of the difficulty in separating the maintenance and investment costs to the water industry, these were embedded in the aggregated OC. Sewer cleansing and maintenance, while constituting an important cost, were not calculated in the analysis mainly because there are no direct costs associated with the disposal of food waste in the sewer. Details on the estimation of these costs can be found in Appendix D. The OC for the anaerobic co-digestion of food waste with sewage sludge was calculated based on the cost of food waste collection, transport and treatment (Table 8-3). The cost of waste collection was based on data from LAs in the Anglian region (Chapter 4), whereas the cost of transport was calculated based on a figure of 6.5litres per tonne food waste transport and a gate fee cost of £52 (Burnley, 2001; WRAP, 2009). The OC of pre-treatment and digestion was based on a figure of £55per tonne food waste processed, which covered the costs of plant construction, operating and disposal of by-products (Eunomia, 2007; Monson et al., 2007).

The RN from RE was based on the recent renewable energy price of £100 per 1MWh generated (WRAP, 2008b). RN from reduction in cost of collection when using FWDs was not included in this analysis (Table 8-3).

Social sub-criteria were evaluated based on a qualitative analysis. For this analysis data collected from the literature with regards to the public's response to and acceptance of the implementation of each

food waste management option, and in health, smell and noise concerns, were used. Because of the inherent difficulty to differentiate the public's response into a wider scale of measurement, it was considered appropriate to use a scale of two degrees of measurement (+/++), with + being used for the process with the least positive response, and ++ for the process with the higher positive response.

In terms of PA, the use of FWDs was reported to be associated with some unwillingness, as a number of people find their use annoying due to noise implications of using the unit on a daily basis, or even laziness (Evans, 2007). With regards to the anaerobic co-digestion of food waste with sewage sludge, the implementation of a separate food waste collection scheme was reported to cause the unwillingness of some people to participate, mainly because of concerns associated with hygiene, odours and vermin issues (WRAP, 2009). Nevertheless, the majority of people involved in the separate collection responded positively to it. Due to the inherent difficulty to decide on which option is more publicly acceptable, it was assumed that in the UK, the public's ignorance with regards to the use of FWDs would possibly make them less preferable than separate collection of food waste (Table 8-3).

HI arising from the storage and rotting of food waste can occur through the inhaling of odorous emissions such as volatile organic compounds (VOCs) and sulphur compounds (Defra, 2007a). VOCs, such as methane-thiol (MeSH) and hydrogen sulphide (H_2S) can cause nausea, bronchitis and gastrointestinal problems to a high level exposure (Defra, 2007a). Also, the personnel involved in the collection and treatment of food waste are subject to certain health impacts (CECED, 2003). HI related to the disposal of food waste into the sink have not been as widely reported. However, in the study of Evans (2007) it was reported that the use of FWDs might help attract rats, although this has not been proved yet (Evans, 2007). Therefore, it was assumed that the use of FWDs has a more positive effect to human health than the anaerobic co-digestion with sewage sludge (Table 8-3).

The storage of food waste in households and its subsequent collection and handling can be associated with SI (CECED, 2003). Food waste grinding into the FWDs has reported not to release any kind of unpleasant odours, though food waste disposed into the sewers has the possibility of producing some.

The anaerobic degradation of food waste particles in the sewer leads to the production of H_2S, the principal cause of malodours even at very low concentrations (Chapter 4). Based on these, it was assumed that FWDs have a more positive effect on SI compared to the anaerobic co-digestion with sewage sludge (Table 8-3).

NI associated with the use of FWDs in the inside of a household can be of a greater magnitude as when coming from the outside and, thus, be more annoying. In the study of Evans (2007), it was reported that a number of people chose not to use a FWD mainly because of the noise implications related to it (Evans, 2007). As such, and with limited evidence in the literature with regards to the public annoyance by the noise generated during the collection of food waste, it was assumed that anaerobic co-digestion with sewage sludge has a more positive effect to NI than the use of FWDs (Table 8-3).

Table 8-3 Impact matrix of the two food waste management options

Criterion	Sub-criterion	Units	Alternatives	
			FWDs	Anaerobic co-digestion with sewage sludge
Environmental	RC	kWh/tonneFW	188.34	265.71
	GHG	kgCO_2/tonneFW	99.26	84.27
	RE	kWh/tonneFW	668.64	2173.08
Economic	OC	£/tonneFW	68.47	192.97
	RN	£/tonneFW	66.86	217.31
Social	PA	+/++	+	++
	HI	+/++	++	+
	SI	+/++	++	+
	NI	+/++	+	++

While there are many ways to analyse the matrix calculations and rank the alternatives, the weighted summation analytical method was chosen to rank the two food waste management alternatives. The weighted summation is a well-established method, widely used because of its ease. The standardised

score of each criterion is multiplied by its assigned weighting, and rankings of the alternatives are then determined based on the weighted sum of the standardised scores of each alternative (Boggia & Cortina, 2010; Chon 2012).

According to the weighted summation, the anaerobic co-digestion of food waste with sewage sludge is shown that it could be more sustainable than the use of FWDs (Figure 8-3).

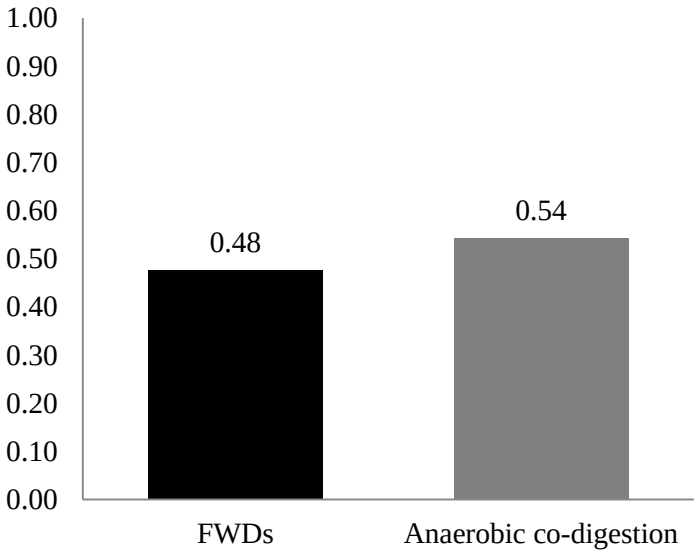


Figure 8-3 Overall score of the evaluation of the alternatives

The performance of the evaluation of the two alternatives based on the selected criteria was additionally assessed with anaerobic co-digestion presenting higher sustainability on environmental and economic criterion than the use of FWDs, with an equal performance on the social criterion (Figure 8-4).

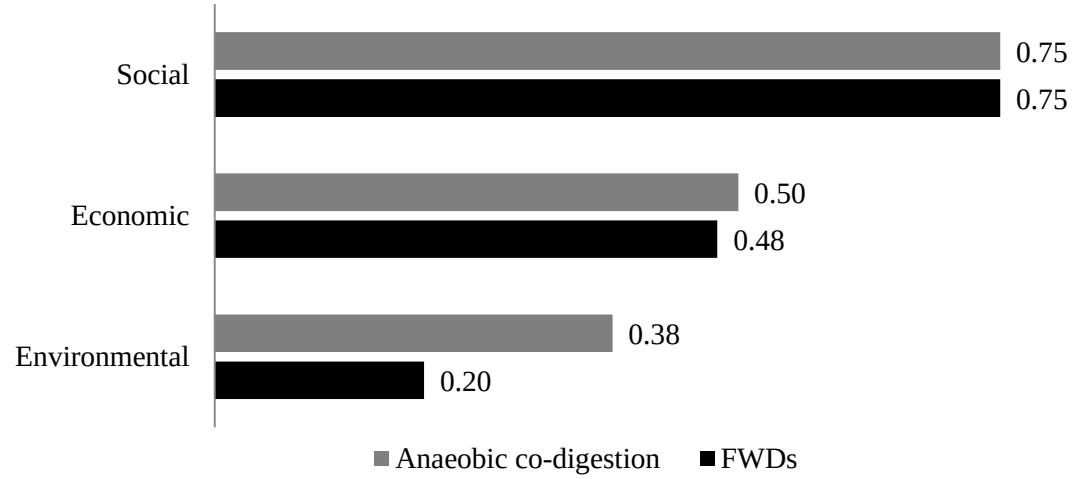
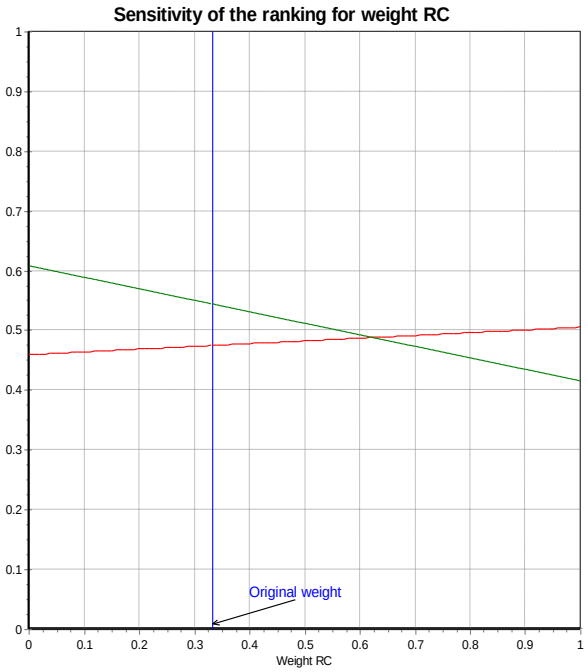
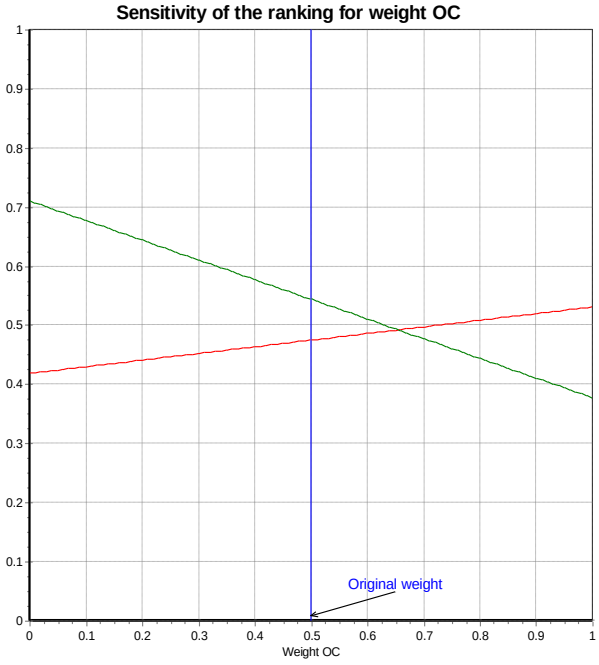


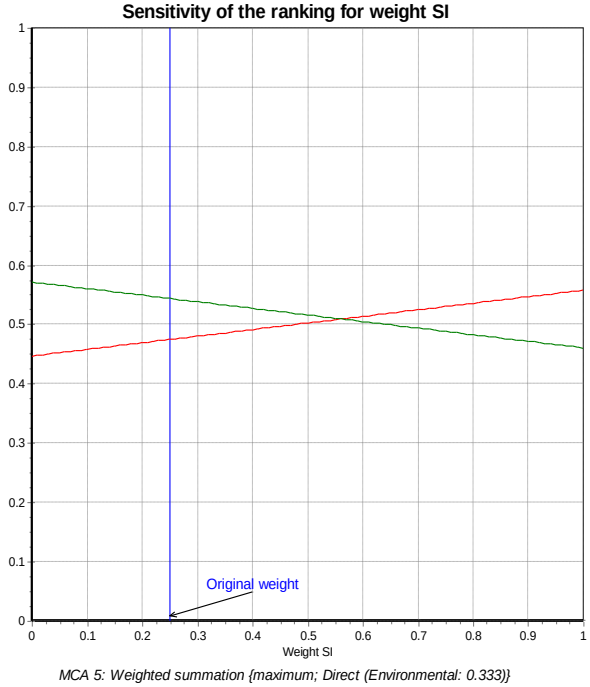
Figure 8-4 Evaluation of the performance of the two alternatives on each criterion

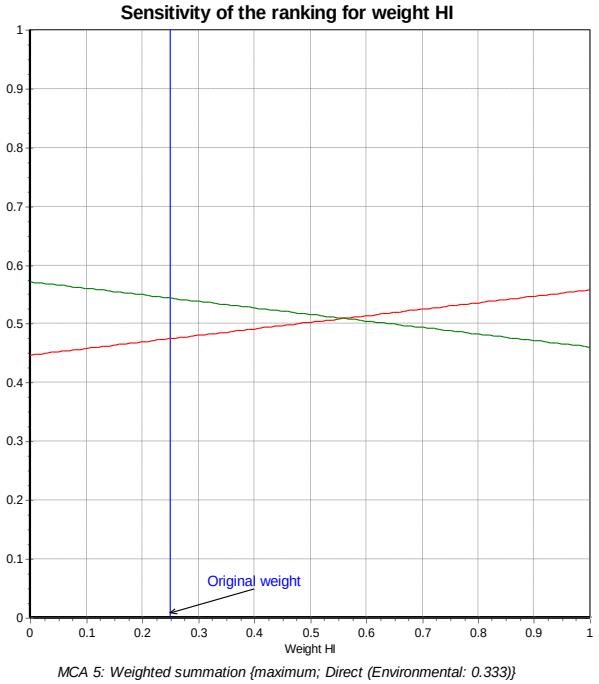
After evaluating the performance of the alternatives in relation to the identified sub-criteria, a sensitivity analysis was carried out. This analysis can help decision makers to better understand the consequences of their decisions and how these can affect the overall results of the comparison (Boggia & Cortina, 2010). It can be performed by changing the weights as a way to reflect the response of the alternatives to the relative importance of the sub-criteria to the criteria and identify the sub-criteria which could result in a ranking reversal (Boggia & Cortina, 2010; Bottero et al., 2011). By increasing or decreasing the weight of the individual criteria or sub-criteria, the ranking of the alternatives can be observed and its stability can be assessed. If the ranking is found to be highly sensitive to small changes in the criteria weights, a careful review of the weights is recommended (Bottero et al., 2011).

The sub-criteria of resources consumption (RC), operational cost (OC), smell and health impacts (SI and HI, respectively) were the most sensitive to changes in the weights, and could cause a reversal in the ranking of the alternatives (Figure 8-5). As evidenced from the sensitivity analysis, the assignment of specific weightings to the sub-criteria would be of substantial importance to the ranking of the alternatives. The results of the sensitivity analysis of all the sub-criteria are presented in Appendix D.









Green line: represents the anaerobic co-digestion of food waste with sewage sludge
Red line: represents the use of FWDs

Figure 8-5 Results of sensitivity analysis of sub-criteria with variation in their weightings

8.3 Discussion

The sustainability framework proposed in this Chapter was developed based on the findings of the previous Chapters in order to support decision-making in the area of food waste management with regards to the use of FWDs and the anaerobic co-digestion of food waste with sewage sludge through the use of the multi-criteria approach. The method used in the context of this framework was outlined based on the benefits and impacts that these two food waste management options pose to LAs and water industry operations, as identified from the research undertaken in Chapters 4, 5, 6 and 7. This methodology constitutes an important tool for both LAs and the water industry to decide on which option is the most sustainable, based on their area-specific characteristics and practices. Nevertheless, based on these characteristics and practices, the proposed methodology can be further developed to better fit the needs and requirements of the relevant authorities.

While there is a variety of assessment tools and approaches available which can be used to examine economic, environmental, social or technical aspects of a problem (Salhofer et al., 2007), the need for combining all aspects together has led to the development of mechanisms such as the Life Cycle Assessment (LCA) and Multi-Criteria Analysis (MCA) (Hermann et al., 2007). While the LCA approach evaluates and assesses the environmental impacts encountered during the entire life cycle of a product, process or activity, it requires a large amount of detailed data, time and knowledge for its application (Hermann et al., 2007). The multi-criteria approach, on the other hand, allows for the comparison of alternative options on the basis of a set of criteria, using a mixture of qualitative and quantitative data, making it appropriate for environmental decision-making (Balasubramaniam & Voulvoulis, 2005; Ferrarini et al., 2001; Hermann et al., 2007). The additional advantage of being able to take some uncertainties into account makes the multi-criteria approach even more attractive, as environmental problems are often associated with uncertainty (Balasubramaniam & Voulvoulis, 2005). However, a limitation of the multi-criteria approach is the subjectivity of assessing the contribution of the sub-criteria to the selected criteria.

While this limitation offers flexibility to decision-makers to evaluate both food waste management options based on their interests and needs, it bears the risk of triggering a controversy between stakeholders with opposing interests. To avoid this disagreement, the collaboration of all the authorities involved in the assessment is required in order to decide on the evaluation procedure and to adopt a sustainability decision-making framework developed based on their common needs and individual requirements.

The application of the sustainability framework in the Anglian region in the UK demonstrated that the anaerobic co-digestion of food waste with sewage sludge could be more sustainable than the use of FWDs. This result only provides guidance to stakeholders, as the different primacies originating from them could possibly lead to a reversal in the overall outcome. This is evidenced by the sensitivity analysis, which revealed that changes in the relative importance of some sub-criteria, such as those of resources consumption (RC), operating cost (OC), health impacts (HI) and smell impacts (SI)), could change the outcome.

It has to be highlighted that while the data that have been collected from the literature came from different sources not fully reflecting the process in its entirety, these data were sufficient for the analysis, which aimed to demonstrate how the sustainability framework developed could be applied in the Anglian region. In addition, the qualitative analysis undertaken for examining some sub-criteria was a simplistic evaluation process which may have led to the under- or over-estimation of some of the sub-criteria. A paradigm is public acceptance which is the prerequisite of the successful implementation of a food waste management option. Public acceptance entails the engagement of the public in doing what is required for a food waste management option to be successfully implemented. If this obligation is not met, it creates implications for the actual sustainability of the implemented food waste management option. Poor food waste separation is likely to reduce anaerobic co-digestion process efficiency, leading to a reduction in the revenue from renewable energy generation, which could also be affected by contaminants present in food waste. With the use of FWDs, the separation of food waste is easier and more reliable, since other wastes would be difficult to grind, but on the other hand the noise implications and laziness associated with operating the units might result in the

disposal of food waste into the residual waste stream. This would result in food waste ending up in landfill, thus reducing the amount of energy that could be recovered and also contributing to negative environmental effects. This behaviour presents a challenge as it could exceed the potential of the two food waste management options to be sustainable. Food waste should be regarded as an opportunity that can provide valuable and viable solutions if properly managed at source and subsequently utilised by the appropriate authorities. If this is realised then such challenges can be overcome and the value of food waste can be recognised. This must be carefully planned in order to pursue sustainability. The interrelationship between water-energy-food must be grasped as the system through which justifiable solutions can be provided, building the foundations of a greener and sustainable future.

Proper consideration of the environmental, economic and social impacts, according to area-specific characteristics and practices followed, is necessary, both for the appropriate evaluation of the sustainability performance of each food waste management alternative, as well as their subsequent comparison involved in decision-making. This proper consideration can also help LAs and the water industry to integrate economically feasible and socially acceptable practices holistically, along with key environmental benefits such as the minimisation of GHG emissions, reduction of energy use and increased renewable energy generation. Nonetheless, challenges related to either the implementation of the use of FWDs or the anaerobic co-digestion of food waste with sewage sludge will persist, as a result of increasing population, urbanisation, an ageing infrastructure and technological development. As such, both authorities and the industry must be ready to adapt to changing circumstances.

However, it is not just the responsibility of LAs and the water industry to decide on the sustainability of a food waste management option, but also of the public. An integrated approach is required to address all relevant challenges more holistically. The achievement of such an approach requires balanced communication and collaboration of all the relevant stakeholders, who, beyond their own interests and needs, take into serious consideration the opinion and needs of the public. This is because, in order for a food waste management option to be considered sustainable, it must maintain socially acceptable levels of service providing benefits to the community and reducing impacts to the environment in both the long- and short-term.

8.4 Recommendations for future work

A thorough collection of detailed data is essential for the comprehensive assessment of the two food waste management options, as it would help to improve decisions on their sustainability. To achieve this, a review of the environmental, economic and social aspects of the use of FWDs and the anaerobic co-digestion of food waste with sewage sludge based on area-specific characteristics and practices is a necessary area for further research.

With regards to the use of FWDs, while many studies have focused on the investigation of their impacts in the wastewater treatment operations (Chapter 4), only a limited number of studies have examined their impact on the sewerage system. It is well known that these impacts will vary depending on the region, size of the drainage system, age and frequency of maintenance (Chapter 4). However, it is an important aspect of the use of FWDs that should be taken into account in the overall assessment of the viability of the use of FWDs as a food waste management option. Therefore, further research should focus on examining these impacts based on the analysis of case studies in order to establish the importance of these impacts to the long-term sustainability of the use of FWDs.

Equally important to the sustainability assessment is the carbon footprint of FWD use. The manufacture and transport of FWDs to their final destination adds to the carbon footprint, along with the carbon emissions related to the disposal of food waste into the sewer and its treatment in the wastewater treatment plant. To assess the importance of carbon footprint in the long-term viability of the use of FWDs as a food waste management option, an exhaustive analysis is required, taking into consideration all steps of the process. This kind of analysis would require both time and data, but would help deliver a superior outcome with regards to the sustainability assessment of the process. Equally, in order to compare the options, the same analysis must be conducted for the anaerobic co-digestion of food waste with sewage sludge.

The use of FWDs as a food waste management option would result in a small fraction of food waste remaining in the residual waste stream, and hence, in landfill. The impact of this diversion to the environment should be appraised and included in the overall assessment of the use of FWDs. A more

thorough investigation would hence shed light on the significance of this diversion to the overall sustainability of the use of FWDs as a food waste management option.

A process that could positively affect the sustainability of the use of FWDs is the incorporation of the anaerobic treatment process into municipal wastewater treatment operations. As reported in Chapter 6, while in warm climates this process could be successfully used, in colder areas its viability is as of yet unclear. This is due to the decreased performance of the anaerobic reactor at low temperatures, as well as the possibility of methane becoming dissolved in the wastewater and thus being emitted into the atmosphere during the aerobic treatment step. As such, further research incorporating the investigation of the environmental and economic sustainability of possible anaerobic-aerobic treatment processes and the use of more advanced and sophisticated technologies would be beneficial for the water industry, as it would help stakeholders to make decisions over new wastewater treatment processes, and move towards a more sustainable future.

While suggesting the implementation of anaerobic co-digestion of food waste with sewage sludge to LAs and the water industry is appropriate due to its numerous benefits, substantial amounts of uncertainty still exist making it difficult to draw firm conclusions. While many studies illustrate the possible benefits and constraints of this process, few studies exist that integrate its environmental, economic and social aspects. As such, these aspects should be further investigated to aid decision making with regards to sustainability. A life cycle assessment would be a useful way to achieve this, as it would provide a detailed analysis able to appropriately assess the overall process.

The multi-criteria approach used for the comparison of the two food waste management options in terms of sustainability is a simple but consistent methodology which can give respectable results. Nevertheless, the involvement of stakeholders in the evaluation procedure and the commencement of a survey to obtain the public's opinion would provide improved insight into the environmental, economic and social aspects of both food waste management options, as a greater objectivity will be given to the evaluation. This would result in an enhanced sustainability comparison with well justified results.

9 CONCLUSIONS

Most food waste in the UK is currently disposed to landfills. New food waste management options have come into consideration as a result of new GHG emission targets, targets relating to the waste reuse, recycling and the diversion of waste from landfill, as well as other policies related to sustainability. Among these options are the use of food waste disposal units (FWDs) and the anaerobic co-digestion of food waste with sewage sludge, both of which are directly relevant to water industry operations. As such, the diversion of food waste from landfill has become an issue of concern to the water industry.

While the use of FWDs, which involves the grinding of food waste under the sink and its disposal to the sewer for treatment with wastewater, offers profound benefits to local authorities (LAs) through reductions in cost from waste collection, it also poses a challenge to water industry operations due to increased energy consumption and the associated costs required for the treatment of the additional wastewater. As a result, for FWDs to be implemented as a sustainable food waste management option a platform must be established within which savings from waste collection and disposal of food waste could be transferred to subsidise the water industry. However, this can only be achieved by generating significant savings to LAs that could compensate for the increased costs to the water industry. As such, policy intervention in the UK is mandatory in order to regulate the use of FWDs, either by promoting them as an alternative to landfill to increase savings from waste management, or by banning them to reduce costs for the water industry.

Given the prospect of implementing FWDs as a food waste management option, the incorporation of anaerobic treatment processes into municipal wastewater treatment operations might be a viable solution, as it would bring about higher removal efficiency, increased biogas generation and less sludge production, thus contributing to the ambitions of the water industry to minimise their energy requirements and carbon emissions. The increased solubility of methane at the low temperatures prevailing in the UK could be a challenge to the water industry, though. This challenge can be dealt with numerous solutions of which, the economics must be assessed to ensure that the balance of

benefits is not shifted away from anaerobic treatment to conventional aerobic treatment. Nevertheless, continued research is a prerequisite for the development of an advanced and sophisticated anaerobic technology which could be successfully implemented to give environmental and economic benefits independent of the prevailing climatic conditions.

The anaerobic co-digestion of food waste with sewage sludge is an option that has recently emerged due to its potential for renewable energy generation. While this option is well-established in other European countries, both as a promising renewable energy and highly sustainable technology, its use is still in its infancy in the UK. Limitations associated with the lack of knowledge on the proper use of anaerobic technology, operational problems related to substrate characteristics, lack of investment, difficulty of getting the incentives provided, and regulatory constraints related to the process as a whole with an emphasis on its inputs and outputs, have hindered the implementation of the anaerobic co-digestion process in the UK. With sustainability becoming a driver for change in waste management practices, the consideration of the anaerobic co-digestion process has come to the forefront, requiring all relevant parties to collaborate in order to revise the regulatory framework that currently acts as an obstacle to co-digestion initiatives. Economic transparency and incentives for the process are also required in order to make decisions on the scale of its implementation, as a significant capital investment and operational expenditure might be required to create new assets or to upgrade the existing ones.

The sustainability framework, developed to inform and facilitate decision-making based on multi-criteria approach, is a flexible methodology that can be used by both the LAs and the water industry to evaluate the use of FWDs and the anaerobic co-digestion of food waste with sewage sludge. The question of which option is more sustainable is difficult to answer, as area-specific characteristics, individual LA and water industry practices, as well as the availability of data used in the evaluation, can all affect the overall performance. But even with these characteristics being fully addressed, the sustainability assessment of the two food waste management alternatives would still be affected by variations in stakeholders' interests. While the flexibility provided by the multi-criteria approach is essential, it could lead to controversies among the relevant stakeholders with regards to selecting the

most sustainable alternative. In the context of sustainability, which requires embracing the problems in an interdisciplinary, integrated and holistic approach, LAs and the water industry should collaborate to decide on the appropriate food waste management option based on a common evaluation procedure developed according to their needs and concerns.

To conclude, it should be emphasised that diverting food waste from landfill is not only a challenge for the water industry, but a challenge for all of us as food consumers and wasters, and also for the government which regulates waste management processes. Individual approaches in waste management can lead to unsustainable solutions. Nonetheless, diverting food waste from landfill and realising its potential as a resource that can provide valuable source of energy marks just the half-way point on the road to sustainability. Balancing the relationship between water, food and energy is the real means to viable solutions and the ultimate approach to achieving sustainability. Collaboration of all stakeholders is, therefore, essential for selecting and implementing the most viable solution for food management, laying the foundations for a sustainable future. Finally, it must be highlighted that it is our ethical obligation to future generations to keep waste prevention and minimisation as the main way to divert food waste from landfill.

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APPENDIX A

: Supplementary material for Chapter 4

A.1 Classification of countries based on their income

All World Bank member economies and other economies with population of more than 30,000 capita are classified according to their income. The classification based on the 2008 gross national income (GNI) per capita, was calculated using the World Bank Atlas method.

The classification among income groups is as follows,

- $975 (£567)[1] or less, for low-income economies
- $976-3,855 (£567-2,236), for lower-middle income economies
- $3,855-11,905 (£2,236-6,905), for upper-middle income economies
- $11,906 (£6,905) or more, for high income economies

Low-income and middle-income economies are sometimes referred to as developing economies; however, this does not imply that the countries of these groups are having the same development status and that the high-income countries have reached a preferred stage of development. As thus, classification by income does not necessarily reflect the development status of each country.

High-income countries are further classified into OECD and non-OECD. OECD stands for the Convention on the Organisation for Economic Co-operation and Development. Twenty countries originally signed this convention on 14th of December 1960 and since then fourteen countries have become members of the Organisation.[2]

The mission of the Organisation is to promote policies that aim to improve the economic and social welfare of people around the world. It provides a forum in which governments can cooperate to solve

[1] Currency conversion rate: $1=£0.58, based on the historical exchange rate data from January 2008 to December 2008 from OANDA Corporation.

[2] OECD (2011), Home page, available at: http://www.oecd.org

common problems and learn through their experiences. It helps governments to understand the origin of their economic, social and environmental changes and measures productivity and global flows of trade and investment.[2] And these are just a few of the responsibilities of OECD.

Table A-1 Classification of countries based on the economy[3]

Economy	Code	Region	Income group
Afghanistan	AFG	South Asia	Low income
Albania	ALB	Europe & Central Asia	Lower middle income
Algeria	DZA	Middle East & North Africa	Upper middle income
American Samoa	ASM	East Asia & Pacific	Upper middle income
Andorra	ADO	..	High income: non-OECD
Angola	AGO	Sub-Saharan Africa	Lower middle income
Antigua and Barbuda	ATG	..	High income: non-OECD
Argentina	ARG	Latin America & Caribbean	Upper middle income
Armenia	ARM	Europe & Central Asia	Lower middle income
Aruba	ABW	..	High income: non-OECD
Australia	AUS	..	High income: OECD
Austria	AUT	..	High income: OECD
Azerbaijan	AZE	Europe & Central Asia	Lower middle income
Bahamas, The	BHS	..	High income: non-OECD
Bahrain	BHR	..	High income: non-OECD
Bangladesh	BGD	South Asia	Low income
Barbados	BRB	..	High income: non-OECD
Belarus	BLR	Europe & Central Asia	Upper middle income
Belgium	BEL	..	High income: OECD
Belize	BLZ	Latin America & Caribbean	Lower middle income
Benin	BEN	Sub-Saharan Africa	Low income
Bermuda	BMU	..	High income: non-OECD
Bhutan	BTN	South Asia	Lower middle income
Bolivia	BOL	Latin America & Caribbean	Lower middle income
Bosnia and Herzegovina	BIH	Europe & Central Asia	Upper middle income
Botswana	BWA	Sub-Saharan Africa	Upper middle income
Brazil	BRA	Latin America & Caribbean	Upper middle income
Brunei Darussalam	BRN	..	High income: non-OECD

[3] The World Bank (2009), Data, available at: http://data.worldbank.org

Bulgaria	BGR	Europe & Central Asia	Upper middle income
Burkina Faso	BFA	Sub-Saharan Africa	Low income
Burundi	BDI	Sub-Saharan Africa	Low income
Cambodia	KHM	East Asia & Pacific	Low income
Cameroon	CMR	Sub-Saharan Africa	Lower middle income
Canada	CAN	..	High income: OECD
Cape Verde	CPV	Sub-Saharan Africa	Lower middle income
Cayman Islands	CYM	..	High income: non-OECD
Central African Republic	CAF	Sub-Saharan Africa	Low income
Chad	TCD	Sub-Saharan Africa	Low income
Channel Islands	CHI	..	High income: non-OECD
Chile	CHL	Latin America & Caribbean	Upper middle income
China	CHN	East Asia & Pacific	Lower middle income
Colombia	COL	Latin America & Caribbean	Upper middle income
Comoros	COM	Sub-Saharan Africa	Low income
Congo, Democratic Republic	ZAR	Sub-Saharan Africa	Low income
Congo, Republic	COG	Sub-Saharan Africa	Lower middle income
Costa Rica	CRI	Latin America & Caribbean	Upper middle income
Côte d'Ivoire	CIV	Sub-Saharan Africa	Lower middle income
Croatia	HRV	..	High income: non-OECD
Cuba	CUB	Latin America & Caribbean	Upper middle income
Cyprus	CYP	..	High income: non-OECD
Czech Republic	CZE	..	High income: OECD
Denmark	DNK	..	High income: OECD
Djibouti	DJI	Middle East & North Africa	Lower middle income
Dominica	DMA	Latin America & Caribbean	Upper middle income
Dominican Republic	DOM	Latin America & Caribbean	Upper middle income
Ecuador	ECU	Latin America & Caribbean	Lower middle income
Egypt, Arab Republic	EGY	Middle East & North Africa	Lower middle income
El Salvador	SLV	Latin America & Caribbean	Lower middle income
Equatorial Guinea	GNQ	..	High income: non-OECD
Eritrea	ERI	Sub-Saharan Africa	Low income
Estonia	EST	..	High income: non-OECD
Ethiopia	ETH	Sub-Saharan Africa	Low income
Faeroe Islands	FRO	..	High income: non-OECD
Fiji	FJI	East Asia & Pacific	Upper middle income
Finland	FIN	..	High income: OECD
France	FRA	..	High income: OECD
French Polynesia	PYF	..	High income: non-OECD

Gabon	GAB	Sub-Saharan Africa	Upper middle income
Gambia, The	GMB	Sub-Saharan Africa	Low income
Georgia	GEO	Europe & Central Asia	Lower middle income
Germany	DEU	..	High income: OECD
Ghana	GHA	Sub-Saharan Africa	Low income
Greece	GRC	..	High income: OECD
Greenland	GRL	..	High income: non-OECD
Grenada	GRD	Latin America & Caribbean	Upper middle income
Guam	GUM	..	High income: non-OECD
Guatemala	GTM	Latin America & Caribbean	Lower middle income
Guinea	GIN	Sub-Saharan Africa	Low income
Guinea-Bissau	GNB	Sub-Saharan Africa	Low income
Guyana	GUY	Latin America & Caribbean	Lower middle income
Haiti	HTI	Latin America & Caribbean	Low income
Honduras	HND	Latin America & Caribbean	Lower middle income
Hong Kong, China	HKG	..	High income: non-OECD
Hungary	HUN	..	High income: OECD
Iceland	ISL	..	High income: OECD
India	IND	South Asia	Lower middle income
Indonesia	IDN	East Asia & Pacific	Lower middle income
Iran, Islamic Republic	IRN	Middle East & North Africa	Lower middle income
Iraq	IRQ	Middle East & North Africa	Lower middle income
Ireland	IRL	..	High income: OECD
Isle of Man	IMY	..	High income: non-OECD
Israel	ISR	..	High income: non-OECD
Italy	ITA	..	High income: OECD
Jamaica	JAM	Latin America & Caribbean	Upper middle income
Japan	JPN	..	High income: OECD
Jordan	JOR	Middle East & North Africa	Lower middle income
Kazakhstan	KAZ	Europe & Central Asia	Upper middle income
Kenya	KEN	Sub-Saharan Africa	Low income
Kiribati	KIR	East Asia & Pacific	Lower middle income
Korea, Democratic Republic	PRK	East Asia & Pacific	Low income
Korea, Republic	KOR	..	High income: OECD
Kosovo	KSV	Europe & Central Asia	Lower middle income
Kuwait	KWT	..	High income: non-OECD
Kyrgyz Republic	KGZ	Europe & Central Asia	Low income
Lao People's Democratic Republic (PDR)	LAO	East Asia & Pacific	Low income
Latvia	LVA	Europe & Central Asia	Upper middle income

Lebanon	LBN	Middle East & North Africa	Upper middle income
Lesotho	LSO	Sub-Saharan Africa	Lower middle income
Liberia	LBR	Sub-Saharan Africa	Low income
Libya	LBY	Middle East & North Africa	Upper middle income
Liechtenstein	LIE	..	High income: non-OECD
Lithuania	LTU	Europe & Central Asia	Upper middle income
Luxembourg	LUX	..	High income: OECD
Macao, China	MAC	..	High income: non-OECD
Macedonia, Former Yugoslav Republic (FYR)	MKD	Europe & Central Asia	Upper middle income
Madagascar	MDG	Sub-Saharan Africa	Low income
Malawi	MWI	Sub-Saharan Africa	Low income
Malaysia	MYS	East Asia & Pacific	Upper middle income
Maldives	MDV	South Asia	Lower middle income
Mali	MLI	Sub-Saharan Africa	Low income
Malta	MLT	..	High income: non-OECD
Marshall Islands	MHL	East Asia & Pacific	Lower middle income
Mauritania	MRT	Sub-Saharan Africa	Low income
Mauritius	MUS	Sub-Saharan Africa	Upper middle income
Mayotte	MYT	Sub-Saharan Africa	Upper middle income
Mexico	MEX	Latin America & Caribbean	Upper middle income
Micronesia, Federated States	FSM	East Asia & Pacific	Lower middle income
Moldova	MDA	Europe & Central Asia	Lower middle income
Monaco	MCO	..	High income: non-OECD
Mongolia	MNG	East Asia & Pacific	Lower middle income
Montenegro	MNE	Europe & Central Asia	Upper middle income
Morocco	MAR	Middle East & North Africa	Lower middle income
Mozambique	MOZ	Sub-Saharan Africa	Low income
Myanmar	MMR	East Asia & Pacific	Low income
Namibia	NAM	Sub-Saharan Africa	Upper middle income
Nepal	NPL	South Asia	Low income
Netherlands	NLD	..	High income: OECD
Netherlands Antilles	ANT	..	High income: non-OECD
New Caledonia	NCL	..	High income: non-OECD
New Zealand	NZL	..	High income: OECD
Nicaragua	NIC	Latin America & Caribbean	Lower middle income
Niger	NER	Sub-Saharan Africa	Low income
Nigeria	NGA	Sub-Saharan Africa	Lower middle income
Northern Mariana Islands	MNP	..	High income: non-OECD
Norway	NOR	..	High income: OECD

Oman	OMN	..	High income: non-OECD
Pakistan	PAK	South Asia	Lower middle income
Palau	PLW	East Asia & Pacific	Upper middle income
Panama	PAN	Latin America & Caribbean	Upper middle income
Papua New Guinea	PNG	East Asia & Pacific	Lower middle income
Paraguay	PRY	Latin America & Caribbean	Lower middle income
Peru	PER	Latin America & Caribbean	Upper middle income
Philippines	PHL	East Asia & Pacific	Lower middle income
Poland	POL	Europe & Central Asia	Upper middle income
Portugal	PRT	..	High income: OECD
Puerto Rico	PRI	..	High income: non-OECD
Qatar	QAT	..	High income: non-OECD
Romania	ROM	Europe & Central Asia	Upper middle income
Russian Federation	RUS	Europe & Central Asia	Upper middle income
Rwanda	RWA	Sub-Saharan Africa	Low income
Samoa	WSM	East Asia & Pacific	Lower middle income
San Marino	SMR	..	High income: non-OECD
São Tomé and Principe	STP	Sub-Saharan Africa	Lower middle income
Saudi Arabia	SAU	..	High income: non-OECD
Senegal	SEN	Sub-Saharan Africa	Low income
Serbia	SRB	Europe & Central Asia	Upper middle income
Seychelles	SYC	Sub-Saharan Africa	Upper middle income
Sierra Leone	SLE	Sub-Saharan Africa	Low income
Singapore	SGP	..	High income: non-OECD
Slovak Republic	SVK	..	High income: OECD
Slovenia	SVN	..	High income: non-OECD
Solomon Islands	SLB	East Asia & Pacific	Lower middle income
Somalia	SOM	Sub-Saharan Africa	Low income
South Africa	ZAF	Sub-Saharan Africa	Upper middle income
Spain	ESP	..	High income: OECD
Sri Lanka	LKA	South Asia	Lower middle income
St. Kitts and Nevis	KNA	Latin America & Caribbean	Upper middle income
St. Lucia	LCA	Latin America & Caribbean	Upper middle income
St. Vincent and the Grenadines	VCT	Latin America & Caribbean	Upper middle income
Sudan	SDN	Sub-Saharan Africa	Lower middle income
Suriname	SUR	Latin America & Caribbean	Upper middle income
Swaziland	SWZ	Sub-Saharan Africa	Lower middle income
Sweden	SWE	..	High income: OECD
Switzerland	CHE	..	High income: OECD

Syrian Arab Republic	SYR	Middle East & North Africa	Lower middle income
Tajikistan	TJK	Europe & Central Asia	Low income
Tanzania	TZA	Sub-Saharan Africa	Low income
Thailand	THA	East Asia & Pacific	Lower middle income
Timor-Leste	TMP	East Asia & Pacific	Lower middle income
Togo	TGO	Sub-Saharan Africa	Low income
Tonga	TON	East Asia & Pacific	Lower middle income
Trinidad and Tobago	TTO	..	High income: non-OECD
Tunisia	TUN	Middle East & North Africa	Lower middle income
Turkey	TUR	Europe & Central Asia	Upper middle income
Turkmenistan	TKM	Europe & Central Asia	Lower middle income
Uganda	UGA	Sub-Saharan Africa	Low income
Ukraine	UKR	Europe & Central Asia	Lower middle income
United Arab Emirates	ARE	..	High income: non-OECD
United Kingdom	GBR	..	High income: OECD
United States	USA	..	High income: OECD
Uruguay	URY	Latin America & Caribbean	Upper middle income
Uzbekistan	UZB	Europe & Central Asia	Low income
Vanuatu	VUT	East Asia & Pacific	Lower middle income
Venezuela, Bolivarian Republic (RB)	VEN	Latin America & Caribbean	Upper middle income
Vietnam	VNM	East Asia & Pacific	Low income
Virgin Islands (U.S.)	VIR	..	High income: non-OECD
West Bank and Gaza	WBG	Middle East & North Africa	Lower middle income
Yemen, Republic	YEM	Middle East & North Africa	Low income
Zambia	ZMB	Sub-Saharan Africa	Low income
Zimbabwe	ZWE	Sub-Saharan Africa	Low income

A.2 Municipal waste generation and food waste composition in different countries around the world

Data on municipal solid waste (MSW) generation and composition, as regards to food waste, were for numerous countries were available in the *2006 IPCC Guidelines for National Greenhouse Gas Inventories.* These data were default data for the year 2000.

Table A-2 Municipal waste generation and fraction of food waste (adopted from IPCC, 2006)

Region	MSW generation (tonnes/cap/y)	Food waste fraction (%)
	ASIA	
Eastern Asia	0.37	26.2
China	0.27	-
Japan	0.47	-
Rep of Korea	0.38	-
South-Central Asia	0.21	40.3
Bangladesh	0.18	-
India	0.17	-
Nepal	0.18	-
Sri Lanka	0.32	-
South-Eastern Asia	0.27	43.5
Indonesia	0.28	-
Lao PDR	0.25	-
Malaysia	0.30	-
Myanmar	0.16	-
Philippines	0.19	-
Singapore	0.40	-
Thailand	0.40	-
Vietnam	0.20	-
Western Asia and Middle East	-	41.1
	AFRICA	
Eastern Africa	0.29	53.9
Middle Africa	0.29	43.4
Northern Africa	0.29	51.1
Egypt	0.29	-
Southern Africa	0.29	23
Western Africa	0.29	40.4

EUROPE		
Eastern Europe	0.38	30.1
Bulgaria	0.52	-
Croatia	-	-
Czech Republic	0.33	-
Estonia	0.44	-
Hungary	0.45	-
Latvia	0.27	-
Lithuania	0.31	-
Poland	0.32	-
Romania	0.36	-
Russian Federation	0.34	-
Slovakia	0.32	-
Slovenia	0.51	-
Northern Europe	0.64	23.8
Denmark	0.67	-
Finland	0.50	-
Iceland	1.00	-
Norway	0.51	-
Sweden	0.43	-
Southern Europe	0.52	36.9
Cyprus	0.68	-
Greece	0.41	-
Italy	0.50	-
Malta	0.48	-
Portugal	0.47	-
Spain	0.60	-
Western Europe	0.56	24.2
Austria	0.58	-
Belgium	0.47	-
France	0.53	-
Germany	0.61	-
Ireland	0.60	-
Luxemburg	0.66	-
Netherlands	0.58	-
Switzerland	0.40	-
AMERICA		
Caribbean	0.49	17
Bahamas	0.95	-
Cuba	0.21	-
Dominican Republic	0.25	-

St. Lucia	0.55	-
Central America	0.21	13.7
Costa Rica	0.17	-
Guatemala	0.22	-
Honduras	0.15	-
Nicaragua	0.28	-
South America	0.26	17.1
Argentina	0.28	-
Bolivia	0.16	-
Brazil	0.18	-
Colombia	0.26	-
Ecuador	0.22	-
Paraguay	0.44	-
Peru	0.20	-
Uruguay	0.26	-
Venezuela	0.33	-
North America	0.65	23.2
Mexico	0.31	-
USA	1.14	-
Canada	0.49	-
OCEANIA		
Australia and New Zealand	0.69	36
Rest of Oceania	0.69	67.5

APPENDIX B

: Supplementary material for Chapter 5

B.1 The Anglian Water Region and the services provided

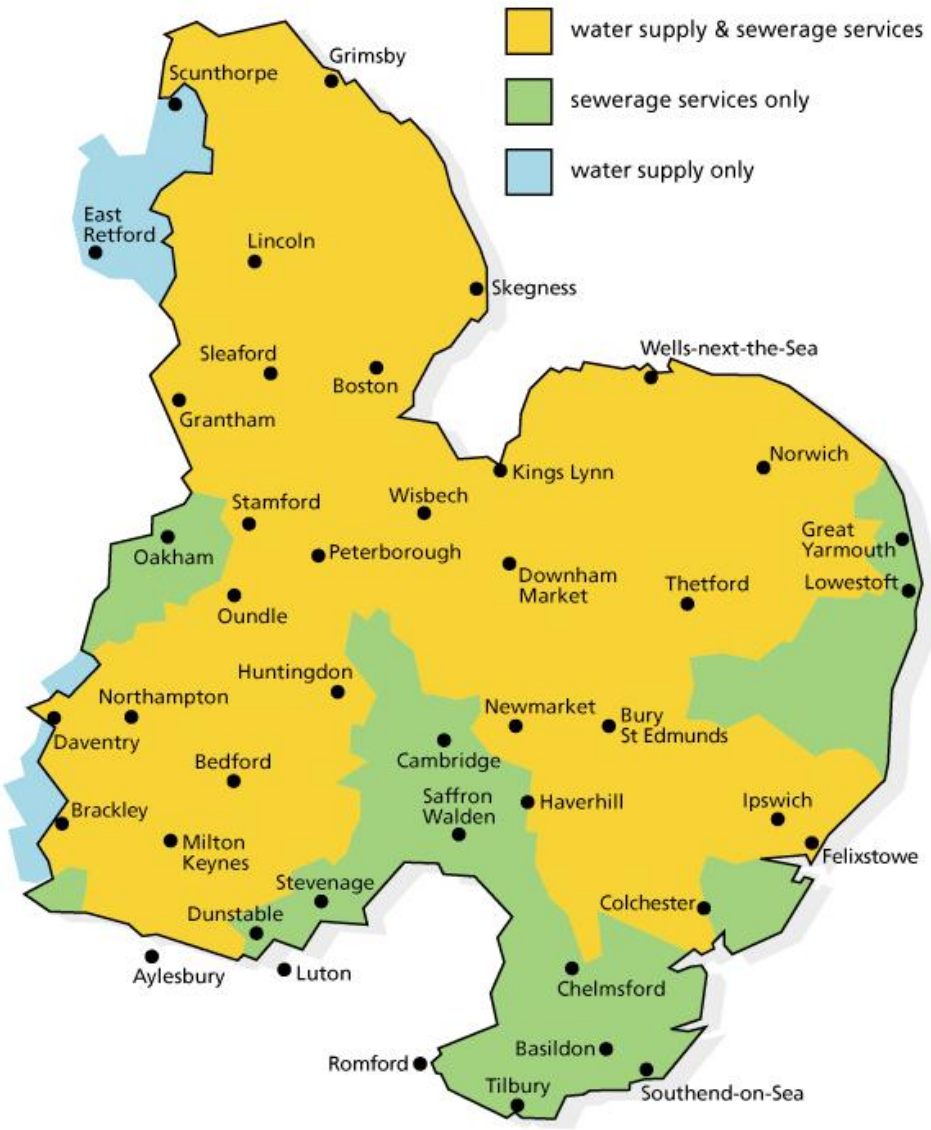


Figure B-1 The Anglian Water Services Region[4]

[4] Anglian Water (2008), *Water Resources Management Plan: Strategic Environmental Assessment*, Anglian Water Services Ltd.

B.2 Data as provided from June Return 2007-08

Table B-1 Data for water services in the Anglian region

Water properties and population		
	Unit	**2007-08**
Household properties connected during the year	000	24.016
Non household properties connected during the year	000	2.165
Households billed unmeasured water	000	707.476
Households billed measured water (external meter)	000	1,002.402
Households billed measured water (not external meter)	000	94.536
Households billed water	000	1,804.414
Household properties (water supply area)	000	1,889.934
Non-households billed unmeasured water	000	4.638
Non-households billed measured water	000	108.958
Non-households billed water	000	113.596
Non-household properties (water supply area)	000	127.826
Void properties	000	99.749
Population - households billed unmeasured water	000	1,944.09
Population - households billed measured water	000	2,193.56
Population - non-households billed unmeasured water	000	0.15
Population - non-households billed measured water	000	72.28
Population - total	000	4,210.08

Table B-2 Data for sewage services in the Anglian region

Sewage properties and population		
	Unit	**2007-08**
Households properties connected during the year	000	30.122
Non-households properties connected during the year	000	2.985
Households billed unmeasured sewage	000	927.641
Households billed measured sewage	000	1,419.694
Households billed sewage	000	2,347.335
Non-households billed unmeasured sewage	000	6.548
Non-households billed measured sewage	000	106.879
Non-households billed sewage	000	113.427
Void properties	000	114.657
Total connected population	000	5,768.351

Table B-3 Data for sewage collection and sewerage treatment in the Anglian region

Sewerage treatment						
	Unit	**2007-08**	**2007-08**	**2007-08**	**2007-08**	**2007-08**
Area		**Essex**	**Lincolnshire**	**Norfolk and Suffolk**	**Ruthamford**	**Total**
Annual average resident connected population	000	1,582.2	767.0	1,424.8	1,710.4	5,484.4
Annual average non-resident population	000	74.1	58.5	127.8	23.6	284.0
Volume of sewage collected (daily average)	Ml/d	268.3	142.6	262.2	288.7	961.8
Total connected properties	nr	742,982	360,160	669,066	803,211	2,575,419.0
Area of sewerage district	km²	4,721	6,680	10,089	5,986	27,476.0
Total length of sewer	km	12,587	6,505	11,848	12,827	43,767.0
Sewerage: direct costs	£000	5,010	4,136	8,136	4,517	21,799.0
Sewerage: power costs	£000	1,030	787	1,796	1,012	4,625.0
Sewerage: Service Charges	£000	260	214	422	234	1,130.0
Sewerage: general & support expenditure	£000	1,412	1,165	2,290	1,272	6,139.0
Sewerage: functional expenditure	£000	6,422	5,301	10,426	5,789	27,938.0

Table B-4 Data on sewage treatment and associated costs

Sewage treatment works - Costs								
	Primary	**Secondary**	**Biological**	**Tertiary**				**Total**
				A1	**A2**	**B1**	**B2**	
Direct costs of STWs in size band 1	14.000	482.000	1,120.000	293.000	19.000	398.000	0.000	2,326.000
Direct costs of STWs in size band 2	0.000	433.000	873.000	202.000	0.000	379.000	0.000	1,887.000
Direct costs of STWs in size band 3	0.000	874.000	2,916.000	596.000	50.000	2,356.000	258.000	7,050.000
Direct costs of STWs in size band 4	0.000	2,024.000	3,265.000	1,041.000	921.000	2,581.000	1,332.000	11,164.000
Direct costs of STWs in size band 5	0.000	1,422.000	733.000	1,058.000	2,796.000	187.000	3,043.000	9,239.000
Direct costs of STWs in size band 6	0.000	10,461.000	453.000	1,872.000	8,831.000	87.000	3,273.000	24,977.000
Total direct costs of STWs - all sizes	14.000	15,696.000	9,360.000	5,062.000	12,617.000	5,988.000	7,906.000	56,643.000
Sewage Treatment: Power costs	1.000	6,446.000	1,637.000	1,880.000	5,389.000	1,053.000	2,154.000	18,560.000
Sewage Treatment: service charges	4.000	1,103.000	1,736.000	416.000	646.000	984.000	691.000	5,580.000
Sewage Treatment: General and Support	3.000	4,120.000	2,153.000	1,312.000	3,385.000	1,414.000	2,038.000	14,425.000
Sewage Treatment: Functional Expenditure	17.000	19,813.000	11,513.000	6,374.000	16,006.000	7,402.000	9,943.000	71,068.000

Table B-5 Data on sewage sludge treatment and disposal and associated costs

Sludge treatment and disposal cost						
		Farmland		**Landfill**	**Land**	**Total**
	Untreated	**Conventional**	**Advanced**		**Reclamation**	
Resident population served	0.0	4,841.1	516.5	2.2	124.5	5,484.3
Amount of sewage sludge	0.0	153.8	16.4	7.6	4.0	181.8
Sludge treatment: direct costs	3.000	16,690.000	4,288.000	7.000	241.000	21,229.000
Sludge disposal: direct costs	0.000	6,042.000	431.000	1,530.000	409.000	8,412.000
Sludge treatment and disposal: direct costs	3.000	22,732.000	4,719.000	1,537.000	650.000	29,641.000
Sludge treatment and disposal: power costs	0.000	822.000	115.000	0.000	-37.000	900.000
Sludge treatment and disposal: service charges	0.000	0.000	0.000	0.000	0.000	0.000
Sludge treatment and disposal: general and support exp.	0.000	7,643.000	1,696.000	363.000	186.000	9,888.000
Sludge treatment and disposal: functional expenditure	3.000	30,375.000	6,415.000	1,900.000	836.000	39,529.000

Table B-6 Data on water abstraction, treatment and distribution costs

Water service - Activity costing analysis			
Service analysis water (£m) - Direct costs			
	Water Resources and treatment	**Water Distribution**	**Water Service Total**
Employment Costs	6.958	7.696	14.654
Power	12.462	8.789	21.251
Agencies	0.000	0.000	0.000
Hired & contracted services	2.865	6.274	9.139
Associated companies	0.000	0.000	0.000
Materials & consumables	3.535	0.367	3.902
Service charges	9.197	0.000	9.197
Bulk supply imports	1.171	0.000	1.171
Other Direct Costs	0.076	0.367	0.443
Total direct costs	36.264	23.493	59.757
General and support expenditure	7.867	8.107	15.974
Functional expenditure	44.131	31.600	75.731

B.3 Results on the impacts of the use of FWDs in the Anglian region for years 2008-35 based on the three different scenarios of FWDs penetration rate – From the Water industry perspective

Using data provided by the Anglian Water (JR, 2008) and other data adopted from the literature, the impacts of the use of FWDs to the water industry were calculated from year 2008 (base year) to year 2035 (projection year), under three FWDs penetration rate scenarios (Table 5-1, *Chapter 5*).

Table B-7 Low scenario results from year 2008 to 2035

Increase in Characteristics

	Unit	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15
Households connected to the sewerage system	-	2,347,335	2,379,024	2,411,141	2,443,691	2,476,681	2,510,116	2,347,335	2,379,024
Increase in households that have a FWD	-	117,367	6,785	13,998	21,287	28,653	36,098	117,367	6,785
Food waste generation	kg/d	1,542,199	1,563,019	1,584,120	1,605,505	1,627,179	1,649,146	1,542,199	1,563,019
Food waste diversion through the use of FWDs	kg/d	73,254	4,235	8,737	13,286	17,884	22,530	73,254	4,235
Percentage reduction in food waste landfilled	%	4.75	0.27	0.55	0.83	1.10	1.37	4.75	0.27
Wastewater flow	Ml/d	919	931	944	957	969	983	919	931
Additional wastewater flow due to the use of FWDs	Ml/d	0.91	0.05	0.11	0.16	0.22	0.28	0.91	0.05
Percentage increase in wastewater flow	%	0.10	0.01	0.01	0.02	0.02	0.03	0.10	0.01
BOD generation	kg/d	329,064	333,506	338,009	342,572	347,197	351,884	329,064	333,506
Additional BOD generation due to the use of FWDs	kg/d	7,174	415	856	1,301	1,751	2,206	7,174	415
Percentage increase in BOD generation	%	2.18	0.12	0.25	0.38	0.50	0.63	2.18	0.12
SS generation	kg/d	438,752	444,675	450,678	456,762	462,929	469,178	438,752	444,675
Additional SS generation due to the use of FWDs	kg/d	6,608	382	788	1,199	1,613	2,032	6,608	382
Percentage increase in SS generation	%	1.51	0.09	0.17	0.26	0.35	0.43	1.51	0.09
DS generation	kg/d	367,521	372,482	377,511	382,607	387,772	393,007	367,521	372,482
Additional DS generation due to the use of FWDs	kg/d	5,535	320	660	1,004	1,351	1,702	5,535	320
Percentage increase in DS generation	%	1.51	0.09	0.17	0.26	0.35	0.43	1.51	0.09
Water consumption	Ml/d	621	624	627	631	634	638	621	624
Additional water consumption due to the use of FWDs	Ml/d	0.86	0.05	0.10	0.16	0.21	0.26	0.86	0.05
Percentage increase in water consumption	%	0.14	0.01	0.02	0.02	0.03	0.04	0.14	0.01

(Continued)

2015/16	2016/17	2017/18	2018/19	2019/20	2020/21	2021/22	2022/23	2023/24	2024/25
2,544,003	2,578,347	2,613,155	2,648,432	2,684,186	2,720,422	2,757,148	2,794,370	2,832,094	2,870,327
43,621	51,226	58,911	66,680	74,532	82,470	90,494	98,606	106,806	115,097
1,671,410	1,693,974	1,716,843	1,740,020	1,763,510	1,787,318	1,811,446	1,835,901	1,860,686	1,885,805
27,226	31,972	36,770	41,618	46,519	51,474	56,482	61,545	66,663	71,838
1.63	1.89	2.14	2.39	2.64	2.88	3.12	3.35	3.58	3.81
996	1,009	1,023	1,037	1,051	1,065	1,079	1,094	1,109	1,124
0.34	0.40	0.46	0.52	0.58	0.64	0.70	0.76	0.83	0.89
0.03	0.04	0.04	0.05	0.05	0.06	0.06	0.07	0.07	0.08
356,634	361,449	366,328	371,274	376,286	381,366	386,514	391,732	397,020	402,380
2,666	3,131	3,601	4,076	4,556	5,041	5,531	6,027	6,528	7,035
0.75	0.87	0.98	1.10	1.21	1.32	1.43	1.54	1.64	1.75
475,512	481,932	488,438	495,032	501,714	508,488	515,352	522,309	529,361	536,507
2,456	2,884	3,317	3,754	4,196	4,643	5,095	5,552	6,013	6,480
0.52	0.60	0.68	0.76	0.84	0.91	0.99	1.06	1.14	1.21
398,313	403,690	409,140	414,663	420,261	425,935	431,685	437,513	443,419	449,405
2,057	2,416	2,778	3,145	3,515	3,889	4,268	4,650	5,037	5,428
0.52	0.60	0.68	0.76	0.84	0.91	0.99	1.06	1.14	1.21
641	645	648	652	656	659	663	667	670	674
0.32	0.37	0.43	0.49	0.54	0.60	0.66	0.72	0.78	0.84
0.05	0.06	0.07	0.07	0.08	0.09	0.10	0.11	0.12	0.12

(Continued)

2025/26	2026/27	2027/28	2028/29	2029/30	2030/31	2031/32	2032/33	2033/34	2034/35
2,948,349	2,988,152	3,028,492	3,069,376	3,110,813	3,152,809	3,195,372	3,238,509	3,282,229	3,326,539
131,953	140,522	149,186	157,946	166,805	175,763	184,822	193,983	203,248	212,618
1,937,065	1,963,216	1,989,719	2,016,580	2,043,804	2,071,395	2,099,359	2,127,701	2,156,425	2,185,536
82,359	87,707	93,114	98,582	104,111	109,703	115,357	121,074	126,857	132,705
4.25	4.47	4.68	4.89	5.09	5.30	5.49	5.69	5.88	6.07
1,154	1,170	1,185	1,201	1,218	1,234	1,251	1,268	1,285	1,302
1.02	1.09	1.15	1.22	1.29	1.36	1.43	1.50	1.57	1.65
0.09	0.09	0.10	0.10	0.11	0.11	0.11	0.12	0.12	0.13
413,318	418,898	424,553	430,284	436,093	441,980	447,947	453,994	460,123	466,335
8,065	8,589	9,119	9,654	10,196	10,743	11,297	11,857	12,423	12,996
1.95	2.05	2.15	2.24	2.34	2.43	2.52	2.61	2.70	2.79
551,090	558,530	566,070	573,712	581,457	589,307	597,263	605,326	613,498	621,780
7,429	7,912	8,400	8,893	9,392	9,896	10,406	10,922	11,443	11,971
1.35	1.42	1.48	1.55	1.62	1.68	1.74	1.80	1.87	1.93
461,621	467,853	474,169	480,570	487,058	493,633	500,297	507,051	513,896	520,834
6,223	6,627	7,036	7,449	7,867	8,289	8,717	9,149	9,586	10,027
1.35	1.42	1.48	1.55	1.62	1.68	1.74	1.80	1.87	1.93
681	685	689	693	696	700	704	708	712	716
0.96	1.03	1.09	1.15	1.22	1.28	1.35	1.42	1.48	1.55
0.14	0.15	0.16	0.17	0.17	0.18	0.19	0.20	0.21	0.22

(Continued) Increase in cost

	Unit	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13
Cost of water consumption	£000	43,318	43,556	43,796	44,037	44,279	44,522
Cost of additional water consumption due to the use of FWDs	£000	59.8	3.5	7.1	10.8	14.6	18.4
Percentage increase in cost of water consumption	%	0.14	0.01	0.02	0.02	0.03	0.04
Cost of wastewater treatment (per tonne BOD)	£000	58,398	59,186	59,985	60,795	61,616	62,448
Cost of additional wastewater treatment due to the use of FWDs (per tonne BOD)	£000	1,273.1	73.6	151.8	230.9	310.8	391.6
Percentage increase in cost of wastewater treatment	%	2.18	0.12	0.25	0.38	0.50	0.63
Cost of sludge treatment and disposal (per tonne DS)	£000	29,167	29,561	29,960	30,365	30,775	31,190
Cost of additional sludge treatment and disposal due to the use of FWDs (per tonne DS)	£000	439.3	25.4	52.4	79.7	107.2	135.1
Percentage increase in cost of sludge treatment and disposal	%	1.51	0.09	0.17	0.26	0.35	0.43
Total cost	£000	130,883	132,304	133,741	135,196	136,669	138,160
Total extra cost due to the use of FWDs	£000	1,772.2	102.5	211.4	321.4	432.7	545.1
% increase in cost	%	1.35	0.08	0.16	0.24	0.32	0.39

(Continued)

2013/14	2014/15	2015/16	2016/17	2017/18	2018/19	2019/20	2020/21	2021/22	2022/23	2023/24
44,767	45,013	45,261	45,510	45,760	46,012	46,265	46,519	46,775	47,020	47,271
22.2	26.1	30.0	34.0	38.0	42.0	46.1	50.3	54.4	58.3	62.3
0.05	0.06	0.07	0.07	0.08	0.09	0.10	0.11	0.12	0.12	0.13
63,291	64,145	65,011	65,889	66,778	67,680	68,594	69,520	70,458	71,297	72,193
473.2	555.7	639.0	723.3	808.5	894.6	981.6	1,069.6	1,158.5	1,239.9	1,325.5
0.75	0.87	0.98	1.10	1.21	1.32	1.43	1.54	1.64	1.74	1.84
31,611	32,038	32,470	32,909	33,353	33,803	34,260	34,722	35,191	35,610	36,057
163.3	191.7	220.5	249.6	279.0	308.7	338.7	369.1	399.8	427.8	457.4
0.52	0.60	0.68	0.76	0.84	0.91	0.99	1.06	1.14	1.20	1.27
139,669	141,196	142,742	144,307	145,891	147,495	149,118	150,761	152,424	153,927	155,521
658.7	773.5	889.5	1,006.9	1,125.4	1,245.3	1,366.4	1,488.9	1,612.7	1,725.9	1,845.2
0.47	0.55	0.62	0.70	0.77	0.84	0.92	0.99	1.06	1.12	1.19

(Continued)

2024/25	2025/26	2026/27	2027/28	2028/29	2029/30	2030/31	2031/32	2032/33	2033/34	2034/35
47,551	47,813	48,076	48,340	48,606	48,873	49,142	49,413	49,684	49,958	50,232
67.3	71.6	76.0	80.5	85.0	89.6	94.2	98.9	103.6	108.4	113.2
0.14	0.15	0.16	0.17	0.17	0.18	0.19	0.20	0.21	0.22	0.23
73,350	74,341	75,344	76,361	77,392	78,437	79,496	80,569	81,657	82,759	83,876
1,431.3	1,524.3	1,618.2	1,713.3	1,809.4	1,906.5	2,004.8	2,104.2	2,204.7	2,306.3	2,409.1
1.95	2.05	2.15	2.24	2.34	2.43	2.52	2.61	2.70	2.79	2.87
36,635	37,130	37,631	38,139	38,654	39,176	39,705	40,241	40,784	41,335	41,893
493.9	526.0	558.4	591.2	624.3	657.9	691.8	726.1	760.7	795.8	831.3
1.35	1.42	1.48	1.55	1.62	1.68	1.74	1.80	1.87	1.93	1.98
157,537	159,283	161,051	162,841	164,652	166,486	168,343	170,222	172,125	174,051	176,001
1,992.5	2,121.8	2,252.7	2,385.0	2,518.7	2,654.0	2,790.8	2,929.1	3,069.0	3,210.5	3,353.6
1.26	1.33	1.40	1.46	1.53	1.59	1.66	1.72	1.78	1.84	1.91

Table B-8 Medium scenario results from year 2008 to 2035

Increase in Characteristics

	Unit	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15
Households connected to the sewerage system		2,347,335	2,379,024	2,411,141	2,443,691	2,476,681	2,510,116	2,544,003	2,578,347
Increase in households that have a FWD		117,367	22,617	46,502	70,474	94,536	118,692	142,945	167,299
Food waste generation	kg/d	1,542,199	1,563,019	1,584,120	1,605,505	1,627,179	1,649,146	1,671,410	1,693,974
Food waste diversion through the use of FWDs	kg/d	73,254	14,116	29,024	43,986	59,005	74,081	89,219	104,419
Percentage reduction in food waste landfilled	%	4.75	0.90	1.83	2.74	3.63	4.49	5.34	6.16
Wastewater flow	Ml/d	919	931	944	957	969	983	996	1,009
Additional wastewater flow due to the use of FWDs	Ml/d	0.91	0.18	0.36	0.55	0.73	0.92	1.11	1.29
Percentage increase in wastewater flow	%	0.10	0.02	0.04	0.06	0.08	0.09	0.11	0.13
BOD generation	kg/d	329,064	333,506	338,009	342,572	347,197	351,884	356,634	361,449
Additional BOD generation due to the use of FWDs	kg/d	7,174	1,382	2,842	4,308	5,778	7,255	8,737	10,226
Percentage increase in BOD generation	%	2.18	0.41	0.84	1.26	1.66	2.06	2.45	2.83
SS generation	kg/d	438,752	444,675	450,678	456,762	462,929	469,178	475,512	481,932
Additional SS generation due to the use of FWDs	kg/d	6,608	1,273	2,618	3,968	5,323	6,683	8,048	9,419
Percentage increase in SS generation	%	1.51	0.29	0.58	0.87	1.15	1.42	1.69	1.95
DS generation	kg/d	367,521	372,482	377,511	382,607	387,772	393,007	398,313	403,690
Additional DS generation due to the use of FWDs	kg/d	5,535	1,067	2,193	3,324	4,458	5,598	6,742	7,890
Percentage increase in DS generation	%	1.51	0.29	0.58	0.87	1.15	1.42	1.69	1.95
Water consumption	Ml/d	620.65	624.06	627.49	630.94	634.41	637.90	641.41	644.94
Additional water consumption due to the use of FWDs	Ml/d	0.86	0.17	0.34	0.51	0.69	0.87	1.04	1.22
Percentage increase in water consumption	%	0.14	0.03	0.05	0.08	0.11	0.14	0.16	0.19

(Continued)

2015/16	2016/17	2017/18	2018/19	2019/20	2020/21	2021/22	2022/23	2023/24	2024/25
2,613,155	2,648,432	2,684,186	2,720,422	2,757,148	2,794,370	2,832,094	2,870,327	2,909,076	2,948,349
191,757	216,324	241,003	265,797	290,710	315,747	340,910	366,205	391,633	417,200
1,716,843	1,740,020	1,763,510	1,787,318	1,811,446	1,835,901	1,860,686	1,885,805	1,911,263	1,937,065
119,685	135,019	150,422	165,897	181,447	197,073	212,779	228,567	244,438	260,396
6.97	7.76	8.53	9.28	10.02	10.73	11.44	12.12	12.79	13.44
1,023	1,037	1,051	1,065	1,079	1,094	1,109	1,124	1,139	1,154
1.48	1.67	1.87	2.06	2.25	2.44	2.64	2.83	3.03	3.23
0.15	0.16	0.18	0.19	0.21	0.22	0.24	0.25	0.27	0.28
366,328	371,274	376,286	381,366	386,514	391,732	397,020	402,380	407,812	413,318
11,721	13,222	14,731	16,246	17,769	19,299	20,837	22,383	23,938	25,500
3.20	3.56	3.91	4.26	4.60	4.93	5.25	5.56	5.87	6.17
488,438	495,032	501,714	508,488	515,352	522,309	529,361	536,507	543,750	551,090
10,796	12,180	13,569	14,965	16,368	17,777	19,194	20,618	22,050	23,490
2.21	2.46	2.70	2.94	3.18	3.40	3.63	3.84	4.06	4.26
409,140	414,663	420,261	425,935	431,685	437,513	443,419	449,405	455,472	461,621
9,044	10,202	11,366	12,535	13,710	14,891	16,078	17,271	18,470	19,676
2.21	2.46	2.70	2.94	3.18	3.40	3.63	3.84	4.06	4.26
648.49	652.05	655.64	659.25	662.87	666.52	670.18	673.87	677.58	681.30
1.40	1.58	1.76	1.94	2.12	2.31	2.49	2.67	2.86	3.05
0.22	0.24	0.27	0.29	0.32	0.35	0.37	0.40	0.42	0.45

(Continued)

2025/26	2026/27	2027/28	2028/29	2029/30	2030/31	2031/32	2032/33	2033/34	2034/35
2,988,152	3,028,492	3,069,376	3,110,813	3,152,809	3,195,372	3,238,509	3,282,229	3,326,539	3,371,448
442,910	468,766	494,772	520,932	547,251	573,732	600,380	627,199	654,192	681,365
1,963,216	1,989,719	2,016,580	2,043,804	2,071,395	2,099,359	2,127,701	2,156,425	2,185,536	2,215,041
276,442	292,580	308,812	325,140	341,567	358,095	374,727	391,466	408,314	425,274
14.08	14.70	15.31	15.91	16.49	17.06	17.61	18.15	18.68	19.20
1,170	1,185	1,201	1,218	1,234	1,251	1,268	1,285	1,302	1,320
3.43	3.63	3.83	4.03	4.24	4.44	4.65	4.85	5.06	5.27
0.29	0.31	0.32	0.33	0.34	0.36	0.37	0.38	0.39	0.40
418,898	424,553	430,284	436,093	441,980	447,947	453,994	460,123	466,335	472,630
27,072	28,652	30,242	31,841	33,449	35,068	36,697	38,336	39,986	41,647
6.46	6.75	7.03	7.30	7.57	7.83	8.08	8.33	8.57	8.81
558,530	566,070	573,712	581,457	589,307	597,263	605,326	613,498	621,780	630,174
24,937	26,393	27,857	29,330	30,812	32,303	33,803	35,313	36,833	38,363
4.46	4.66	4.86	5.04	5.23	5.41	5.58	5.76	5.92	6.09
467,853	474,169	480,570	487,058	493,633	500,297	507,051	513,896	520,834	527,865
20,889	22,108	23,334	24,568	25,809	27,058	28,315	29,580	30,853	32,134
4.46	4.66	4.86	5.04	5.23	5.41	5.58	5.76	5.92	6.09
685.05	688.82	692.61	696.42	700.25	704.10	707.97	711.86	715.78	719.72
3.23	3.42	3.61	3.80	4.00	4.19	4.38	4.58	4.78	4.98
0.47	0.50	0.52	0.55	0.57	0.60	0.62	0.64	0.67	0.69

(Continued) Increase in cost

	Unit	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13
Cost of water consumption	£000	43,318	43,556	43,796	44,037	44,279	44,522
Cost of additional water consumption due to the use of FWDs	£000	59.82	11.53	23.70	35.92	48.18	60.49
Percentage increase in cost of water consumption	%	0.14	0.03	0.05	0.08	0.11	0.14
Cost of wastewater treatment (per tonne BOD)	£000	58,398	59,186	59,985	60,795	61,616	62,448
Cost of additional wastewater treatment due to the use of FWDs (per tonne BOD)	£000	1,273	245	504	764	1,025	1,287
Percentage increase in cost of wastewater treatment	%	2.18	0.41	0.84	1.26	1.66	2.06
Cost of sludge treatment and disposal (per tonne DS)	£000	29,167	29,561	29,960	30,365	30,775	31,190
Cost of additional sludge treatment and disposal due to the use of FWDs (per tonne DS)	£000	439	85	174	264	354	444
Percentage increase in cost of sludge treatment and disposal	%	1.51	0.29	0.58	0.87	1.15	1.42
Total cost	£000	130,883	132,304	133,741	135,196	136,669	138,160
Total extra cost due to the use of FWDs	£000	1,772	342	702	1,064	1,427	1,792
% increase in cost	%	1.35	0.26	0.53	0.79	1.04	1.30

(Continued)

2013/14	2014/15	2015/16	2016/17	2017/18	2018/19	2019/20	2020/21	2021/22	2022/23	2023/24
44,767	45,013	45,261	45,510	45,760	46,012	46,265	46,519	46,775	47,033	47,291
72.86	85.27	97.73	110.26	122.83	135.47	148.17	160.93	173.76	186.65	199.61
0.16	0.19	0.22	0.24	0.27	0.29	0.32	0.35	0.37	0.40	0.42
63,291	64,145	65,011	65,889	66,778	67,680	68,594	69,520	70,458	71,409	72,373
1,551	1,815	2,080	2,347	2,614	2,883	3,153	3,425	3,698	3,972	4,248
2.45	2.83	3.20	3.56	3.91	4.26	4.60	4.93	5.25	5.56	5.87
31,611	32,038	32,470	32,909	33,353	33,803	34,260	34,722	35,191	35,666	36,147
535	626	718	810	902	995	1,088	1,182	1,276	1,371	1,466
1.69	1.95	2.21	2.46	2.70	2.94	3.18	3.40	3.63	3.84	4.06
139,669	141,196	142,742	144,307	145,891	147,495	149,118	150,761	152,424	154,108	155,812
2,158	2,526	2,895	3,266	3,639	4,013	4,390	4,768	5,148	5,530	5,914
1.55	1.79	2.03	2.26	2.49	2.72	2.94	3.16	3.38	3.59	3.80

(Continued)

2024/25	2025/26	2026/27	2027/28	2028/29	2029/30	2030/31	2031/32	2032/33	2033/34	2034/35
47,551	47,813	48,076	48,340	48,606	48,873	49,142	49,413	49,684	49,958	50,232
212.64	225.74	238.92	252.18	265.51	278.92	292.42	306.00	319.67	333.43	347.28
0.45	0.47	0.50	0.52	0.55	0.57	0.60	0.62	0.64	0.67	0.69
73,350	74,341	75,344	76,361	77,392	78,437	79,496	80,569	81,657	82,759	83,876
4,525	4,804	5,085	5,367	5,651	5,936	6,223	6,512	6,803	7,096	7,391
6.17	6.46	6.75	7.03	7.30	7.57	7.83	8.08	8.33	8.57	8.81
36,635	37,130	37,631	38,139	38,654	39,176	39,705	40,241	40,784	41,335	41,893
1,562	1,658	1,755	1,852	1,950	2,048	2,147	2,247	2,348	2,449	2,550
4.26	4.46	4.66	4.86	5.04	5.23	5.41	5.58	5.76	5.92	6.09
157,537	159,283	161,051	162,841	164,652	166,486	168,343	170,222	172,125	174,051	176,001
6,300	6,688	7,078	7,471	7,866	8,263	8,663	9,066	9,471	9,878	10,288
4.00	4.20	4.40	4.59	4.78	4.96	5.15	5.33	5.50	5.68	5.85

Table B-9 High scenario results from year 2008 to 2035

Increase in Characteristics

	Unit	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15
Households connected to the sewerage system	-	2,347,335	2,379,024	2,411,141	2,443,691	2,476,681	2,510,116	2,544,003	2,578,347
Increase in households that have a FWD	-	117,367	339,249	650,032	919,081	1,152,721	1,356,330	1,534,481	1,691,061
Food waste generation	kg/d	1,542,199	1,563,019	1,584,120	1,605,505	1,627,179	1,649,146	1,671,410	1,693,974
Food waste diversion through the use of FWDs	kg/d	73,254	211,742	405,718	573,645	719,471	846,554	957,747	1,055,476
Percentage reduction in food waste landfilled	%	4.75	13.55	25.61	35.73	44.22	51.33	57.30	62.31
Wastewater flow	Ml/d	918.80	931.21	943.78	956.52	969.43	982.52	995.78	1,009.22
Additional wastewater flow due to the use of FWDs	Ml/d	0.91	2.63	5.03	7.11	8.92	10.50	11.88	13.09
Percentage increase in wastewater flow	%	0.10	0.28	0.53	0.74	0.92	1.07	1.19	1.30
BOD generation	kg/d	329,064	333,506	338,009	342,572	347,197	351,884	356,634	361,449
Additional BOD generation due to the use of FWDs	kg/d	7,174	20,736	39,732	56,176	70,457	82,902	93,791	103,362
Percentage increase in BOD generation	%	2.18	6.22	11.75	16.40	20.29	23.56	26.30	28.60
SS generation	kg/d	438,752	444,675	450,678	456,762	462,929	469,178	475,512	481,932
Additional SS generation due to the use of FWDs	kg/d	6,608	19,101	36,599	51,747	64,901	76,365	86,395	95,211
Percentage increase in SS generation	%	1.51	4.30	8.12	11.33	14.02	16.28	18.17	19.76
DS generation	kg/d	367,521	372,482	377,511	382,607	387,772	393,007	398,313	403,690
Additional DS generation due to the use of FWDs	kg/d	5,535	16,000	30,657	43,346	54,365	63,967	72,369	79,754
Percentage increase in DS generation	%	1.51	4.30	8.12	11.33	14.02	16.28	18.17	19.76
Water consumption	Ml/d	620.65	624.06	627.49	630.94	634.41	637.90	641.41	644.94
Additional water consumption due to the use of FWDs	Ml/d	0.86	2.48	4.75	6.71	8.42	9.90	11.21	12.35
Percentage increase in water consumption	%	0.14	0.40	0.76	1.06	1.33	1.55	1.75	1.91

(Continued)

2015/16	2016/17	2017/18	2018/19	2019/20	2020/21	2021/22	2022/23	2023/24	2024/25
2,613,155	2,648,432	2,684,186	2,720,422	2,757,148	2,794,370	2,832,094	2,870,327	2,909,076	2,948,349
1,829,375	1,952,234	2,062,027	2,160,786	2,250,240	2,331,860	2,406,895	2,476,410	2,541,310	2,602,366
1,716,843	1,740,020	1,763,510	1,787,318	1,811,446	1,835,901	1,860,686	1,885,805	1,911,263	1,937,065
1,141,805	1,218,487	1,287,014	1,348,655	1,404,487	1,455,430	1,502,263	1,545,651	1,586,158	1,624,266
66.51	70.03	72.98	75.46	77.53	79.28	80.74	81.96	82.99	83.85
1,022.85	1,036.66	1,050.65	1,064.84	1,079.21	1,093.78	1,108.55	1,123.51	1,138.68	1,154.05
14.16	15.11	15.96	16.72	17.42	18.05	18.63	19.17	19.67	20.14
1.38	1.46	1.52	1.57	1.61	1.65	1.68	1.71	1.73	1.75
366,328	371,274	376,286	381,366	386,514	391,732	397,020	402,380	407,812	413,318
111,816	119,325	126,036	132,072	137,540	142,529	147,115	151,364	155,331	159,063
30.52	32.14	33.49	34.63	35.58	36.38	37.05	37.62	38.09	38.48
488,438	495,032	501,714	508,488	515,352	522,309	529,361	536,507	543,750	551,090
102,999	109,916	116,098	121,658	126,695	131,290	135,515	139,429	143,083	146,520
21.09	22.20	23.14	23.93	24.58	25.14	25.60	25.99	26.31	26.59
409,140	414,663	420,261	425,935	431,685	437,513	443,419	449,405	455,472	461,621
86,277	92,071	97,249	101,907	106,126	109,975	113,514	116,792	119,853	122,733
21.09	22.20	23.14	23.93	24.58	25.14	25.60	25.99	26.31	26.59
648.49	652.05	655.64	659.25	662.87	666.52	670.18	673.87	677.58	681.30
13.36	14.26	15.06	15.78	16.43	17.03	17.58	18.08	18.56	19.00
2.06	2.19	2.30	2.39	2.48	2.55	2.62	2.68	2.74	2.79

(Continued)

2025/26	2026/27	2027/28	2028/29	2029/30	2030/31	2031/32	2032/33	2033/34	2034/35
2,988,152	3,028,492	3,069,376	3,110,813	3,152,809	3,195,372	3,238,509	3,282,229	3,326,539	3,371,448
2,660,233	2,715,472	2,768,558	2,819,896	2,869,833	2,918,664	2,966,641	3,013,979	3,060,863	3,107,451
1,963,216	1,989,719	2,016,580	2,043,804	2,071,395	2,099,359	2,127,701	2,156,425	2,185,536	2,215,041
1,660,385	1,694,862	1,727,995	1,760,038	1,791,206	1,821,684	1,851,629	1,881,175	1,910,438	1,939,515
84.57	85.18	85.69	86.12	86.47	86.77	87.02	87.24	87.41	87.56
1,169.63	1,185.42	1,201.43	1,217.64	1,234.08	1,250.74	1,267.63	1,284.74	1,302.08	1,319.66
20.59	21.02	21.43	21.82	22.21	22.59	22.96	23.33	23.69	24.05
1.76	1.77	1.78	1.79	1.80	1.81	1.81	1.82	1.82	1.82
418,898	424,553	430,284	436,093	441,980	447,947	453,994	460,123	466,335	472,630
162,600	165,976	169,221	172,359	175,411	178,396	181,328	184,222	187,087	189,935
38.82	39.09	39.33	39.52	39.69	39.83	39.94	40.04	40.12	40.19
558,530	566,070	573,712	581,457	589,307	597,263	605,326	613,498	621,780	630,174
149,778	152,888	155,877	158,768	161,579	164,329	167,030	169,695	172,335	174,958
26.82	27.01	27.17	27.31	27.42	27.51	27.59	27.66	27.72	27.76
467,853	474,169	480,570	487,058	493,633	500,297	507,051	513,896	520,834	527,865
125,462	128,067	130,571	132,992	135,347	137,650	139,913	142,145	144,356	146,553
26.82	27.01	27.17	27.31	27.42	27.51	27.59	27.66	27.72	27.76
685.05	688.82	692.61	696.42	700.25	704.10	707.97	711.86	715.78	719.72
19.43	19.83	20.22	20.59	20.96	21.31	21.66	22.01	22.35	22.69
2.84	2.88	2.92	2.96	2.99	3.03	3.06	3.09	3.12	3.15

(Continued) Increase in cost

	Unit	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13
Cost of water consumption	£000	43,318	43,556	43,796	44,037	44,279	44,522
Cost of additional water consumption due to the use of FWDs	£000	59.82	172.91	331.31	468.44	587.52	691.29
Percentage increase in cost of water consumption	%	0.14	0.40	0.76	1.06	1.33	1.55
Cost of wastewater treatment (per tonne BOD)	£000	58,398	59,186	59,985	60,795	61,616	62,448
Cost of additional wastewater treatment due to the use of FWDs (per tonne BOD)	£000	1,273	3,680	7,051	9,969	12,504	14,712
Percentage increase in cost of wastewater treatment	%	2.18	6.22	11.75	16.40	20.29	23.56
Cost of sludge treatment and disposal (per tonne DS)	£000	29,167	29,561	29,960	30,365	30,775	31,190
Cost of additional sludge treatment and disposal due to the use of FWDs (per tonne DS)	£000	439	1,270	2,433	3,440	4,315	5,077
Percentage increase in cost of sludge treatment and disposal	%	1.51	4.30	8.12	11.33	14.02	16.28
Total cost	£000	130,883	132,304	133,741	135,196	136,669	138,160
Total extra cost due to the use of FWDs	£000	1,772	5,123	9,815	13,878	17,406	20,480
% increase in cost	%	1.35	3.87	7.34	10.27	12.74	14.82

(Continued)

2013/14	2014/15	2015/16	2016/17	2017/18	2018/19	2019/20	2020/21	2021/22	2022/23	2023/24
45,013	45,261	45,510	45,760	46,012	46,265	46,519	46,775	47,033	45,013	45,261
861.90	932.39	995.01	1,050.97	1,101.31	1,146.90	1,188.50	1,226.74	1,262.18	861.90	932.39
1.91	2.06	2.19	2.30	2.39	2.48	2.55	2.62	2.68	1.91	2.06
64,145	65,011	65,889	66,778	67,680	68,594	69,520	70,458	71,409	64,145	65,011
18,343	19,844	21,176	22,367	23,438	24,409	25,294	26,108	26,862	18,343	19,844
28.60	30.52	32.14	33.49	34.63	35.58	36.38	37.05	37.62	28.60	30.52
32,038	32,470	32,909	33,353	33,803	34,260	34,722	35,191	35,666	32,038	32,470
6,329	6,847	7,307	7,718	8,088	8,422	8,728	9,009	9,269	6,329	6,847
19.76	21.09	22.20	23.14	23.93	24.58	25.14	25.60	25.99	19.76	21.09
141,196	142,742	144,307	145,891	147,495	149,118	150,761	152,424	154,108	141,196	142,742
25,535	27,623	29,478	31,136	32,627	33,978	35,211	36,344	37,393	25,535	27,623
18.08	19.35	20.43	21.34	22.12	22.79	23.36	23.84	24.26	18.08	19.35

(Continued)

2024/25	2025/26	2026/27	2027/28	2028/29	2029/30	2030/31	2031/32	2032/33	2033/34	2034/35
47,551	47,813	48,076	48,340	48,606	48,873	49,142	49,413	49,684	49,958	50,232
1,326.37	1,355.87	1,384.02	1,411.08	1,437.24	1,462.69	1,487.58	1,512.04	1,536.16	1,560.06	1,583.80
2.79	2.84	2.88	2.92	2.96	2.99	3.03	3.06	3.09	3.12	3.15
73,350	74,341	75,344	76,361	77,392	78,437	79,496	80,569	81,657	82,759	83,876
28,228	28,856	29,455	30,031	30,588	31,130	31,659	32,180	32,693	33,202	33,707
38.48	38.82	39.09	39.33	39.52	39.69	39.83	39.94	40.04	40.12	40.19
36,635	37,130	37,631	38,139	38,654	39,176	39,705	40,241	40,784	41,335	41,893
9,740	9,957	10,164	10,362	10,555	10,741	10,924	11,104	11,281	11,456	11,631
26.59	26.82	27.01	27.17	27.31	27.42	27.51	27.59	27.66	27.72	27.76
157,537	159,283	161,051	162,841	164,652	166,486	168,343	170,222	172,125	174,051	176,001
39,295	40,169	41,003	41,805	42,580	43,334	44,071	44,796	45,510	46,218	46,922
24.94	25.22	25.46	25.67	25.86	26.03	26.18	26.32	26.44	26.55	26.66

B.4 Cost of waste collection and disposal in the Anglian Region

The costs of waste collection and disposal (Table 5-2, *Chapter 5*), were the average costs calculated from data provided by the LAs in the Anglian Region. These data were collected from the Audit Commission's Best value performance indicators (BVPIs) on waste and cleanliness, which were last reported for 2007-08 (Table B-10). It should by highlighted, that in April 2010, the National Indicator set (NIS) came into force, which replaced the BVPIs.[5]

The BVPIs used for calculating the average cost of waste collection and disposal in the Anglian region were,

- BV86: cost of household waste collection per household (£)
- BV87: cost of household waste disposal per tonne municipal waste (£)

Table B-10 Data collected from BVPIs for LAs in the Anglian region for calculating the average cost of waste collection and disposal

Authority Short Name	BV 86	BV 87
Babergh	30.19	n/a
Basildon	54.42	n/a
Bedford	57.62	n/a
Bedfordshire	n/a	56.59
Boston	41.04	n/a
Braintree	65.34	n/a
Breckland	25.28	n/a
Brentwood	53.04	n/a
Broadland	30.51	n/a
Cambridge	46.49	n/a
Cambridgeshire	n/a	50.58
Chelmsford	67.72	n/a
Colchester	53.17	n/a
Corby	73.80	n/a

[5] Audit Commission (2012), Best Value Performance Indicators, available at: http://www.audit-commission.gov.uk/performance-information/performance-data-collections-and-guidance/Pages/best-value-performance-indicators.aspx

Daventry	84.90	n/a
East Cambridgeshire	59.96	n/a
East Northamptonshire	48.58	n/a
Essex	n/a	64.01
Great Yarmouth	34.85	n/a
Hertfordshire	n/a	47.49
Huntingdonshire	52.95	n/a
Ipswich	36.56	n/a
Kings Lynn	44.63	n/a
Lincoln	59.11	n/a
Lincolnshire	n/a	56.38
Luton	53.71	55.45
Mid Beds	51.81	n/a
Mid Suffolk	49.75	n/a
Milton Keynes	72.43	47.34
Norfolk	n/a	63.10
North East Lincolnshire	37.10	71.25
North Hertfordshire	59.15	n/a
North Kesteven	52.29	n/a
North Norfolk	42.47	n/a
Northampton	46.16	n/a
Northamptonshire	n/a	44.34
Norwich	59.54	n/a
Peterborough	35.73	30.74
Rochford	43.55	n/a
Rutland	60.29	51.52
South Bedfordshire	47.29	n/a
South Cambridgeshire	47.31	n/a
South Kesteven	49.48	n/a
South Norfolk	45.32	n/a
South Northamptonshire	65.20	n/a
Southend	35.05	62.86
St Edmundsbury	44.28	n/a
Stevenage	55.04	n/a
Suffolk	n/a	50.68
Suffolk Coastal	35.86	n/a
Uttlesford	62.37	n/a
Waveney	37.04	n/a
Average:	***50.20***	***53.70***

B.5 Results on the impacts of the use of FWDs in the Anglian region for years 2008-35 based on the three different scenarios of FWDs penetration rate – From the LAs perspective

Using data collected from the Audit Commission (BVPI, 2008) and other adopted from the literature, the impacts of the use of FWDs to LAs in the Anglian region, were calculated from year 2008 (base year) to year 2035 (projection year), under three FWDs penetration rate scenarios (Table 5-1, *Chapter 5*).

Table B-11 Low scenario results from year 2008 to 2035

	Unit	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15
Households with FWD	‘000	117,367	6,785	13,998	21,287	28,653	36,098	43,621	51,226
Residual waste generation	tonnes/year	1,532,810	1,553,503	1,574,475	1,595,730	1,617,273	1,639,106	1,661,234	1,683,661
Food waste generation	tonnes/year	563,360	570,966	578,674	586,486	594,403	602,428	610,561	618,803
Food waste diversion from landfill though the use of FWDs	tonnes/year	26,760	1,547	3,192	4,853	6,533	8,230	9,946	11,679
Cost of residual waste collection	£000	117,836	119,427	121,039	122,673	124,329	126,008	127,709	129,433
Cost of disposal of the residual waste collected	£000	82,312	83,423	84,549	85,691	86,848	88,020	89,208	90,413
Cost of disposal of the residual waste collected due to the use of FWDs	£000	80,875	83,340	84,378	85,430	86,497	87,578	88,674	89,785
Percentage reduction in cost of waste disposal	%	1.75	0.10	0.20	0.30	0.40	0.50	0.60	0.69
Total cost to LAs	£000	200,148	202,850	205,589	208,364	211,177	214,028	216,917	219,846
Total cost to LAs with the use of FWDs	£000	198,711	202,767	205,417	208,103	210,826	213,586	216,383	219,218
Percentage reduction in total cost	%	0.72	0.04	0.08	0.13	0.17	0.21	0.25	0.29
Savings	£000	1,436.99	83.07	171.39	260.63	350.82	441.97	534.08	627.19

(Continued)

2015/16	2016/17	2017/18	2018/19	2019/20	2020/21	2021/22	2022/23	2023/24	2024/25
58,911	66,680	74,532	82,470	90,494	98,606	106,806	115,097	123,479	131,953
1,706,390	1,729,426	1,752,773	1,776,436	1,800,418	1,824,723	1,849,357	1,874,324	1,899,627	1,925,272
627,157	635,624	644,205	652,901	661,716	670,649	679,702	688,878	698,178	707,604
13,432	15,203	16,993	18,803	20,633	22,482	24,352	26,242	28,153	30,085
131,180	132,951	134,746	136,565	138,409	140,277	142,171	144,090	146,036	148,007
91,633	92,870	94,124	95,395	96,682	97,988	99,310	100,651	102,010	103,387
90,912	92,054	93,211	94,385	95,574	96,780	98,003	99,242	100,498	101,772
0.79	0.88	0.97	1.06	1.15	1.23	1.32	1.40	1.48	1.56
222,813	225,821	228,870	231,960	235,091	238,265	241,482	244,742	248,046	251,394
222,092	225,005	227,958	230,950	233,983	237,058	240,174	243,332	246,534	249,779
0.32	0.36	0.40	0.44	0.47	0.51	0.54	0.58	0.61	0.64
721.29	816.40	912.55	1,009.73	1,107.97	1,207.29	1,307.69	1,409.20	1,511.82	1,615.58

(Continued)

2025/26	2026/27	2027/28	2028/29	2029/30	2030/31	2031/32	2032/33	2033/34	2034/35
140,522	149,186	157,946	166,805	175,763	184,822	193,983	203,248	212,618	222,094
1,951,263	1,977,605	2,004,303	2,031,361	2,058,784	2,086,578	2,114,747	2,143,296	2,172,230	2,201,555
717,156	726,838	736,650	746,595	756,674	766,889	777,242	787,735	798,369	809,147
32,039	34,014	36,012	38,032	40,074	42,139	44,228	46,340	48,477	50,637
150,005	152,030	154,083	156,163	158,271	160,408	162,573	164,768	166,992	169,247
104,783	106,197	107,631	109,084	110,557	112,049	113,562	115,095	116,649	118,224
103,062	104,371	105,697	107,042	108,405	109,786	111,187	112,606	114,046	115,504
1.64	1.72	1.80	1.87	1.95	2.02	2.09	2.16	2.23	2.30
254,788	258,228	261,714	265,247	268,828	272,457	276,135	279,863	283,641	287,470
253,068	256,401	259,780	263,205	266,676	270,194	273,760	277,374	281,038	284,751
0.68	0.71	0.74	0.77	0.80	0.83	0.86	0.89	0.92	0.95
1,720.49	1,826.57	1,933.83	2,042.29	2,151.97	2,262.89	2,375.05	2,488.48	2,603.20	2,719.23

Table B-12 Medium scenario results from year 2008 to 2035

	Unit	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15
Households with FWD	‘000	117,367	22,617	46,502	70,474	94,536	118,692	142,945	167,299
Residual waste generation	tonnes/year	1,532,810	1,553,503	1,574,475	1,595,730	1,617,273	1,639,106	1,661,234	1,683,661
Food waste generation	tonnes/year	563,360	570,966	578,674	586,486	594,403	602,428	610,561	618,803
Food waste diversion from landfill though the use of FWDs	tonnes/year	26,760	5,157	10,602	16,068	21,554	27,062	32,591	38,144
Cost of residual waste collection	£000	117,836	119,427	121,039	122,673	124,329	126,008	127,709	129,433
Cost of disposal of the residual waste collected	£000	82,312	83,423	84,549	85,691	86,848	88,020	89,208	90,413
Cost of disposal of the residual waste collected due to the use of FWDs	£000	80,875	83,146	83,980	84,828	85,690	86,567	87,458	88,364
Percentage reduction in cost of waste disposal	%	1.75	0.33	0.67	1.01	1.33	1.65	1.96	2.27
Total cost to LAs	£000	200,148	202,850	205,589	208,364	211,177	214,028	216,917	219,846
Total cost to LAs with the use of FWDs	£000	198,711	202,573	205,019	207,501	210,019	212,575	215,167	217,797
Percentage reduction in total cost	%	0.72	0.14	0.28	0.41	0.55	0.68	0.81	0.93
Savings	£000	1,436.99	276.91	569.35	862.85	1,157.46	1,453.21	1,750.16	2,048.34

(Continued)

2015/16	2016/17	2017/18	2018/19	2019/20	2020/21	2021/22	2022/23	2023/24	2024/25
191,757	216,324	241,003	265,797	290,710	315,747	340,910	366,205	391,633	417,200
1,706,390	1,729,426	1,752,773	1,776,436	1,800,418	1,824,723	1,849,357	1,874,324	1,899,627	1,925,272
627,157	635,624	644,205	652,901	661,716	670,649	679,702	688,878	698,178	707,604
43,721	49,322	54,949	60,602	66,282	71,990	77,728	83,495	89,292	95,122
131,180	132,951	134,746	136,565	138,409	140,277	142,171	144,090	146,036	148,007
91,633	92,870	94,124	95,395	96,682	97,988	99,310	100,651	102,010	103,387
89,285	90,222	91,173	92,140	93,123	94,122	95,137	96,168	97,215	98,279
2.56	2.85	3.13	3.41	3.68	3.95	4.20	4.45	4.70	4.94
222,813	225,821	228,870	231,960	235,091	238,265	241,482	244,742	248,046	251,394
220,466	223,173	225,919	228,706	231,532	234,399	237,308	240,258	243,251	246,286
1.05	1.17	1.29	1.40	1.51	1.62	1.73	1.83	1.93	2.03
2,347.80	2,648.58	2,950.74	3,254.31	3,559.34	3,865.88	4,173.97	4,483.66	4,795.00	5,108.04

(Continued)

2025/26	2026/27	2027/28	2028/29	2029/30	2030/31	2031/32	2032/33	2033/34	2034/35
442,910	468,766	494,772	520,932	547,251	573,732	600,380	627,199	442,910	468,766
1,951,263	1,977,605	2,004,303	2,031,361	2,058,784	2,086,578	2,114,747	2,143,296	1,951,263	1,977,605
717,156	726,838	736,650	746,595	756,674	766,889	777,242	787,735	717,156	726,838
100,983	106,879	112,808	118,773	124,773	130,811	136,887	143,001	100,983	106,879
150,005	152,030	154,083	156,163	158,271	160,408	162,573	164,768	150,005	152,030
104,783	106,197	107,631	109,084	110,557	112,049	113,562	115,095	104,783	106,197
99,360	100,458	101,573	102,706	103,856	105,025	106,211	107,416	99,360	100,458
5.18	5.40	5.63	5.85	6.06	6.27	6.47	6.67	5.18	5.40
254,788	258,228	261,714	265,247	268,828	272,457	276,135	279,863	254,788	258,228
249,365	252,488	255,656	258,869	262,127	265,432	268,784	272,184	249,365	252,488
2.13	2.22	2.31	2.40	2.49	2.58	2.66	2.74	2.13	2.22
5,422.81	5,739.38	6,057.79	6,378.09	6,700	7,025	7,351	7,679	5,422.81	5,739.38

Table B-13 High scenario results from year 2008 to 2035

	Unit	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13
Households with FWD	‘000	117,367	339,249	650,032	919,081	1,152,721	1,356,330
Residual waste generation	tonnes/year	1,532,810	1,553,503	1,574,475	1,595,730	1,617,273	1,639,106
Food waste generation	tonnes/year	563,360	570,966	578,674	586,486	594,403	602,428
Food waste diversion from landfill though the use of FWDs	tonnes/year	26,760	77,349	148,207	209,551	262,820	309,243
Cost of residual waste collection	£000	117,836	119,427	121,039	122,673	124,329	126,008
Cost of residual waste collection based on the biweekly scheme	£000	117,836	119,427	121,039	122,673	124,329	126,008
Percentage reduction in cost of waste collection		-	-	-	-	-	-
Cost of disposal of the residual waste collected	£000	82,312	83,423	84,549	85,691	86,848	88,020
Cost of disposal of the residual waste collected due to the use of FWDs	£000	80,875	79,269	76,591	74,438	72,734	71,414
Percentage reduction in cost of waste disposal	%	1.75	4.98	9.41	13.13	16.25	18.87
Total cost to LAs	£000	200,148	202,850	205,589	208,364	211,177	214,028
Total cost to LAs with the use of FWDs	£000	198,711	198,696	197,630	197,111	197,063	197,421
Percentage reduction in total cost	%	0.72	2.05	3.87	5.40	6.68	7.76
Savings	£000	1,437	4,154	7,959	11,253	14,113	16,606
Total cost to LAs with the use of FWDs based on the biweekly scheme	£000	198,711	198,696	197,630	197,111	197,063	197,421
Percentage reduction in total cost (incl. the biweekly scheme)	%	0.72	2.05	3.87	5.40	6.68	7.76
Savings	£000	1,437	4,154	7,959	11,253	14,113	16,606

(Continued)

2013/14	2014/15	2015/16	2016/17	2017/18	2018/19	2019/20	2020/21	2021/22	2022/23	2023/24
1,534,481	1,691,061	1,829,375	1,952,234	2,062,027	2,160,786	2,250,240	2,331,860	2,406,895	2,476,410	2,541,310
1,661,234	1,683,661	1,706,390	1,729,426	1,752,773	1,776,436	1,800,418	1,824,723	1,849,357	1,874,324	1,899,627
610,561	618,803	627,157	635,624	644,205	652,901	661,716	670,649	679,702	688,878	698,178
349,862	385,562	417,098	445,109	470,142	492,659	513,055	531,664	548,772	564,621	579,419
127,709	129,433	131,180	132,951	134,746	136,565	138,409	140,277	142,171	144,090	146,036
127,709	129,433	131,180	132,951	134,746	136,565	138,409	140,277	142,171	72,332	73,309
-	-	-	-	-	-	-	-	-	49.80	49.80
89,208	90,413	91,633	92,870	94,124	95,395	96,682	97,988	99,310	100,651	102,010
70,421	69,708	69,235	68,968	68,877	68,939	69,131	69,437	69,841	70,331	70,895
21.06	22.90	24.44	25.74	26.82	27.73	28.50	29.14	29.67	30.12	30.50
216,917	219,846	222,813	225,821	228,870	231,960	235,091	238,265	241,482	244,742	248,046
198,130	199,141	200,415	201,919	203,623	205,504	207,540	209,715	212,013	214,421	216,931
8.66	9.42	10.05	10.58	11.03	11.41	11.72	11.98	12.20	12.39	12.54
18,788	20,705	22,398	23,902	25,247	26,456	27,551	28,550	29,469	30,320	31,115
198,130	199,141	200,415	201,919	203,623	205,504	207,540	209,715	212,013	142,089	143,622
8.66	9.42	10.05	10.58	11.03	11.41	11.72	11.98	12.20	41.94	42.10
18,788	20,705	22,398	23,902	25,247	26,456	27,551	28,550	29,469	102,652	104,424

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2024/25	2025/26	2026/27	2027/28	2028/29	2029/30	2030/31	2031/32	2032/33	2033/34	2034/35
2,602,366	2,660,233	2,715,472	2,768,558	2,819,896	2,869,833	2,918,664	2,966,641	3,013,979	3,060,863	3,107,451
1,925,272	1,951,263	1,977,605	2,004,303	2,031,361	2,058,784	2,086,578	2,114,747	2,143,296	2,172,230	2,201,555
707,604	717,156	726,838	736,650	746,595	756,674	766,889	777,242	787,735	798,369	809,147
593,339	606,533	619,128	631,231	642,936	654,322	665,455	676,394	687,187	697,877	708,499
148,007	150,005	152,030	154,083	156,163	158,271	160,408	162,573	164,768	166,992	169,247
74,298	75,301	76,318	77,348	78,392	79,451	80,523	81,610	82,712	83,829	84,960
49.80	49.80	49.80	49.80	49.80	49.80	49.80	49.80	49.80	49.80	50
103,387	104,783	106,197	107,631	109,084	110,557	112,049	113,562	115,095	116,649	118,224
71,525	72,212	72,950	73,734	74,558	75,420	76,314	77,240	78,193	79,173	80,177
30.82	31.08	31.31	31.49	31.65	31.78	31.89	31.98	32.06	32.13	32.18
251,394	254,788	258,228	261,714	265,247	268,828	272,457	276,135	279,863	283,641	287,470
219,532	222,217	224,981	227,817	230,721	233,691	236,722	239,813	242,961	246,165	249,424
12.67	12.78	12.88	12.95	13.02	13.07	13.12	13.15	13.19	13.21	13.23
31,862	32,571	33,247	33,897	34,526	35,137	35,735	36,322	36,902	37,476	38,046
145,233	146,916	148,663	150,468	152,329	154,240	156,199	158,202	160,249	162,336	164,463
42.23	42.34	42.43	42.51	42.57	42.63	42.67	42.71	42.74	42.77	42.79
106,161	107,872	109,565	111,245	112,918	114,588	116,258	117,933	119,614	121,305	123,007

APPENDIX C

: Supplementary material for Chapter 6

C.1 Input data in GPS-X

Table C-1 Influent characterisation

Characteristic	Unit	Concentration
Total carbonaceous biochemical oxygen demand (BOD_5)	mgO_2/l	650
Total Chemical oxygen demand (COD)	mgCOD/l	1080
Total suspended solids (TSS)	mg/l	563
Volatile suspended solids (VSS)	mg/l	394
Total inorganic suspended solids	mg/l	169
Total TKN	mgN/l	46.5
Total nitrogen (TN)	mgN/l	46.5
Total phosphorous (TP)	mgP/l	11.3

BOD_5: is a measure of the amount of oxygen required to stabilize a waste biologically.

COD: is a measure of the amount of oxygen required for the chemical decomposition of organic and inorganic solids.

TSS: is the portion of total solids retained by a filter with a specified pore size.

VSS: is the portion of TSS than can be volatilized and burned off when ignited at 500± 50^{o}C

Inorganic suspended solids: is the residue that remains after the ignition of TSS

TKN: total kjeldahl nitrogen is the organic nitrogen present in wastewater which is converted to ammonia (NH_3) and then to ammonium (NH_4^+) through the action of heat and acid, such as TKN is equal to the sum of organic N, NH_3 and NH_4^+.

TN: is comprised by organic N, NH_3, NH_4^+, nitrite (NO_2^-) and nitrate (NO_2^-).

TP: is comprised by organic orthophosphates ($PO_4^{3-}, HPO_4^{2-}, H_2PO_4^-, H_3PO_4$), polyphosphates and organic phosphates.

Table C-2 Physical characteristics of the objects

Primary Sedimentation		
Surface:	m^2	75
Water depth:	m	3

UASB reactor		
Maximum volume	m^3	450
Headspace volume	m^3	22.5
Water height	m	10
Gas-Solid separator (GSS) height	m	2.3

Completely mixed activated sludge		
Maximum volume	m^3	350
Tank depth	m	3

Secondary sedimentation		
Surface	m^2	120
Water depth	m	3

Thickener		
Surface area	m^2	30
Depth	m	3

Anaerobic digester		
Maximum volume	m^3	65
Headspace volume	m^3	3.25

C.2 Conventional aerobic treatment – Mass flow and operational variables

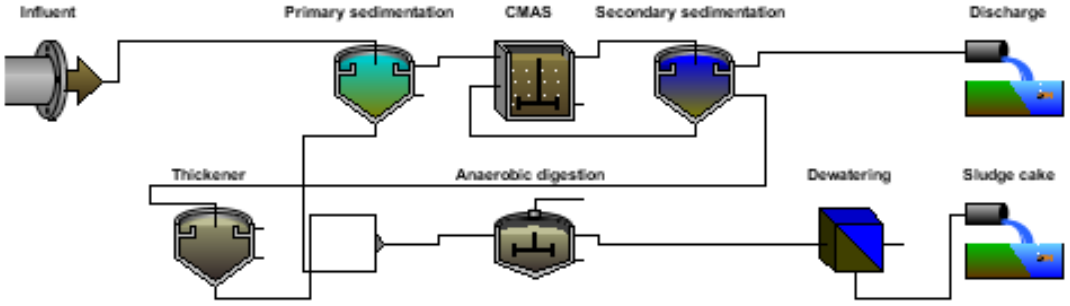


Table C-3 Primary treatment

Characteristic	Unit	In	Out as wastewater	Out as raw sludge
Flow	m^3/d	1335	1325	10
TSS	kg/d	751.45	270.60	480.85
BOD_5	kg/d	867.75	515.10	352.65
COD	kg/d	1446.25	858.50	587.75
TN	kg/d	62.08	46.18	15.90
TP	kg/d	15.09	12.50	2.58

Operational variables		In	Out
HRT	h	4.04	-
Surface overflow rate	$m^3/(m^2.d)$	17.67	-
Solids loading rate	$kg/(m^2.d)$	10.02	-
TSS removal efficiency	%	63.72	-
$cBOD_5$ removal efficiency	%	38.23	-
TN removal efficiency	%	25.05	-
TP removal efficiency	%	16.50	-
Water level	m	3.00	-
Sludge blanket height	m	2.24	-
Raw sludge flow	m3/d	-	10.00
Raw sludge solids	mg/L	-	48084.96
Raw sludge production	kg/d	-	480.85

Table C-4 Completely-mixed activated sludge (CMAS) treatment followed by secondary sedimentation

Characteristic	Unit	In	Out as wastewater	Out as waste activated sludge
Flow	m^3/d	1325	1285	40
TSS	kg/d	270.60	10.84	340.28
BOD_5	kg/d	515.10	38.01	210.86
COD	kg/d	858.50	63.35	351.43
TN	kg/d	46.18	20.29	22.77
TP	kg/d	12.50	6.38	6.12

Operational variables for the CMAS		In
F to M Ratio	$kgBOD_5/(kgMLVSS.d)$	0.52
Vol. organic loading	$kgBOD_5/(m^3.d)$	1.52
RAS recycle ratio	%	100.75

Operational variables for secondary sedimentation		In	Recycling	Out
HRT	h	3.25	-	-
Surface overflow rate	$m^3/(m^2.d)$	10.71	-	-
Solids loading rate	$kg/(m^2.d)$	97.57	-	-
Water level	m	3.00	-	-
Sludge blanket height	m	0.37	-	-
RAS flow	m^3/d	-	1335.00	-
RAS solids	mg/L	-	8506.96	-
WAS flow	m^3/d	-	-	40.00
WAS solids	mg/L	-	-	8506.96
WAS production	kg/d	-	-	340.28

Table C-5 Discharge of treated wastewater

Characteristic	Unit	Out
Flow	m^3/d	1285
TSS	kg/d	10.84
BOD_5	kg/d	38.01
COD	kg/d	63.35
TN	kg/d	20.29
TP	kg/d	6.38

Table C-6 Sludge (WAS) thickening

Characteristic	Unit	In	Out as liquor	Out as thickened sludge
Flow	m^3/d	40	34	6
TSS	kg/d	340.28	28.92	311.35
BOD_5	kg/d	210.86	18.67	192.19
COD	kg/d	351.43	31.11	320.31
TN	kg/d	22.77	2.40	20.37
TP	kg/d	6.12	0.67	5.45

Operational variables		In	Out
HRT	h	54.00	-
Solids loading rate	$kg/(m^2.d)$	11.34	-
Sludge blanket height	m	2.00	-
Solids capture - mass basis	%	91.50	-
Sludge mass flow	kg/d	-	311.35

Table C-7 Anaerobic digestion of sewage sludge

Characteristic	Unit	In	Out as digestate	Out as biogas
Flow	m^3/d	16	16	-
TSS	kg/d	792.20	473.47	-
BOD_5	kg/d	544.84	223.03	313.51
COD	kg/d	908.06	371.72	522.52
TN	kg/d	36.26	36.09	0.24
TP	kg/d	8.03	8.03	-

Operational variables		Sludge digestion	Biogas generation
HRT	d	4.06	-
VSS loading rate	$kgVSS/(m^3.d)$	8.38	-
VSS destruction	%	58.14	-
Gas production per VSS destroyed	$m^3/kgVSS$	1.05	-
Total gas flow rate	m^3/d	-	333.23
CH_4 content	%	-	61.85
CO_2 content	%	-	37.84

Table C-8 Sludge dewatering and production of a sludge cake

Dewatering unit	**Unit**	**In**	**Out as liquor**	**Out as sludge**
Flow	m^3/d	16	14.5	1.5
TSS	kg/d	473.47	21.45	452.01
BOD_5	kg/d	223.03	11.62	211.41
COD	kg/d	371.72	19.37	352.35
TN	kg/d	36.09	13.61	22.48
TP	kg/d	8.03	1.50	6.53

Operational variables		**Dewatering**	**Sludge cake**
Solids capture - mass basis	%	95.47	-
Cake mass flow	kg/d	-	452.01

C.3 Anaerobic-aerobic treatment – Mass flow and operational variables

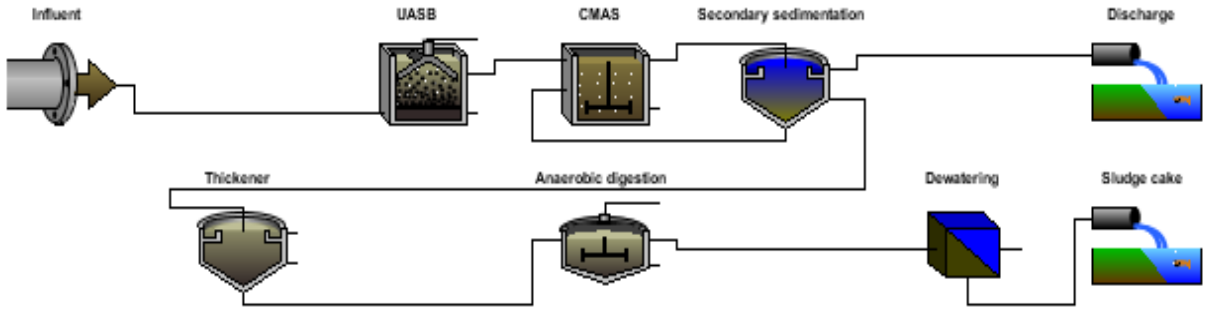


Table C-9 Anaerobic treatment at the UASB reactor

Characteristic	Unit	In	Out as wastewater	Out as biogas
Flow	m^3/d	1335	1335	10
TSS	kg/d	751.45	553.83	-
BOD_5	kg/d	867.75	447.65	420.10
COD	kg/d	1446.25	746.08	700.17
TN	kg/d	62.08	63.23	15.07
TP	kg/d	15.09	15.09	-

Operational variables		In	Out
HRT	h	8.09	-
SRT	d	37.90	-
COD loading rate	$kgCOD/(m^3.d)$	3.21	-
Superficial liquid velocity	m/h	1.2361111	-
Superficial gas velocity	m/h	0.3149317	-
Total gas flow rate	m3/d	-	340.12623
CH_4 content	%	-	77.105562
CO_2 content	%	-	18.211398

Table C-10 Completely-mixed activated sludge (CMAS) treatment followed by secondary sedimentation

Characteristic	Unit	In	Out as wastewater	Out as waste activated sludge
Flow	m^3/d	1335	1295	40
TSS	kg/d	553.83	12.46	469.97
BOD_5	kg/d	447.65	37.55	224.75
COD	kg/d	746.08	62.59	374.59
TN	kg/d	63.23	37.72	23.19
TP	kg/d	15.09	9.32	5.76

Operational variables for the CMAS		In
F to M Ratio	$kgBOD_5/(kgMLVSS.d)$	0.26
Vol. organic loading	$kgBOD_5/(m^3.d)$	0.76
RAS recycle ratio	%	100

Operational variables for secondary sedimentation		In	Recycling	Out
HRT	h	3.24	-	-
Surface overflow rate	$m^3/(m^2.d)$	10.79	-	-
Solids loading rate	$kg/(m^2.d)$	134.73	-	-
Water level	m	3.00	-	-
Sludge blanket height	m	0.66	-	-
RAS flow	m^3/d	-	1335.00	-
RAS solids	mg/L	-	11749.17	-
WAS flow	m^3/d	-	-	40.00
WAS solids	mg/L	-	-	11749.17
WAS production	kg/d	-	-	469.97

Table C-11 Discharge of treated wastewater

Characteristic	Unit	Out
Flow	m^3/d	1295
TSS	kg/d	12.46
BOD_5	kg/d	37.55
COD	kg/d	62.59
TN	kg/d	37.72
TP	kg/d	9.32

Table C-12 Sludge (WAS) thickening

Characteristic	Unit	In	Out as liquor	Out as thickened sludge
Flow	m^3/d	40	34	6
TSS	kg/d	469.97	39.95	430.02
BOD_5	kg/d	224.75	19.85	204.90
COD	kg/d	374.59	33.09	341.50
TN	kg/d	23.19	2.85	20.34
TP	kg/d	5.76	0.70	5.05

Operational variables		In	Out
HRT	h	54.00	-
Solids loading rate	$kg/(m^2.d)$	9.40	-
Sludge blanket height	m	2.00	-
Solids capture - mass basis	%	91.50	-
Sludge mass flow	kg/d	-	430.02

Table C-13 Anaerobic digestion of sewage sludge

Characteristic	Unit	In	Out as digestate	Out as biogas
Flow	m^3/d	6	6	-
TSS	kg/d	430.02	371.66	-
BOD_5	kg/d	204.90	155.42	48.49
COD	kg/d	341.50	259.04	80.82
TN	kg/d	20.34	20.19	0.20
TP	kg/d	5.05	5.05	-

Operational variables		Sludge digestion	Biogas generation
HRT	d	4.06	-
VSS loading rate	$kgVSS/(m^3.d)$	4.23	-
VSS destruction	%	26.04	-
Gas production per VSS destroyed	$m^3/kgVSS$	0.92	-
Total gas flow rate	m^3/d	-	50.79
CH_4 content	%	-	62.80
CO_2 content	%	-	36.78

Table C-14 Sludge dewatering and production of a sludge cake

Dewatering unit	**Unit**	**In**	**Out as liquor**	**Out as sludge**
Flow	m^3/d	6	4.5	1.5
TSS	kg/d	371.66	13.94	357.73
BOD_5	kg/d	155.42	5.95	149.42
COD	kg/d	259.04	9.91	249.04
TN	kg/d	20.19	4.35	15.84
TP	kg/d	5.05	0.59	4.46

Operational variables		**Dewatering**	**Sludge cake**
Solids capture - mass basis	%	96.25	-
Cake mass flow	kg/d	-	357.73

APPENDIX D

: Supplementary material for Chapter 8

D.1 Additional data for the sub-criteria evaluation of the use of FWDs in the Anglian region

Using the latest data provided by the Anglian Water (JR, 2010-11) and other UK figures adopted from the literature, the sub-criteria were evaluated based on the methodology described in Table 8-1 (*Chapter 8*).

Details on these data are presented in the following tables.

Table D-1 General calculations based on a tonne of food waste ground[6]

	Unit	Estimate
Water consumption for food waste grinding	l/kg	11.7
Water consumption per tonne food waste ground	l/tonneFW	11,700
Food waste moisture content	%	70
Moisture content per tonne food waste ground	l/tonneFW	700
Wastewater generation	l/tonneFW	12,400
BOD concentration in food waste	g/l	8.37
BOD generation	g/tonneFW	97,929
SS concentration in food waste	g/l	7.7
SS generation	g/tonneFW	90,090
DS concentration in food waste (per m^3 WW or 0.418kgSS, kg/d)	g/l	3.04
DS generation	g/tonneFW	37,657.62

Table D-2 RC calculations

	Unit	Estimate
FWDs power [7]	W	400

[6] Using the methodology developed in Chapter 5

[7] InSinkErator, Product Selector, available at: http://www.insinkerator.co.uk/products/selector.aspx

Times of a FWD use [8]	d	3
Duration of use [8]	sec/use	30
Energy use for a FWD per day	W/d	11
Energy use for grinding a tonne of food waste	kWh/d	22
Energy use for sewage collection		
Cost of power for sewage collection [9]	£/year	4,717,000
Industrial electricity price in the UK in 2010 [10]	p/kWh	7.84
Volume of sewage collected [9]	Ml/d	946.86
Energy consumption for sewage collected	kWh/m^3	0.17
Energy consumption for the collection of sewage generated by a tonne of food waste	kWh/m^3	2.16
Energy consumption for sewage treated		
Cost of power for sewage treatment [9]	£/year	19,230,000
Industrial electricity price in the UK in 2010 [10]	p/kWh	7.84
Tonnes of BOD treated [9]	tonneBOD/d	403.34
Energy consumption for sewage treated	kWh/tonne BOD	1,666.09
Energy consumption for the treatment of sewage generated by a tonne of food waste	kWh/tonneBOD	163.16
Energy consumption for sludge treatment		
Cost of power for sewage sludge treatment [9]	£/year	420,000
Industrial electricity price in the UK in 2010 [10]	p/kWh	7.84
Volume of sewage sludge treated [9]	tonneDS/year	195,700
Energy consumption for sewage sludge treated	kWh/tonneDS	27.37

[8] MTP, 2008, Based on the average waste diversion potential of 0.5kg per day

[9] Ofwat, 2012, Anglian Water Services: June Return 2010-11

[10] DECC, 2011, Quarterly energy prices

Energy consumption for the treatment of sewage sludge produced by the treatment of a tonne of food waste	kWh/tonneDS	1.03

Table D-3 GHG emissions calculations

	Unit	**Estimate**
Electricity from fossil fuels generates	kgCO_2/kWh	0.527
Carbon emissions from grinding a tonne of food waste	kgCO_2/t	11.594
Carbon emissions from the energy consumption for the collection of wastewater generated from food waste	kgCO_2/tonneFW	1.14
Carbon emissions from energy consumption for the treatment of wastewater generated from food waste	kgCO_2/tonne FW	85.99
Carbon emissions from the treatment of sewage generated by a tonne of food waste	kgCO_2/tonneFW	0.54

Table D-4 OC calculations

	Unit	**Estimate**
Cost of electricity for grinding [10]	p/kWh	0.13
Cost of grinding	£/tonneFW	2.86
Cost of water consumption [9]	£/Ml	242.32
Cost of water consumption for food waste grinding	£/tonneFW	2.84
Cost of wastewater treatment [9]	£/tonneBOD	595
Cost of wastewater treatment for food waste added	£/tonneFW	58.31
Cost of sludge treatment [9]	£/tonneDS	194
Cost of sludge treatment for food waste added	£/tonneFW	7.29

D.2 Additional data for the sub-criteria evaluation of the anaerobic co-digestion of food waste with sewage sludge in the Anglian region

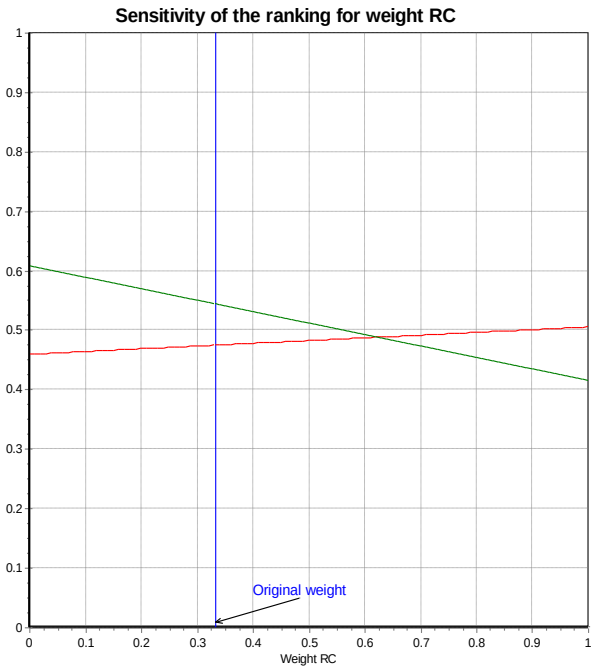
Table D-5 RC calculations

	Unit	Estimate
Fuel consumption (Diesel) for Refuse Collection Vehicles [11]	l/km	0.464
Fuel consumption (Diesel) for food waste collection from kerbside collection schemes in the UK [11]	l/tonne	11.6
Fuel consumption (Diesel) for food waste transport assuming a road trip to waste management facilities of 40km [11]	l/tonne	6.5
Energy consumption involved in the waste management facilities [12]	kWh/tonne	68.6

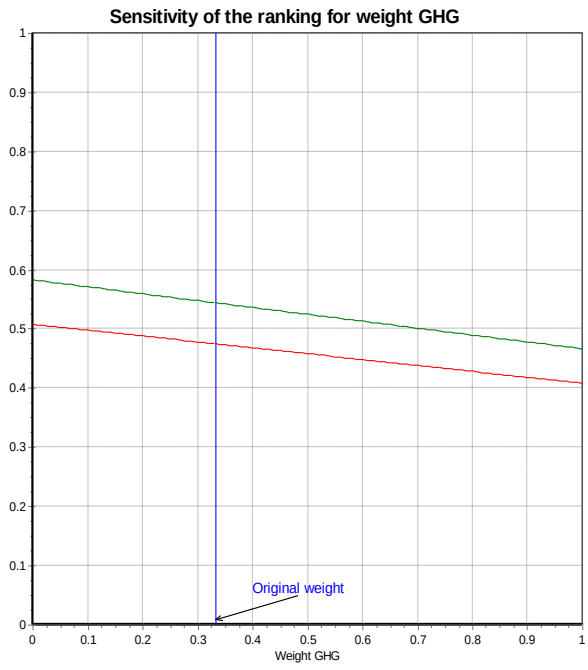
[11] Burnley et al., 2011, Energy implications of the thermal recovery of biodegradable municipal wastematerials in the United Kingdom. *Waste Management*, 31 (9–10), 1949-1959.

[12] Involves the operation of the plant using raw waste shredder, macerators, feed pumps, biogas compressor pumps, CHP and boiler water feed pumps, belt press, air filtration and minor ancillary equipment such as conveyors, and also the energy required to raise the temperature of the feedstock, pasteurise the digestate and maintain the temperature of the heated tanks as described by Banks et al. (2011) Available at: http://www.sciencedirect.com/science/article/pii/S0960852410013404

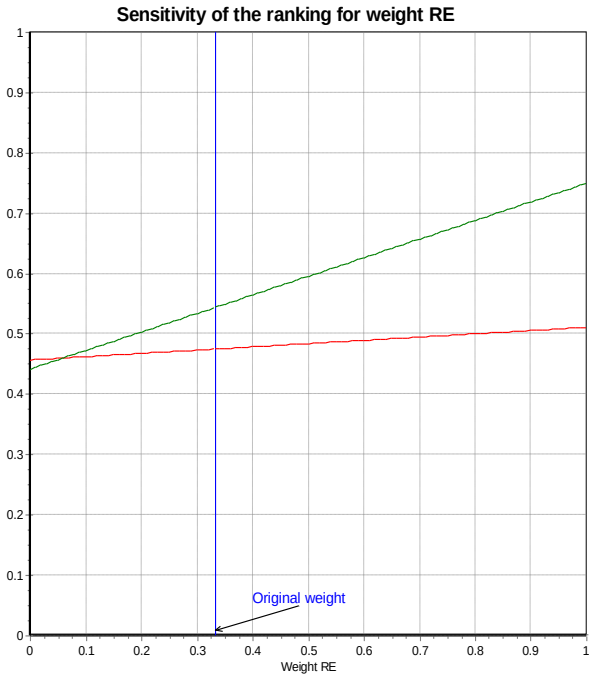
D.3 Results of sensitivity analysis



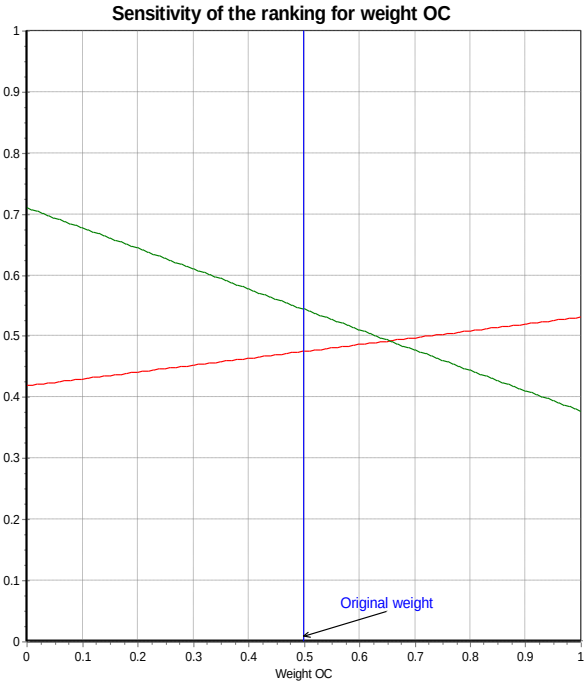
MCA 5: Weighted summation {maximum; Direct (Environmental: 0.333)}



MCA 5: Weighted summation {maximum; Direct (Environmental: 0.333)}

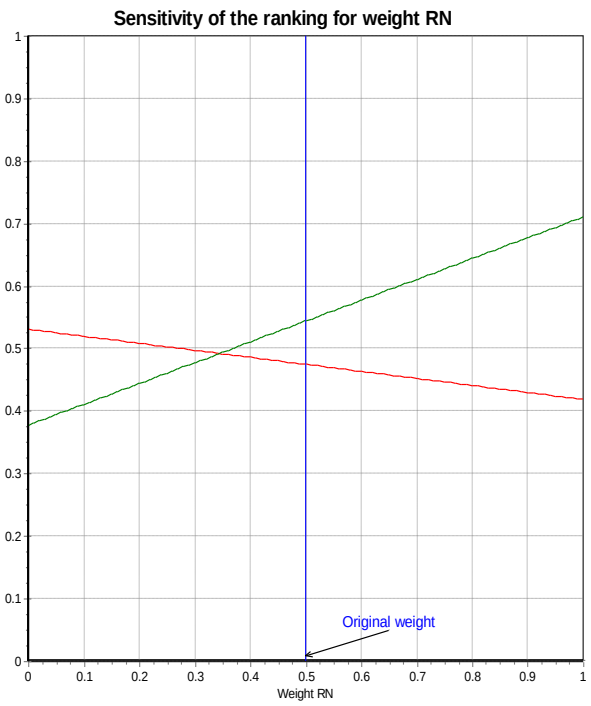


MCA 5: Weighted summation {maximum; Direct (Environmental: 0.333)}

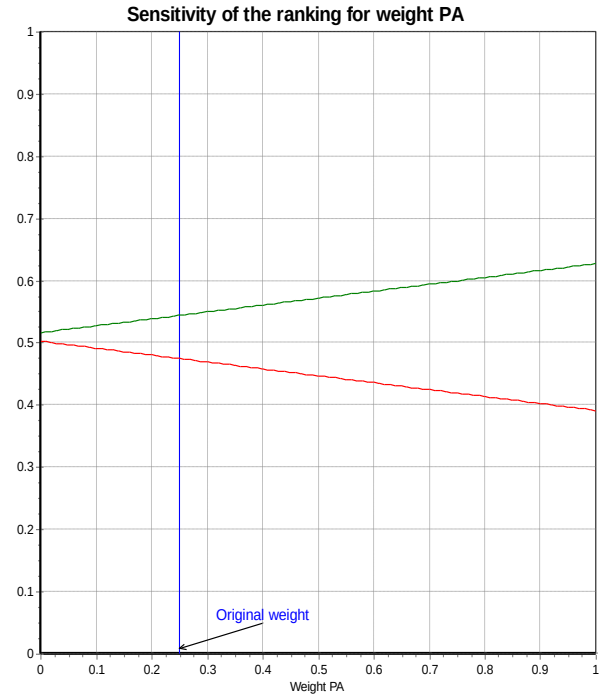


MCA 5: Weighted summation {maximum; Direct (Environmental: 0.333)}

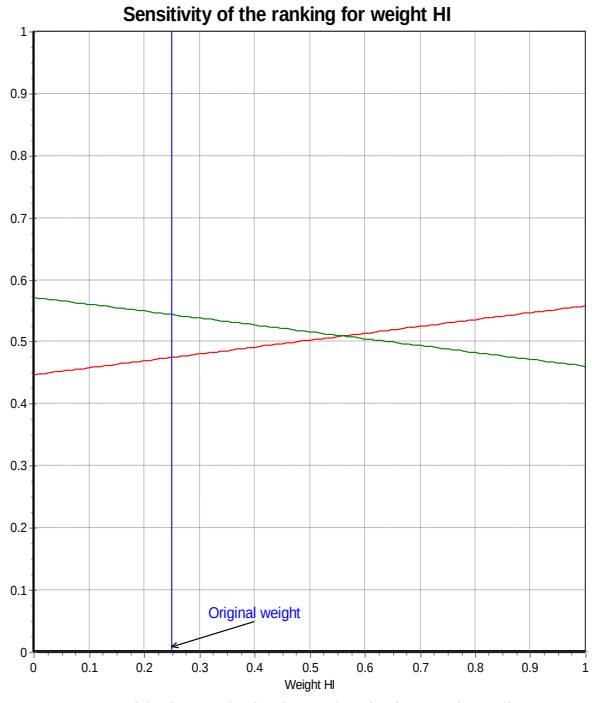
Green line: represents the anaerobic co-digestion of food waste with sewage sludge
Red line: represents the use of FWDs



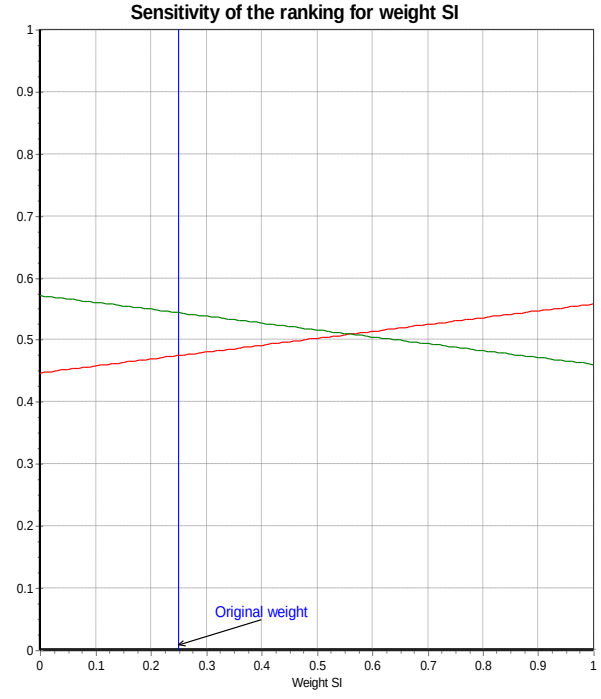
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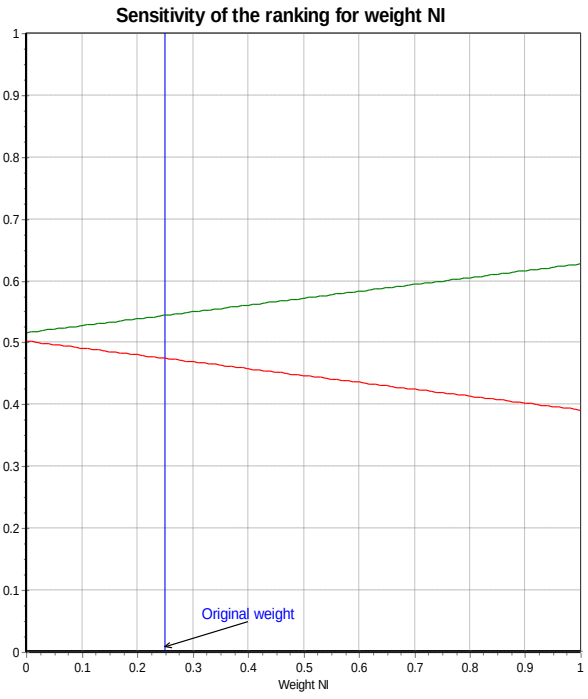


MCA 5: Weighted summation {maximum; Direct (Environmental: 0.333)}



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Green line: represents the anaerobic co-digestion of food waste with sewage sludge
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