Science of the Total Environment 693 (2019) 133516

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Environmental and economic implications of recovering resources from food waste in a circular economy

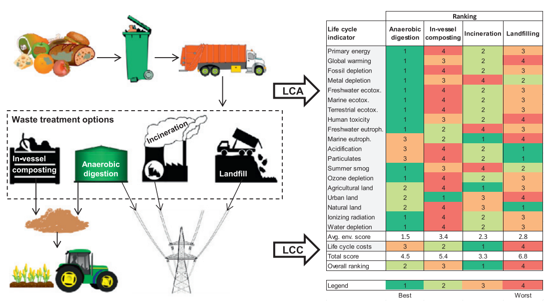
Peter C. Slorach, Harish K. Jeswani, Rosa Cuéllar-Franca, Adisa Azapagic *

Sustainable Industrial Systems, School of Chemical Engineering and Analytical Science, The University of Manchester, Manchester M13 9PL, UK

HIGHLIGHTS

- Anaerobic digestion, in-vessel composting, incineration and landfill are considered.
- Incineration is currently the most sustainable option per tonne of waste treated.
- Anaerobic digestion is the best option based on the annual volume of waste treated.
- Treating waste by anaerobic digestion could save annually £251 m and 490 kt CO_2 eq.
- Far greater environmental and cost savings would be gained through waste avoidance.

GRAPHICAL ABSTRACT



ARTICLE INFO

Article history:
Received 22 June 2019
Received in revised form 19 July 2019
Accepted 19 July 2019
Available online 20 July 2019

Editor: Damia Barcelo

Keywords:
Climate change
Costs
Environmental impacts
Food
Sustainability assessment
Waste treatment

ABSTRACT

Around a third of food is wasted globally, requiring significant resources for its treatment and disposal, in addition to wasting valuable resources. Following the circular economy principles, this waste should ideally be avoided, and if not possible, treated to recover resources. This paper considers the life cycle environmental and economic implications of recovering energy and material resources from food waste, focusing on the UK situation. Four treatment methods are considered: anaerobic digestion, in-vessel composting, incineration and landfilling. The results show that per tonne of waste treated, anaerobic digestion has the lowest environmental impacts in 13 out of the 19 categories considered in the study, including net-negative global warming potential. In-vessel composting is the least sustainable option environmentally, in contrast to being preferred over incineration according to the circular economy principles. Incineration has the lowest life cycle costs (£71/t), while landfilling is the costliest option (£123/t). Managing the 4.9 Mt of food waste collected annually from UK households via the four methods generates 340,000 t CO_2 eq. and costs £452 m, in addition to causing a number of other environmental impacts. However, it also saves 1.9 PJ of primary energy, primarily due to electricity generation through incineration. If all of this food waste was incinerated, £103 m and 360,000 t CO_2 eq./year could be saved compared to current waste management, rendering incineration a carbon-negative technology. This would also result in savings in 14 other impacts, but would increase summer smog by 30% and metal depletion by 56%. The environmental benefits of incineration would be exceeded only if all food waste was treated by anaerobic digestion, which would save 490,000 t CO_2 eq./year and produce 50% more electricity per tonne of waste than incineration. Anaerobic digestion would also lead to savings in 14 other impacts compared to the present situation, but would result in a four times higher acidification and three times greater emissions of particulate matter. In addition, it would save £251 m/year compared to the current costs. Nevertheless, prevention of avoidable food waste would realise far greater environmental and economic savings, estimated here at 14 Mt CO_2 eq. and £10.7 bn annually.

* Corresponding author.
E-mail address: adisa.azapagic@manchester.ac.uk (A. Azapagic).

https://doi.org/10.1016/j.scitotenv.2019.07.322

1. Introduction

With approximately a third of food intended for human consumption wasted through the food supply chains, the loss of embedded resources is receiving increasing global attention (Gustavsson et al., 2011). The total production value of global food waste (FW) is estimated at $1 trillion, increasing to $2.6 trillion when social and economic costs are considered (FAO, 2014) and the greenhouse gas (GHG) emissions associated with the FW supply chains account for 6.8% of global emissions (FAO, 2015; The World Bank, 2016). In their Sustainable Development Goals (SDG 12.3), the United Nations have stated a target to half the per-capita food waste at the retail and consumer levels by 2030 (UN, 2015). The issue of food waste is also becoming pertinent in the EU with the growing pressures to implement a circular economy (EC, 2015). In the UK, the food industry has already agreed to reduce food and drink waste across the whole supply chain, but the target set through the voluntary Courtauld Commitment 2025 (WRAP, 2017b) is not as ambitious, aiming only for a 20% reduction between 2015 and 2025. Food waste reduction fits into the UK's wider environmental commitment to the EU's nationally determined contribution (NDC) of a 40% reduction in GHG emissions by 2030 (EC, 2014) and to its self-set target for net zero emissions by 2050 (UK Government, 2019).

Within the EU, the majority of FW is generated by households, with an estimated 47 Mt disposed of each year (Stenmarck et al., 2016). In the UK, household FW reached a total of 7.3 Mt in 2015 (Quested and Parry, 2017). Following a campaign to reduce household FW, a 16% reduction had been achieved between 2007 and 2012 (Quested et al., 2013b), but the trend has since been reversed, with a 4% increase in FW from 2012 to 2015. At present, more than half of the UK household FW is landfilled or incinerated, with the rest treated by anaerobic digestion (AD), composted or used as animal feed. Although these methods help to deal with existing waste while at the same time recovering energy and resources, they also generate additional environmental impacts and economic costs.

The circular economy principles (EMF, 2015) as well as the waste management hierarchy (DEFRA, 2018a) favour AD and composting over incineration and landfill. However, their environmental and economic sustainability will vary depending on many factors, including the composition and volume of waste as well as the geographical region where it is generated and treated. This is reflected in previous studies which considered the sustainability of treating FW in different countries, whose findings on the preference of different treatment methods differed depending on the location. For example, Oldfield et al. (2016) undertook a national life cycle assessment (LCA) study for Ireland, assessing the global warming, eutrophication and acidification potentials of FW treatment via AD, in-vessel composting (IVC), incineration and landfilling, in comparison with waste minimisation. They found that AD had the lowest impacts of the three treatment options for all three environmental indicators, but the difference between AD, IVC and incineration was small when compared to waste minimisation, which was the best option. The benefits of waste minimisation were also discussed by Beretta and Hellweg (2019), who showed that a 38% reduction in FW in Switzerland would lead to a 41% lower climate change impact.

Bernstad and la Cour Jansen (2011) also considered the AD, IVC and incineration but for a Swedish town, focusing on five impacts: global warming, acidification, ozone depletion, photochemical oxidants formation (summer smog) and nutrient enrichment. Their results suggested that AD resulted in the greatest reduction in global warming and photochemical oxidants formation, but both AD and IVC increased nutrient enrichment and acidification in comparison to incineration. An LCA study based in Seoul showed that producing wet animal feed from household FW by sterilising and mixing with corn starch, had the greatest savings in global warming potential compared to landfilling and composting (Kim and Kim, 2010). For a review of studies in other countries, see Bernstad and la Cour Jansen (2012).

Some studies of the life cycle costs (LCC) of FW management have also been carried out to evaluate their economic sustainability. For instance, in the follow up to their above-mentioned LCA study (Kim and Kim, 2010), Kim et al. (2011) assessed the LCC of the same treatment routes in Seoul and found that wet feed also had the lowest life cycle cost, when compared to dry feed, landfilling, composting and AD. Two other LCC studies on consumer FW management (Carlsson Reich, 2005; Martinez-Sanchez et al., 2015) focused primarily on methodology and included only brief case studies for Danish and Swedish systems. In a wider study, Martinez-Sanchez et al. (2016) estimated LCC of all FW generated in Denmark and found incineration had marginally lower costs than either co-digestion or conversion to animal feed via the source separation of plant-derived food waste, when both direct and indirect costs were considered. However, the study focused primarily on the large savings possible if avoidable FW was prevented.

As far as the authors are aware, there are no LCA and LCC studies of FW management for the UK. Apart from one LCA of household FW management in a London borough (Evangelisti et al., 2014) and another on incineration and landfilling of municipal solid waste (MSW) at the UK level (Jeswani and Azapagic, 2016), no other studies exist. Therefore, this paper presents for the first time the life cycle environmental impacts and costs of the four household FW management routes currently used in the UK: AD, IVC, incineration and landfilling. The aim is to identify environmentally and economically the most sustainable options, placing them in the context of a circular economy. Following the concept of the latter, the treatment methods are also compared to the avoidance of food waste. The outcomes of this study will be of interest to local authorities, policy makers and consumers.

The next section details the LCA and LCC methodologies applied to evaluate the environmental and economic sustainability of FW treatment, along with the assumptions and data used in the study. The results are presented and discussed in Section 3 and the conclusions and recommendations in Section 4.

2. Methods

The LCA study has been completed in accordance with the ISO 14040/14044 standards (ISO, 2006a; ISO, 2006b). The LCC methodology is based on guidelines published by Swarr et al. (2011) and Hunkeler et al. (2008), which is consistent with and complementary to the LCA methodology. The following sections detail the goal and scope of the study, methods used for the estimation of environmental impacts and costs, and the inventory data.

2.1. Goal and scope

The goal of the study is to estimate the life cycle environmental impacts and economic costs of the current household FW treatment system in the UK. This comprises AD, IVC, incineration with energy recovery and landfill with utilisation of landfill gas (LFG). As all the methods recover either energy and/or material resources, the aim is to identify the most sustainable option within the context of a circular economy. A further goal is to compare the sustainability implications of food treatment with waste avoidance.

When comparing the treatment methods, the term “food waste” refers to any edible or inedible fractions of food discarded by households – this is further defined as avoidable, possibly avoidable and unavoidable FW (Quested and Parry, 2017). When prevention is considered, only the avoidable and possibly avoidable fractions are considered as preventable.

The scope of the study, shown in Fig. 1, encompasses FW collection from households, its treatment by one of the above-mentioned methods and recovery of energy and/or materials, including the related environmental and revenue credits. Construction of the treatment plants has also been considered but their decommissioning and the resulting waste are excluded. The environmental impacts of FW are also excluded, i.e. the FW is considered burden-free.

The functional unit is defined as the "treatment of 1 tonne of household food waste with recovery of resources". These results are subsequently scaled up to the annual amount of household FW currently collected in the UK.

2.2. Estimation of life cycle impacts and costs

The LCA impacts have been estimated following the Hierarchist ReCiPe 1.08 impact assessment method (Goedkoop et al., 2013). In total, 18 impacts are included in this method, all of which are considered. In addition, primary energy demand has also been estimated, using the method in GaBi V8 (Thinkstep, 2017). The latter has also been used for LCA modelling and estimation of the impacts. An uncertainty analysis has been carried out using the Monte Carlo method with 10,000 simulations for each treatment method, assuming a uniform distribution between the minimum and maximum values of the inventory data.

The LCC have been estimated according to Eq. (1) and they comprise costs to local authorities, operators of treatment facilities and consumers:

$$LCC = C_{LA} + C_O + C_C \qquad (£/t\ FW) \tag{1}$$

where:

LCC total life cycle costs of waste treatment option (£/t FW)
C_{LA} costs to the local authority (£/t FW)
C_O cost to the operator of treatment facility (£/t FW)
C_C costs to the consumer (£/t FW).

Costs to the local authority consist of FW collection costs and gate fees:

$$C_{LA} = C_{WC} + C_{GF} \qquad (£/t\ FW) \tag{2}$$

where:

C_{WC} costs of waste collection (£/t FW)
C_{GF} gate fees (£/t FW).

Costs to the operator involve construction and operating costs associated with the waste treatment facility, less any revenue:

$$C_O = C_{CC} + C_{OC} - R_O \qquad (£/t\ FW) \tag{3}$$

where:

C_{CC} capital (overnight) costs of waste treatment facility (£/t FW)
C_{OC} operating costs of waste treatment facility (£/t FW)
R_O operator's revenue from gate fees, government incentives and sales of recovered energy or materials (£/t FW).

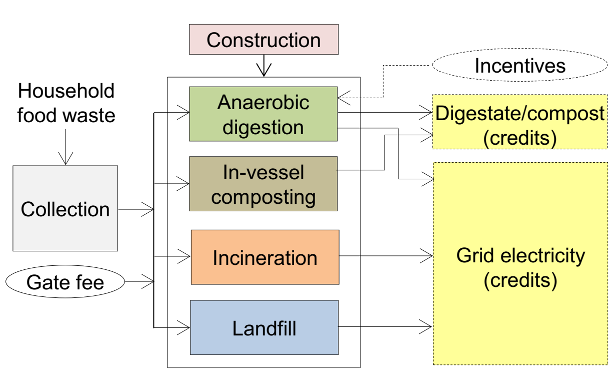


Fig. 1. Life cycle stages considered in study. [The dashed lines indicate system credits.]

The operating costs comprise consumables, utilities, insurance, and maintenance:

$$C_{OC} = C_{Con} + C_U + C_I + C_M \qquad (£/t\ FW) \tag{4}$$

where:

C_{Con} cost of operating consumables (£/t FW)
C_U cost of utilities (water and electricity) (£/t FW)
C_I cost of insurance (£/t FW)
C_M maintenance costs of the waste treatment facility (£/t FW).

Typically, no economic value is assigned to waste streams and therefore the value of the cost to the consumer (C_C) is zero; however, the shelf value of the avoidable fraction of food waste is considered in the results section to provide context. Labour costs are not considered, in congruence with the LCA methodology which does not consider the impacts associated with labour.

2.3. Inventory data and assumptions

The primary inventory data have been obtained from operators of the treatment plants and supplemented from literature. Background life cycle inventory data have been sourced from the Ecoinvent database V3.3 (Ecoinvent, 2016). Further details on the data are provided below.

2.3.1. UK household food waste management

The amounts of kerbside-collected FW treated by different disposal methods have been determined based on local authority data collated in the database WasteDataFlow (2018) and annual reports produced by the national governments of the UK; the results are summarised in Fig. 2. Food waste disposed to the sewer, home-composted or fed to livestock is outside the scope of this study. For a full breakdown of all household waste, including FW, see Table S1 in the Supplementary information (SI).

As can be seen in Fig. 2, the majority of FW is embedded in MSW, of which 1.65 Mt is landfilled and 2.44 Mt incinerated annually. Approximately 29% of UK households have a separate food waste collection and 20% have food waste collections co-mingled with green waste (WRAP, 2016a). In total, 0.51 Mt of FW are collected separately and 0.3 Mt are co-mingled with green waste. There are no definitive figures for the final treatment of these waste streams but they are split between AD and IVC. For the separately-collected FW, it is assumed that 0.31 Mt is sent to AD in accordance with estimates by Burns et al. (2017). The remaining 0.2 Mt of separately collected FW and all FW co-mingled within green waste (0.3 Mt) are assumed to be treated at an IVC facility, as woody plant matter is not suited to AD. Thus, the total amount of FW collected by local authorities, either separately or co-mingled with other waste is estimated at 4.9 Mt/year.

2.3.2. Food waste composition and value

The composition of household FW is not homogenous and will vary across different locations and at different times of the year. A study in Wales (Esteves and Devlin, 2010) took summer and winter samples from 22 local authorities and analysed their elemental composition. The resulting average data are presented in Table S2 in the SI and are assumed to be representative of UK household FW. All carbon is assumed to be of biogenic origin. This composition has been used for incineration and landfill. However, for AD and IVC, it has not been possible to model the complex biological process for a specific composition and, therefore, average composition values for food-derived biogas, digestate and compost have been used instead, as detailed further below.

To estimate the implications of waste avoidance, two FW fractions are considered: avoidable (suitable for human consumption) and potentially avoidable (e.g. potato peel and bread crusts) waste. Their volumes have been estimated at 2.9 Mt/year and 0.8 Mt/year (Quested and Parry, 2017), respectively. Only GHG emissions data are available

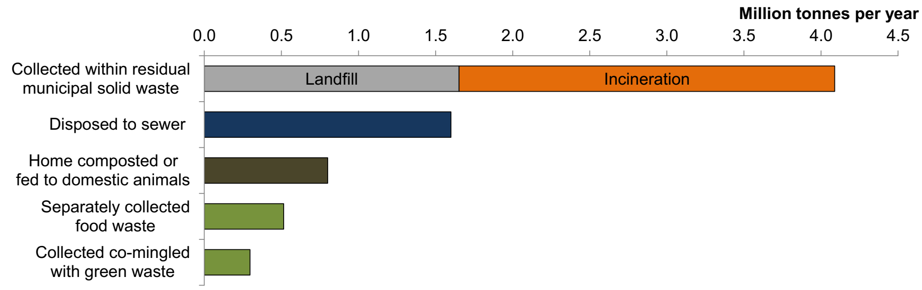


Fig. 2. Disposal routes for UK household food waste in 2016 (Quested and Parry, 2017; DEFRA, 2017a; SEPA, 2017; StatsWales, 2017; DAERA-NI, 2017; WasteDataFlow, 2018).

for FW (Quested et al., 2013a) and hence it has only been possible to estimate the global warming potential of avoiding these two FW fractions. Their shelf value (cost to consumer) has been determined using average cost data from UK family food datasets (DEFRA, 2017b) and scaled up to the total household FW of 4.9 Mt/year (Quested and Parry, 2017).

2.3.3. Waste collection

The FW is collected by a conventional municipal waste lorry (21 t), which has been modelled using Ecoinvent (2016) for an average distance of 20 km. Other distances (5–30 km) are considered as part of the uncertainty analysis.

The collection costs have been estimated using a local authority kerbside costing tool (WRAP, 2018). A sample of different local authorities has been taken to represent a national average, with different costs considered for urban, mixed urban/rural and rural populations, as well as for different levels of economic deprivation. It has been assumed that there are fortnightly collections of residual waste and a weekly collection of separated FW. The average costs for FW and MSW collections have been weighted based on the mix of household types in the UK (Table S3).

2.3.4. Waste treatment

The model for each treatment method is representative of the mix of facilities operating in the UK as discussed next. In the base case, the average values provided below have been used across the treatment options, with the minimum and maximum values considered in the uncertainty analysis.

2.3.4.1. Anaerobic digestion. The life cycle of the AD system (Fig. 3a) comprises waste collection, construction and operation of AD and CHP plants to generate respectively biogas and electricity, and application of digestate to agricultural land. The excess electricity exported to grid has been assumed to displace UK grid electricity, for which the system has been credited. System credits have also been applied for the digestate which replaces an equivalent amount of mineral fertilisers. The heat is used to maintain the reactor temperature and to destroy pathogens in the digestate (UK Government, 2014). As there is no market for excess heat from AD plants (Scholes and Areikin, 2014), no credits have been applied for the heat.

The inventory data, summarised in Table 1, have been obtained from a number of operating and pilot scale plants within the EU. The average data have been used in the base case, with the minimum and maximum values considered in the uncertainty analysis. Food waste is a component of the feedstock at all plants but not the exclusive feed. The majority of AD plants in the UK have single-stage mesophilic (30–40 °C) reactors (Scholes and Areikin, 2014). The representative plant used in this study has a capacity of 25,000 t/year over a 25 year lifespan and operates using a high-solids 2500 m^3 single-stage mesophilic digester. The biogas is assumed to be 60% methane and 40% carbon dioxide (Ecoinvent, 2016). It is utilised in a 1 MW CHP reciprocating internal combustion engine to supply all on-site demand, with the excess electricity exported to the grid. The CHP plant produces 277 kWh electricity and 476 kWh heat per tonne of FW.

Methane losses are assumed to be equivalent to 1%–3% of the total biogas produced (Bernstad and la Cour Jansen, 2011; Evangelisti et al., 2014; Naroznova et al., 2016). Both ammonia and nitrous oxide are generated during the AD, which are also considered (Ecoinvent, 2016).

The composition of the digestate (Table S4) is based on the analysis of digestate from multiple UK AD plants processing FW (Rigby and Smith, 2011). As can be seen, the nutrient content of the digestate ranges widely, which has been explored in the uncertainty analysis. The quantities of fertilisers displaced by the digestate have been estimated based on the mass of nitrogen, phosphorus, and potassium. Phosphorus and potassium have an equivalent uptake efficiency compared to mineral fertiliser but the uptake of nitrogen ranges from 30% to 146% (Bernstad and la Cour Jansen, 2011). In this work, the nitrogen uptake and emissions are based on UK field results for the application of food-derived digestate, with the results shown in Table 1 (Nicholson et al., 2016). The ammonia and nitrate emissions that would occur following the application of ammonium nitrate have been included in the credits applied for displacing the mineral fertiliser.

Nicholson et al. (2016) found that the average nitrous oxide emissions from digestate application ranged from 0.2% to 0.9% of the nitrogen applied. The Intergovernmental Panel on Climate Change (IPCC) assume a default value of 1% for all nitrogen fertilisers (De Klein et al., 2006), while Harty et al. (2016) have found nitrous oxide emissions from application of calcium ammonium nitrate to range from 0.6% to 3.8%. While these results show that it is likely that nitrous oxide emissions from digestate are lower than those of ammonium nitrate, due to the uncertainty, they have been modelled as equal and no credit has been applied.

An overview of the LCC data for AD is provided in Table 2. The capital costs of the AD plant have been estimated based on BEIS (2017b). General AD plant maintenance has been assumed to cost 2% of the capital cost and maintenance of the CHP unit is estimated at 1 p/kWh_e (The Andersons Centre, 2010). AD plants are eligible to receive incentives for the heat and electricity they produce. A feed-in tariff is paid for all electricity generated, including the amount used on site. An additional export tariff is paid for the electricity supplied to the grid (Ofgem, 2018). As mentioned earlier, no heat is exported and hence no revenue is considered.

Based on the available data, the median gate fee paid by local authorities to dispose of waste at AD plants is £29 (WRAP, 2017a). The demand for digestate is low and, on average, AD operators pay a small fee for it to be removed.

2.3.4.2. In-vessel composting. Fig. 3b provides an overview of the life cycle of the IVC system considered in this study. Following collection, the FW enters the air-controlled facility via an airlock and is stored before being shredded and fed to the process vessels (Eades et al., 2011). The composting takes place under aerobic conditions, in horizontal rotating steel drums, with a retention time of four days. The output is screened to

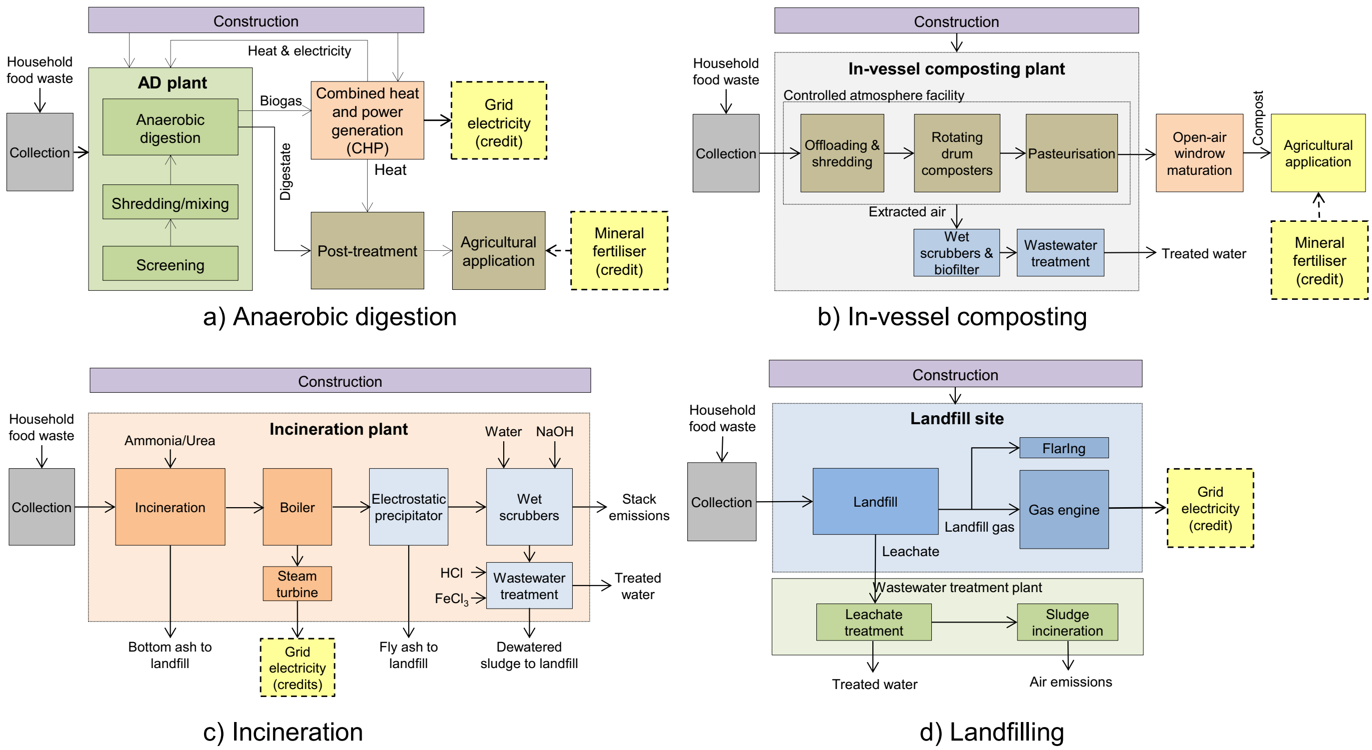


Fig. 3. Life cycle stages considered for the four food waste treatment methods. [The dashed lines indicate system credits.]

ensure particle size is compliant with animal-by-products regulations (UK Government, 2014), with oversized materials returned to the shredder. The compliant material is pasteurised at 70 °C for an hour using steam to kill the pathogens; the steam is supplied by a gas boiler. The treated compost is then matured in open windrows over 10–14 weeks (WRAP, 2016b), ready for use in agriculture. Like AD, the

Table 1
Inventory data for the treatment of household food waste via anaerobic digestion.
(Adapted from Slorach et al. (2019).)

Parameter	Unit	Average	Min.	Max.	Reference
Biogas production	Nm^3/t FW	137	85	187	Bernstad and la Cour Jansen (2011); Banks et al. (2011); Monson et al. (2007)
Electricity generated	kWh/t FW	277	172	378	Ecoinvent (2016)
Electricity exported	kWh/t FW	254	128	368	~II~
Heat generated	kWh/t FW	476	295	649	~II~
Parasitic electricity use by AD plant	kWh/t FW	23	10	44	Bernstad and la Cour Jansen (2011); Banks et al. (2011); Monson et al. (2007)
Parasitic heat use by AD plant	kWh/t FW	82	36	113	~II~
Mineral oil	kg/t FW	0.093			Ecoinvent (2016)
Emissions					
Methane (AD facility)	% of produced CH_4	2	1	3	Bernstad and la Cour Jansen (2011); Evangelisti et al. (2014); Naroznova et al. (2016)
Ammonia (AD facility)	kg/t FW	0.24			Ecoinvent (2016)
Nitrous oxide (AD facility)	kg/t FW	0.015			~II~
Hydrogen sulphide (AD facility)	kg/t FW	0.0082			~II~
Methane (CHP unit)	kg/t FW	0.072			~II~
Carbon monoxide (CHP unit)	kg/t FW	0.15			~II~
Sulphur dioxide (CHP unit)	kg/t FW	0.078			~II~
Nitrogen oxides (CHP unit)	kg/t FW	0.047			~II~
NMVOC[a] (CHP unit)	kg/t FW	0.0062			~II~
Platinum (CHP unit)	mg/t FW	0.022			~II~
Nitrogen fertiliser displaced	Equiv. % of the mass of N in digestate	40	18	65	Derived from Nicholson et al. (2016)
Potassium fertiliser displaced	Equiv. % of the mass of K in digestate	100			Bernstad and la Cour Jansen (2011); Møller et al. (2009)
Phosphorus fertiliser displaced	Equiv. % of the mass of P in digestate	100			~II~
Ammonia emission to air from					
Digestate application	% total N emitted as NH_3-N	42	18	65	Nicholson et al. (2016)
Ammonium nitrate application	% total N emitted as NH_3-N	2	0	13	DEFRA (2006a)
Nitrate leaching to soil from					
Digestate application	% total N emitted as Nitrate-N	15	10	20	Nicholson et al. (2016)
Ammonium nitrate application	% total N emitted as Nitrate-N	7.5	5	10	Assumption based on DEFRA (2006b)
Nitrous oxide emission air from					
Digestate application	% total N emitted as N_2O-N	0.45	0.20	0.90	Nicholson et al. (2016)

[a] Non-methane volatile organic compounds.

Table 2
Life cycle costs of the treatment of household food waste via anaerobic digestion.

Anaerobic digestion costs	Unit	Amount	Reference
Capital (overnight) cost (1 MW facility)	£ m	4.628	BEIS (2017b)
Operating costs			
General AD plant maintenance	% of capital	2	The Andersons Centre (2010)
CHP maintenance	£/kWh$_e$	0.01	The Andersons Centre (2010)
Insurance	% of capital	1	The Andersons Centre (2010)
Digestate removal costs	£/t digestate	3.73	Scholes and Areikin (2014)
Disposal costs			
Gate fees	£/t FW	29	WRAP (2017a)
Revenue			
Feed-in tariff (0.5–5 MW facilities)	£/kWh$_e$	0.016	Ofgem (2018)
Export tariff	£/kWh$_e$	0.050	Ofgem (2018)

IVC system has been credited for displacing an equivalent amount of mineral fertilisers. The latter is shown in Table 3, alongside other inventory data. The average values have been used in the base case, and the minimum and maximum values in the uncertainty analysis.

The data for composting facility are based on an IVC facility in Leominster, UK (Eades et al., 2011). However, as this facility has a relatively high electricity consumption per tonne of waste, an average electricity consumption from several plants has been used instead. The capacity has been assumed at 50,000 t of FW per year, based on the capacity of UK plants (Letsrecycle, 2010, 2015, 2018, 2009, 2006). The assumed lifespan of the plant is 25 years.

Emissions data for the IVC facility, windrow maturation and compost application have been sourced from literature (Table 3). The air from the controlled facility is treated first in a sulphuric-acid scrubber to control emissions of ammonia and then in sodium-hydroxide scrubber to remove hydrogen sulphide. Finally, the gases are passed through a biofilter, which contains bark, woodchip and shells, to oxidise odorous compounds. A small quantity of nitrous oxide is emitted to the air from the IVC biofilter. During the maturation process, ammonia, methane and non-methane volatile organic compounds are released to the air (Table 3). Emissions of biogenic carbon dioxide from the breakdown of organic matter have not been considered, in congruence with common LCA practice.

In the absence of specific data for FW compost, the composition of the matured compost has been assumed equal to a UK average for mixed green waste and food compost (Bhogal et al., 2016); see Table S5 in the SI for details. The software tool MANNER-NPK (ADAS, 2013) has been used to determine the amount of mineral fertilisers displaced and the emissions released from the use of food-derived compost. The emissions that would otherwise have occurred from mineral fertilisers have been accounted for as discussed previously for the AD system. A range of displacement and emissions values have been determined for nitrogen fertiliser, dependent on the soil type (clay or sandy) and whether the compost is incorporated into the soil (Table 3). Analogous to digestate application, it has been assumed that there are no equivalent variations for phosphorus and potassium. The gaseous emissions of nitrous oxide following the application of compost do not significantly differ from the application of ammonium nitrate (Nicholson et al., 2017); therefore, neither a credit nor an impact has been considered.

The data for LCC can be found in Table 4. Accounting for inflation (Bank of England, 2018), the capital cost per tonne of annual capacity has decreased from around £150 in the late 2000s to about £90 in 2018 (Letsrecycle, 2018, 2015, 2010, 2009, 2006; MRWA, 2010). Consequently, a value of £100 per tonne of annual capacity has been assumed, which equates to an overall capital cost of £5 m for the capacity of 50,000 t/year in Table 4. Maintenance costs have been assumed to be 3% of the capital costs, in accordance with common engineering practice (Towler and Sinnott, 2013). Electricity is the major operating expense and it has been costed based on prices for a medium business (BEIS, 2017c). The cost of steam for pasteurisation, supplied by a gas boiler, is based on the UK average prices in 2016/2017 for non-household use with annual consumption under 1000 GJ (Eurostat, 2018).

The median gate fee paid by local authorities to dispose of waste at IVC plants is £46 (WRAP, 2017a). Compost is sold to agricultural users at a mean price of £0.75/t (Horne et al., 2013).

2.3.4.3. Incineration with energy recovery. As indicated in Fig. 3c, collected FW is incinerated in a (moving-grate) furnace and the combustion gases

Table 3
Inventory data for the treatment of household food waste via in-vessel composting.

Parameter	Unit	Average	Min.	Max.	References
Compost	kg/t FW	300	200	400	Breitenbeck and Schellinger (2004)
Electricity	kWh/t FW	93	30	189	Eades et al. (2011); Levis and Barlaz (2011); Turner et al. (2016); Cabaraban et al. (2008); Levis and Barlaz (2013)
Diesel	MJ/t FW	21			Eades et al. (2011)
Natural gas (for steam)	MJ/t FW	0.07[a]			Eades et al. (2011)
Sulphuric acid, 98% (before dilution in scrubber)	kg/t FW	2.1			Assumption
Sodium hydroxide, 50% (before dilution in scrubber)	kg/t FW	0.1			Assumption
Water					
Air emissions treatment	kg/t FW	22			Assumption
Steam for pasteurisation	kg/t FW	0.03			Eades et al. (2011)
Emissions					
Nitrous oxide (biofilter)	kg/t FW	0.027	0	0.068	Amlinger et al. (2008); Leinemann (1998); Fisher (2006); Baky and Eriksson (2003); Martínez-Blanco et al. (2009); Thinkstep (2017)
NMVOC[b] (biofilter)	kg/t FW	0.11	0.007	0.33	Fletcher et al. (2014)
NMVOC[b] (windrow maturation)	kg/t FW	0.20	0.006	1.21	Cadena et al. (2009); Baky and Eriksson (2003); Martínez-Blanco et al. (2009)
Methane (windrow maturation)	kg/t FW	0.62	0.02	1.13	Amlinger et al. (2008); Leinemann (1998); Fisher (2006); Baky and Eriksson (2003); Martínez-Blanco et al. (2009); Thinkstep (2017)
Ammonia (windrow maturation)	kg/t FW	3.9	0.05	6	Cadena et al. (2009); Amlinger et al. (2008)
Ammonia (compost application)	kg/t compost	0.52	0.36	0.67	ADAS (2013); Nicholson et al. (2013)
Nitrate (compost application)	kg/t compost	0.11	0	0.22	~II~
Displaced mineral fertiliser					
Ammonium nitrate	kg/t compost	3.3	2.9	3.7	~II~
Triple superphosphate	kg/t compost	9.1			~II~
Potassium fertiliser, as K_2O	kg/t compost	6.5			~II~

[a] For 20 kWh of steam per tonne of food waste used for pasteurisation.
[b] Non-methane volatile organic compounds.

are used to generate steam, which then drives a turbine to produce power. The system has been credited for the latter, assuming the displacement of an equivalent amount of grid electricity. Based on currently operating UK facilities, the assumed capacity of the plant is 300,000 t of waste per year (Environment Agency, 2016) and a lifespan is 25 years.

The fly ash is removed from the flue gas by an electrostatic precipitator, while acid gases, heavy metals and particulates are treated in a multistage wet scrubber (Doka, 2009). The scrubber effluents are neutralised with hydrochloric acid and solids precipitated with iron chloride, before being sent for wastewater treatment. Nitrogen oxides are removed in selective non-catalytic reduction unit using ammonia. The treated flue gas contains predominantly carbon dioxide, nitrogen, water and trace amounts of other gases. These are detailed in Table 5, together with other inventory data.

The bottom ash has been assumed to be landfilled (Doka, 2009). Although it can be used as aggregate, since the major solid components of FW are combustible and FW contains negligible amounts of non-organic components, its contribution to the bottom ash is negligible. Therefore, the credits have not been considered for its potential use in the construction sector. The same applies for credits for any recyclables separated out of the MSW.

As the FW is incinerated together with the rest of the MSW, the electricity produced by the incineration plant has to be allocated between them. Lower heating value (LHV) has been used for this purpose, with the value of 3.8 MJ/kg used for FW (LoRe and Harder, 2012; Roberts, 2015) and 8.9 MJ/kg for MSW (Environment Agency, 2016). Therefore, based on the average electricity generated from MSW incineration of 597 kWh/t (Environment Agency, 2016), the average electricity output per tonne of FW is 255 kWh, with a range of 172–338 kWh/t. The on-site use of electricity averages 81 kWh/t (48–179 kWh/t) and imported electricity averages 6 kWh/t; both these values have been assumed to be independent of waste composition.

The incineration system has been simulated using an Ecoinvent incineration tool (Ecoinvent, 2008), taking into account the specific FW composition considered here and the electricity outputs from operating plants in the UK. As the only output is electricity, the uncertainty analysis is limited to the amount of electricity produced and the amount of electricity consumed on site (Table 5).

Data used for the estimation of the LCC are summarised in Table 6. The capital costs of the incineration facility of £213 m are based on recently-installed UK plants, with an average cost of £710 per tonne of annual capacity (Ryan, 2013; Amec Foster Wheeler, 2015; Green Investment Group, 2016). The capital costs have been allocated to FW by assuming equal share of capital costs for all waste treated, which equates to £28/t for a £213 m facility treating 300,000 t per year over a 25 year lifespan. The operating costs have been estimated using a costing model from an engineering contractor (Amec Foster Wheeler, 2015). The sale of electricity to the grid may be agreed within specific contracts but for this study it is assumed to be similar to the export tariff set by government (Ofgem, 2018). The median gate fee paid by local authorities to dispose of waste at incineration plants is £83 (WRAP, 2017a).

2.3.4.4. Landfilling with energy recovery. The food waste is disposed of at a sanitary landfill, which is a contained system where landfill gas (LFG) is collected and utilised for energy generation and leachate is collected for treatment (Fig. 3d). The capture of LFG is required in the UK to reduce methane emissions from landfills (Environment Agency, 2009).

The inventory data for landfilling are summarised in Table 7. These have been modelled through the Ecoinvent tool for sanitary MSW landfills (Ecoinvent, 2008). The tool enables specifying waste composition, enabling estimation of the impacts associated exclusively with the FW component of MSW. The model assumes 27% degradation of the carbon in FW over 100 years, with the remaining matter held within the landfill (Doka, 2009). The landfill covers an area of 90,000 m^2 and is 20 m deep. The leachate is sent via sewer to a municipal wastewater treatment plant and the resulting sludge is incinerated. The LFG capture rate from large modern landfill sites in the UK has been estimated at 68%, of which 9.1% is flared (Gregory et al., 2014). UK landfill plants use a gas engine to generate electricity and heat is not recovered due to a lack of infrastructure. The average gross engine efficiency is 40% with the net efficiency of 36% after the subtraction of the parasitic loads (Gregory et al., 2014). At the average capture rate, this equates to 61 kWh electricity exported per tonne of FW sent to landfill.

The uncertainty analysis focuses on the amount of LFG captured (55% to 85%) as this determines the direct methane emissions and the amount of electricity generated.

An overview of the life cycle costs of landfilling in the UK is given in Table 8. Capital costs of landfill facilities are not available in the public domain. However, as landfills tend to be extended continuously over their lifetime, capital costs of a landfill extension have been used instead, assuming a capacity of 175,000 t/year and a lifespan of the extension of 10 years (Hogg, 2002). The capital costs have been allocated to FW following the same approach as for incineration: all waste is assumed to carry an equal burden of the capital cost, equating to £12/t FW for the £21.7 m extension (Table 8). The costs have been adjusted for inflation to their current value (Bank of England, 2018). The generated electricity has been assumed to be sold to the grid at the export tariff set by government. The median gate fee paid by local authorities to dispose of waste at landfill is £107, including £84.40 landfill tax (WRAP, 2017a).

2.3.5. UK grid electricity

The 2017 UK electricity grid mix has been used to estimate the environmental impacts of the supplied and displaced electricity, with the generation mix shown in Fig. S1 in the SI. The individual electricity technologies have been modelled using inventory data from Ecoinvent (2016). The estimated environmental impacts of the grid mix can be found in Table S6 in the SI.

Table 4
Life cycle costs of the treatment of household food waste via in-vessel composting.

In-vessel composting costs	Unit	Amount	Reference
Capital (overnight) costs	£ m	5	Letsrecycle (2018, 2015, 2010, 2009, 2006); MRWA (2010)
Operating costs			
Maintenance	% of capital	3	Towler and Sinnott (2013)
Electricity import price	£/kWh_e	0.0973	BEIS (2017c)
Natural gas for steam	£/kWh_{th}	0.039	Eurostat (2018)
Red diesel for telehandler	£/L	0.503	Statista (2018)
Sulphuric acid	£/t	49	ICIS (2017)
Sodium hydroxide	£/t	341	S&P Global Platts (2016)
Disposal costs			
Gate fees	£/t FW	46	WRAP (2017a)
Revenue			
Compost	£/t compost	0.75	Horne et al. (2013)

Table 5
Inventory data for the treatment of household food waste via incineration with energy recovery.

Parameter	Unit	Average	Min.	Max.	Reference
Gross electricity produced	kWh/t FW	255	172	338	Environment Agency (2016)
Electricity consumed on site	kWh/t FW	81	48	179	Environment Agency (2016)
Natural gas (auxiliary fuel)	kWh/t FW	11.7			Ecoinvent (2008)
DeNox process consumables					
Ammonia	kg/t FW	0.4			~II~
Chromium oxide	g/t FW	0.25			~II~
Scrubber consumables					
Sodium hydroxide	kg/t FW	0.5			~II~
Hydrochloric acid	g/t FW	0.59			~II~
Wastewater treatment consumables					
Quicklime (CaO)	g/t FW	4.2			~II~
Iron (III) chloride	g/t FW	0.33			~II~
Bottom ash landfilled	kg/t FW	6.2			~II~
Residual waste landfilled (fly ash and dewatered sludge)	kg/t FW	2.7			~II~
Emissions to air[a]					
Ammonia	g/t FW	6.8			Ecoinvent (2008)
Carbon monoxide	kg/t FW	0.22			~II~
Methane	g/t FW	6.4			~II~
Nitrogen oxides	kg/t FW	0.27			~II~
Nitrous oxides	kg/t FW	0.036			~II~
Sulphur dioxide	g/t FW	5.8			~II~
Particulates, PM_{10}	kg/t FW	0.063			~II~
Heavy metals to air	mg/t FW	0.44			~II~
Emissions to water[a]					
Nitrate	kg/t FW	0.42			Ecoinvent (2008)
Sulphate	kg/t FW	4.1			~II~
Phosphate	kg/t FW	0.027			~II~
Heavy metals to water	mg/t FW	4.8			~II~

[a] Direct emissions from the combustion of food waste.

3. Results and discussion

The results are first presented for the functional unit of 1 t of household FW treated, starting with the environmental impacts and followed by the life cycle costs. The environmental and economic implications of managing household FW at the UK level are discussed in Section 3.3.

3.1. Life cycle environmental impacts

The environmental impacts of the four waste management routes for the treatment of 1 t of food waste are presented in Fig. 4 and Table S7 in the SI. The results refer to the average, minimum and maximum inventory values in Tables 1, 3, 5 and 7. The minimum and maximum values have been used in the uncertainty analysis to estimate the 90th and 10th percentile values of the impacts, shown as error bars in Fig. 4.

As can be inferred from Fig. 4, in the average case, anaerobic digestion has the lowest impacts for 13 out of 19 categories, including global warming potential. Incineration has the best values for marine eutrophication and agricultural land occupation, and landfill for terrestrial acidification, particulate matter formation and natural land transformation. In-vessel composting is overall the worst option, with 12 impacts being higher than for any other method. However, its urban land occupation is the lowest among the alternatives considered. Landfilling leads to the highest global warming potential, alongside human toxicity, marine eutrophication and urban land occupation. Incineration is the least preferred method for metal depletion, freshwater eutrophication and particulate matter formation. These results are discussed in more detail in the following sections.

3.1.1. Primary energy demand (PED)

For the average case, AD has a net-negative PED of −1.93 GJ/t FW. This is primarily attributable to the grid electricity displaced but also to the avoided production of mineral fertilisers. Incineration with energy also has a net-negative PED (−0.9 GJ/t FW) for the average case, but this is highly dependent on the efficiency of energy recovery and parasitic use. For example, in the 90th percentile case, the electricity from FW cannot cover the parasitic load and the system becomes a net user of energy (0.07 GJ/t FW). The modest recovery of energy from the landfill gas is not sufficient to exceed the energy demand of the landfill process and, in the average case, it is a net consumer with a PED of 0.14 GJ/t FW. In the 10th percentile case, where landfill gas capture rate approaches the upper range of 85%, the process becomes a net saver of PED at −0.15 GJ/t FW. IVC has the highest PED, with 1.31 GJ/t FW in the average case and ranging from 0.82 to 1.99 GJ/t FW. This is due to the energy-intensive air handling system required for the

Table 6
Life cycle costs of the treatment of food waste via incineration with energy recovery.

Incineration costs	Unit	Amount	Reference
Capital (overnight) costs[a]	£ m	213	Ryan (2013); Amec Foster Wheeler (2015); Green Investment Group (2016)
Operating costs			
Maintenance cost	£/t FW	6	Amec Foster Wheeler (2015)
Consumables	£/t FW	5	Amec Foster Wheeler (2015)
Auxiliary fuel (natural gas)	£/kWh_{th}	0.039	Eurostat (2018)
Disposal of ashes	£/t FW	14	Amec Foster Wheeler (2015)
Disposal costs			
Gate fees	£/t FW	83	WRAP (2017a)
Revenue			
Electricity export price	£/kWh_e	0.05	Ofgem (2018)

[a] Equal to £28 per tonne FW treated, based on the treatment of 300,000 t of waste per year over a 25 year lifespan.

Table 7
Inventory data for landfilling of household food waste with energy recovery.

Parameter	Unit	Average	Min.	Max.	Reference
Gross electricity produced	kWh/t FW	68	55	85	Gregory et al. (2014)
Electricity consumed on site	kWh/t FW	6.7			Gregory et al. (2014)
LFG capture efficiency	%	68	55	85	Gregory et al. (2014)
Fraction of captured LFG flared	%	9.1			Ecoinvent (2008)
Heat (light fuel oil)	kWh/t FW	0.14			~II~
Leachate production	m^3/t FW	2.5			~II~
Leachate treatment					
Iron sulphate	g/t FW	16			~II~
Aluminium sulphate	g/t FW	4.4			~II~
Titanium dioxide	g/t FW	0.23			~II~
Sludge incineration					
Chromium oxide	g/t FW	0.0048			~II~
Hydrochloric acid	g/t FW	0.00053			~II~
Natural gas (auxiliary fuel)	kWh/t FW	0.39			~II~
Sodium hydroxide	g/t FW	0.41			~II~
Iron (III) chloride	g/t FW	22			~II~
Quicklime (CaO)	g/t FW	0.074			~II~

controlled atmosphere. Furthermore, the system credits for displacing the production of mineral fertilisers are small, as only a total of 19 kg of mineral fertilisers are avoided through the composting of 1 t of FW.

3.1.2. Global warming potential (GWP)

AD has a net-saving in GHG emissions of −31.6 kg CO_2-eq./t FW due to the displacement of grid electricity. Incineration with energy recovery has the second lowest GWP at −5 kg CO_2 eq./t FW. However, in the worst case, where the electricity generated is near the lower limit and the on-site demand is near the upper limit, incineration has the potential to become a net contributor to the GWP (31 kg CO_2 eq./t FW for the 90th percentile). It should be noted that these results correspond to FW incineration alone, not taking into account the energy recovered from other MSW fractions, so that the overall GWP from incineration is still likely to be net-negative (Jeswani and Azapagic, 2016).

The high PED of in-vessel composting leads to a GWP of 77.5 kg CO_2 eq./t FW. This is attributable to the electricity use (34 kg CO_2 eq.) and FW transportation (26 kg CO_2 eq.) but also emissions of nitrous oxide and methane from process gases and windrow maturation (15.4 kg CO_2 eq.). Some studies (Smith et al., 1997; Favoino and Hogg, 2008) have suggested that the application of compost to agricultural land can sequester carbon in the soil over long time periods, particularly depleted soils, and therefore has a potential for carbon mitigation. There is currently no defined methodology to quantify this effect and hence it cannot be incorporated into these results. However, this should be borne in mind when interpreting the GWP of composting.

Landfill has the highest GWP (195 kg CO_2 eq.), predominantly due to the untreated emissions of methane: in the average case, 32% of LFG is not captured (Section 2.3.4.4), approximately 50% of which is methane. Landfill remains the worst option, followed by IVC, across the range of values considered in the uncertainty analysis. In the best case, incineration is comparable to the average GWP of AD and vice versa: in the worst case, the impact from AD is similar to the average GWP of incineration (Fig. 4).

3.1.3. Fossil depletion (FD) and metal depletion (MD)

The ranking of the waste treatment options for FD correlates with the amount of electricity generated and displaced from the grid, with AD being the best option (−19.8 kg oil eq./t FW), followed by incineration (−5 kg oil eq.) and landfill (8.3 kg oil eq.). At 20.3 kg oil eq./t FW, IVC has the highest FD as it recovers no energy. The waste transportation (8.8 kg oil eq./t FW) is the largest contributor (59%–83%) to FD for AD, incineration and landfill, while for IVC, it is the use of grid electricity (9.9 kg oil eq./t FW). In both the worst and best cases, AD and IVC have the lowest and highest FD, respectively. In the worst case, incineration has a similar impact as landfill for the average case (Fig. 4).

The MD is predominantly related to the construction and operation of the facilities, as well as transportation. The incineration plant is the most process- and equipment-intensive facility and this is reflected in the highest impact of 1.6 kg Fe eq./t FW. IVC is the second worst option for MD (0.73 kg Fe eq./t FW), primarily due to the construction of the facility (0.8 kg Fe eq.) and grid electricity infrastructure (0.4 kg Fe eq.). The displacement of mineral fertilisers provides a total credit to the IVC system of 1.1 kg Fe eq./t FW. The landfill facility itself has the lowest MD but, with the credits applied, AD emerges as the best option overall MD (−1 kg Fe eq./t FW). There is no change in the treatment rankings across the range of impact values.

3.1.4. Freshwater, marine and terrestrial ecotoxicity (FET, MET and TET) and human toxicity (HT)

The ecotoxicities and HT all follow the same trend, with AD having the lowest impacts, followed by incineration and landfill. An exception is HT of landfill which is the highest across the options considered. For this impact, IVC is the second best option after AD, as it benefits from the credit for displacing phosphate fertiliser (−8.8 kg 1,4-DB eq./t FW). However, IVC has the greatest impacts for all three ecotoxicities. These trends are aligned with the electricity use/generation and the related credits. For all four toxicities, AD remains the preferred option in both the best and worst cases. In the worst case, incineration is comparable to the average FET and MET of landfill and has higher TET and HT than the averages for both landfill and IVC (Fig. 4).

3.1.5. Freshwater and marine eutrophication (FE and ME)

Incineration has the highest FE at 4.8 g P eq./t FW, due to the phosphates in the fly ash which are emitted to freshwater following both wastewater treatment and its landfilling. The impact from landfilling is lower at 2.8 g P eq./t. The credits for mineral fertilisers cancel out the impact from electricity used for IVC, resulting in a net-negative FE of −0.9 g P eq./t FW. Nevertheless, AD is the best option, saving −11.6 g P eq./t FW, as the electricity and fertiliser credits exceed the process emissions. Across the range of values used in the uncertainty analysis, AD remains the lowest-impact option. In the worst case, IVC approaches the average. The average FE of landfill is comparable to the worst-case impact of IVC and best-case impact from incineration.

Landfilling has the highest ME (7.4 kg N eq./t FW) due to the emissions of ammonia and nitrate from leachates. This is more than ten times greater than the impact of AD (0.7 kg N eq./t FW), the second worst option. The majority of the ME from AD derives from ammonia emissions following the application of digestate in agriculture. However, due to the uncertainty in these emissions, the 10th percentile case has a net saving of −0.03 kg N eq./t FW. The overall ME for IVC is estimated at 0.4 kg N eq./t FW. Although some ammonia is removed within the controlled atmosphere at the IVC facility and the final

Table 8
Life cycle costs of landfilling of household food waste with energy recovery.

Landfilling costs	Unit	Amount	Reference
Capital (overnight) cost[a]	£ m	21.7	Hogg (2002)
Operating cost	£ m/y	7.5	Hogg (2002)
Disposal costs			
Gate fees (inc. tax)	£/t FW	107	WRAP (2017a)
Landfill tax	£/t FW	84.40	HMRC (2017)
Revenue			
Electricity export price	£/kWh$_e$	0.05	Ofgem (2018)

[a] Equal to £12 per tonne FW treated, based on the treatment of 175,000 t of waste per year over a 10 year lifespan.

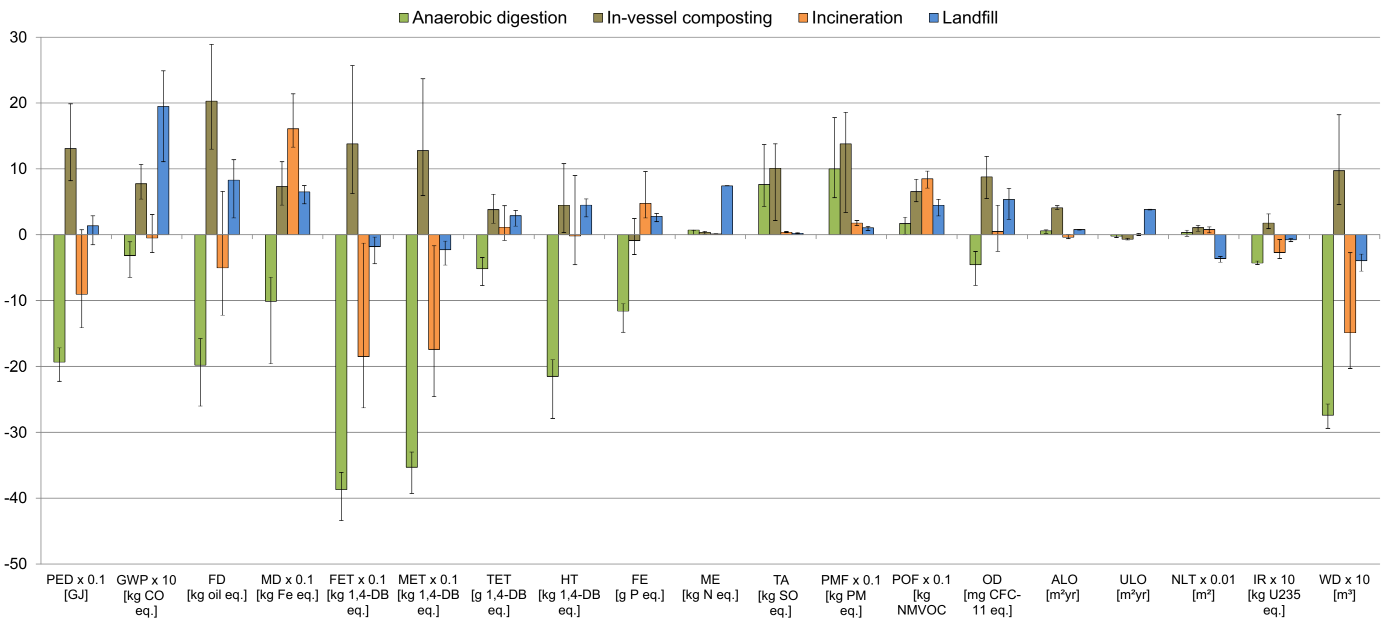


Fig. 4. Environmental impacts of treating 1 t of food waste via anaerobic digestion, in-vessel composting, incineration with energy recovery and landfill with energy recovery. [The graph bars correspond to the average inventory data in Tables 1, 3, 5 and 7. The error bars represent the 90th (higher values) and 10th (lower values) percentiles estimated in the uncertainty analysis using the minimum and maximum inventory values given in the above tables. Some impacts have been scaled to fit and should be multiplied by the factor shown on the x-axis to obtain the original values. PED: primary energy demand; GWP: global warming potential; FD: fossil depletion; MD: metal depletion; FET: freshwater ecotoxicity; MET: marine ecotoxicity; TET: terrestrial ecotoxicity; HT: human toxicity; ME: marine eutrophication; FE: freshwater eutrophication; TA: terrestrial acidification; PMF: particulate matter formation; POF: photochemical oxidants formation (summer smog); OD: ozone depletion; ALO: agricultural land occupation; ULO: urban land occupation; NLT: natural land transformation; IR: ionising radiation; WD: water depletion; DB: dichlorobenzene. NMVOC: non-methane volatile organic compounds.]

product is more stable than digestate, some ammonia is still emitted during open-air windrow maturation. Finally, incineration has the lowest ME (0.1 kg N eq./t FW) as the flue gases are treated to limit emissions of nitrogen oxides.

3.1.6. Terrestrial acidification (TA) and particulate matter formation (PMF)

TA is high for both IVC (10.1 kg SO_2 eq./t FW) and AD (7.6 kg SO_2 eq./t FW), with the significant contributory factor being the ammonia released to the air during compost maturation and digestate application, respectively. The small quantities of nitrogen oxides and sulphur dioxide emitted from incineration result in 0.4 kg SO_2 eq./t FW, compared to 0.2 kg SO_2 eq./t FW for landfill, which has limited ammonia emissions to air.

PMF follows the same trend as TA, with the impact primarily dependent on nitrogen species emitted to the air. Incineration has the lowest PMF at 0.18 kg PM_{10} eq./t FW, followed by landfilling with 0.11 kg PM_{10} eq. The impact from AD is nearly ten times higher (1.00 kg PM_{10} eq.) than from landfilling, and that from IVC nearly eight times greater than from incineration (1.38 kg PM_{10} eq.).

The uncertainty analysis for both TA and PMF shows the same trends, with AD and IVC having comparable impacts in the worst case but IVC being a better option than AD in the best case. The latter is primarily due to the lower value of ammonia volatilisation during windrow maturation.

3.1.7. Photochemical oxidants formation (POF)

Although treated before release, the nitrogen oxides and non-methane volatile organic compounds (NMVOC) in flue gases from incineration result in this option having the highest average POF (summer smog) of 0.85 kg NMVOC eq./t FW. The electricity use and NMVOC from the composting process place IVC as the second worst alternative for this impact (0.66 kg NMVOC/t FW). Emissions of nitrogen oxides and methane are the main cause of POF from the landfill system, estimated at 0.45 kg NMVOC eq./t FW. Other than minor combustion emissions from the biogas utilisation, the POF of AD is caused by transportation (0.17 kg NMVOC eq./t FW) which is the best option for this impact category.

Considering the results of the uncertainty analysis, in the worst case, the POF of IVC is similar to the average impact of incineration and vice versa: the POF of incineration in the best case is comparable to the average value for IVC (Fig. 4).

3.1.8. Ozone depletion (OD)

The OD is influenced by the electricity credits, with AD having the net-negative impact of −4.6 mg CFC-11 eq./t FW. Incineration is ranked second best with 0.5 mg CFC-11 eq./t FW. Landfill and IVC have a much higher impact at 5.4 and 8.8 mg CFC-11 eq./t FW, respectively. For all treatment routes, the OD is predominantly due to the emissions from the waste collection and transportation (4.7 mg CFC-11 eq./t FW). The trends in the OD estimated in the uncertainty analysis (Fig. 4) reflect the ranges of electricity used or credited: in the best case, the OD of IVC is similar to the average value for landfill and in the worst case, the impact of incineration is comparable to the average for landfill.

3.1.9. Agricultural and urban land occupation (ALO and ULO) and natural land transformation (NLT)

The IVC and AD facilities have been assumed to occupy agricultural, and landfill and incineration urban land. The AD plant has a small land footprint as all processing is enclosed in vessels, whereas the maturation of composting typically occurs in large windrows, requiring more land. However, it is the electricity used in IVC that results in this option having the highest ALO of 4.1 m^2 year/t FW.

Landfilling is the only treatment with a significant ULO (3.9 m^2 year/t FW). However, landfills are reclaimed for natural land at the end of their lives (DEFRA, 2004). Therefore, with the related credit, the landfill system has a net-negative NLT of −0.04 m^2/t FW, rendering it the best option for this impact.

There are no changes in the rankings of the treatment options when considering the range of values. For NLT in the best case, both IVC and incineration are comparable to the average NLT of AD and, in the worst case, the NLT of AD is comparable to the average values for IVC and incineration (Fig. 4).

3.1.10. Ionising radiation (IR) and water depletion (WD)

None of the treatment plants requires large amounts of water or cause any direct ionising radiation. Therefore, the ranking of the options for these two impacts is directly correlated with the electricity used and credited for each treatment process. Consequently, in the average case, AD is the best and IVC the worst option for both impacts. Considering the worst case, incineration is comparable to the average impacts of landfilling. For the best case, there is no change to the trend found for the average case.

3.1.11. Sensitivity analysis: electricity source

As the above results suggest, many impact categories are influenced significantly by the use of grid electricity and the credits for its displacement, which are in turn determined by the electricity mix considered. Therefore, this section explores the effect on the impacts of assuming the use and displacement of the following alternative electricity sources: a combined cycle gas turbine (CCGT) powered by natural gas, onshore wind turbines and a solar photovoltaics installed on the ground. CCGT has been chosen as currently the most prevalent electricity source in the UK (BEIS, 2018). Furthermore, both solar and wind are becoming more important in the UK market and either could be installed at an IVC facility. All three options have been modelled using Ecoinvent (2016) data and the results are compared with the grid electricity in Fig. 5 and Table S8, considering the average case for all the electricity sources. Landfill is omitted as it is the least dependent on the electricity supply. For clarity and brevity, only selective impacts are shown, exhibiting differing trends.

The significant differences between CCGT and the UK grid are higher FD and lower MD, HT, FE and OD for CCGT. However, the ranking of the treatment options remains the same, regardless of whether the systems have been credited for grid or CCGT electricity (Table S8).

If AD and incineration are credited with displacing wind or solar energy, then both treatment options become net contributors to GWP and FD as the credits are lower compared to those for the grid electricity. On the other hand, the impact from IVC is reduced on the base case with these two renewable energy sources. For wind power, the three treatment methods converge in terms of GWP, with AD at 48.1 kg CO_2 eq./t FW, incineration at 47.6 kg CO_2 eq., and IVC at 48.3 kg CO_2 eq.

Solar PV, and to a lesser extent wind, is associated with greater MD and toxicity impacts; therefore, their use instead of grid electricity benefits AD and disadvantages IVC. By contrast, ME, TA and PMF are predominantly dependent on direct process emissions and are not significantly affected by the source of electricity.

While it is not realistic to assume the electricity credits would solely consist of wind or solar energy, this analysis demonstrates that the environmental benefits of AD are highly dependent on system credits. IVC is equally disadvantaged by considering only grid electricity, while solar panels or a wind turbine could be installed on site to cover the electricity demands and thus improve on their environmental impacts.

3.1.12. Comparison of results with literature

As mentioned in the introduction, four other LCA studies of household food waste management have been found in the literature. However, only three presented the results in a way that could be compared to the results in the current work: one for Ireland (Oldfield et al., 2016), another for a London borough (Evangelisti et al., 2014) and the third for Seoul (Lee et al., 2007). As all three used the CML 2001 method (Guinée, 2002) to estimate the impacts, to enable comparisons, the results in the current study have been recalculated using the same method. All three studies considered GWP, acidification potential (AP) and eutrophication potential (EP), as discussed below.

The GWP values are compared in Fig. 6. Although the absolute values differ, the same overall trend can be observed across all the studies, with landfill having the highest impact, followed by composting and incineration. AD was the best option in all studies in which it was included.

The higher GWP values for landfill found by Oldfield et al. (2016) and Evangelisti et al. (2014) are due to the assumptions regarding emissions of LFG. Both studies assumed that 50% of LFG was emitted to the atmosphere, compared to 32% in this study. Furthermore, Oldfield et al. (2016) did not consider utilisation of the LFG for energy recovery. The high GWP for the IVC in this study is attributable to the electricity required to maintain a controlled atmosphere in the facility, whereas neither of the compared studies mentioned any significant electricity usage at the plant. Furthermore, the GWP of both incineration and AD is dominated by the credits for the displaced grid electricity and all three studies in the literature assumed electricity mixes with a higher share of coal power; thus, they had higher credits than in this study.

For the AP, the impact from incineration and landfill is similar to that reported in the literature (Fig. 7a). While both Oldfield et al. (2016) and Lee et al. (2007) also found composting to have the highest AP, the impact estimated in the current study is more than four times greater. Similarly, the AP of AD is far greater in this study than in the literature. This is due to different assumptions for the emissions of ammonia from both digestate and compost. In this study, these are based on large-scale UK field tests whereas Evangelisti et al. (2014) used lower values derived from a theoretical model; Oldfield et al. (2016) did not state any emission values. If the lower values from UK field tests are used instead of the average, the AP of AD reduces from 5 to 2.2 kg SO_2 eq./t FW but is still significantly greater than the literature values (Fig. 7). Further differences in the results can be attributed to the electricity credits – the much lower share of coal in the current study means lower credits for the displaced electricity compared to the literature. Coal power has a high AP compared to the grid average (Stamford and Azapagic, 2012), which helps to explain why the AP in the other two studies is net-negative.

For the EP, both incineration and AD fall within the ranges of the other studies (Fig. 7b). However, the impact of the other two treatment options is much higher in this study. For the landfill system, both Oldfield et al. (2016) and Evangelisti et al. (2014) used GaBi to determine the impact, which leads to a much lower estimate than the Ecoinvent model used in this study. The larger EP of IVC obtained in this study is attributable to the greater electricity consumption associated with the controlled-atmosphere facility and ammonia emissions from the open-air windrow maturation.

Bernstad and la Cour Jansen (2011) also considered AD, IVC and incineration, focusing on Sweden; however, they did not present the results in a format that can be directly compared to the results in this work. Nevertheless, they found that both AD and IVC had a net saving in GWP, while incineration had a slightly net-positive impact, mainly attributable to the life cycle of the plastic bags used to collect waste – a factor not considered in this study. The nutrient enrichment (NE) and AP were comparatively very high for IVC and NE was also high for the digestate applied to clay soils. This mirrors the high EP and AP found in this paper with regards to applying processed organic waste in agriculture.

3.2. Life cycle costs

The LCC of treating a tonne of FW by each of the four methods are presented in Fig. 8. These have been estimated assuming that the 4.9 Mt of household FW collected by local authorities annually is treated as described in Section 2.3.1. As can be seen in the figure, incineration has the lowest costs estimated at £71 per tonne of FW, followed by IVC with £80/t FW. Landfilling is the most expensive option, costing £123/t while the cost of AD is £110/t. Collection of FW is the main contributor to the total costs of AD and IVC; gate fees dominate for the other two methods. These costs are discussed in more detail below, broken down into the costs to the local authorities responsible for the waste

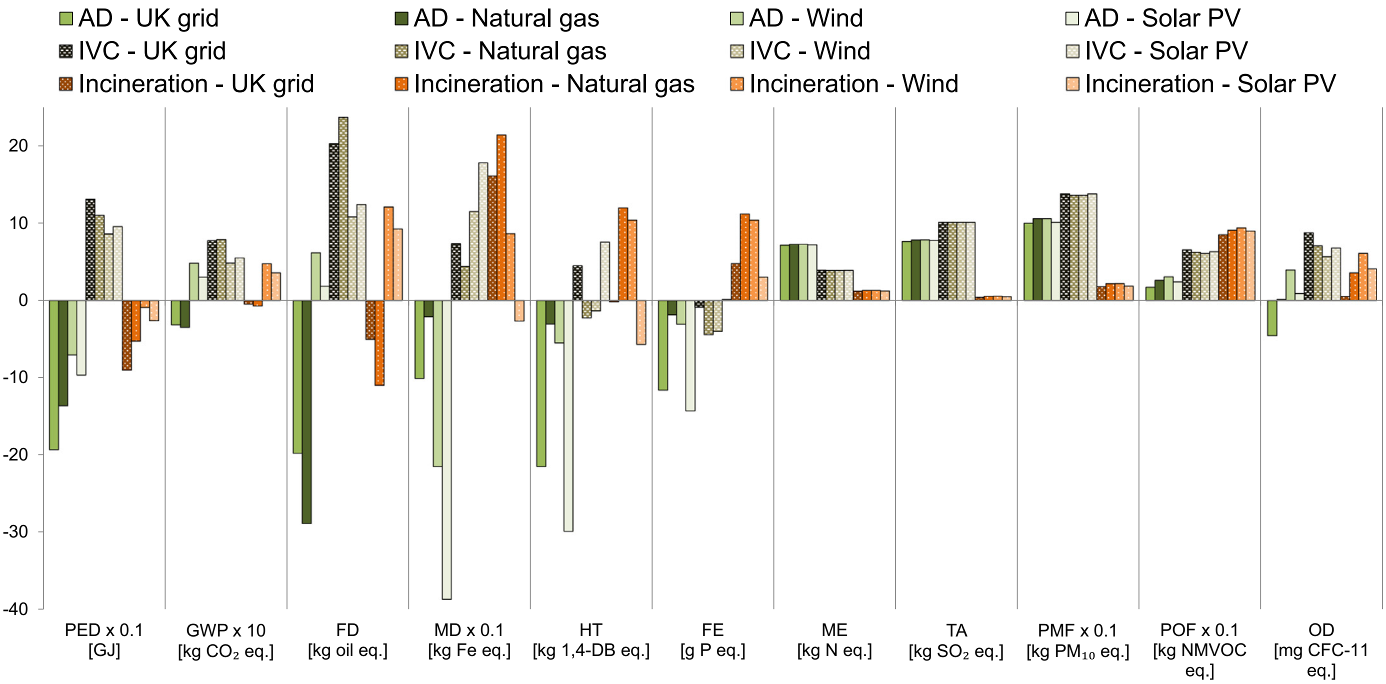


Fig. 5. Sensitivity analysis for different electricity options used to supply and credit the treatment of 1 t of food waste. [AD: anaerobic digestion; IVC: in-vessel composting. For impacts nomenclature, see Fig. 4. Some impacts have been scaled to fit and should be multiplied by the factor shown on the x-axis to obtain the original values.]

collections and the facility operators treating the waste. All the costs have been rounded to the nearest pound.

3.2.1. Costs to local authorities

These comprise costs of collecting the FW and gate fees paid to the operator of treatment facilities (Eq. (2)). Overall, incineration has the lowest cost to local authorities at £108/t, followed closely by IVC at £109/t. The cost for AD and landfilling are significantly higher at £137/t and £132/t, respectively.

3.2.1.1. Collection costs. The costs of collecting FW separately from the rest of MSW are lower per household (£7) compared to residual waste (£14) as the food collections require a smaller collection vehicle (WRAP, 2008). However, as the total amount of FW collected per household is much lower than of residual MSW, the collection costs are far greater per tonne of FW. With the current amounts of FW collected (Section 2.3.1), separate FW collection is estimated here to cost £108/t while collecting mixed food and garden waste costs £63/t. Based on the 14.8 Mt of household MSW collected for incineration or landfill (see Table S1 in the SI), the current collection cost of FW within MSW is £25/t.

3.2.1.2. Gate fees. As shown in Fig. 8 (and specified previously in Section 2.3), the gate fees for the different treatment options vary significantly. Landfill is the most expensive route at £107/t FW; however, £84 of this is landfill tax, with the average gate fee received by the operator of £23/t. The tax has increased year-on-year since its introduction in 1996, with the intention of reducing the amount of waste sent to landfill in compliance with the EU Landfill Directive (EC, 1999). The gate fees for incineration are set below that of landfill, at an average of £83/t FW, taking into account the difference in fees for facilities with contracts agreed before 2000 (£56/t FW), and those with contracts agreed subsequently (£91/t).

The average gate fees for IVC are £46/t FW and for AD £29/t. The AD facilities can afford to charge the lowest gate fees as significant revenue to the operator may be attained from the electricity generated from the biogas and associated incentives. Despite the lower gate fees, AD is not competitive with incineration for local authorities due to the higher costs of separate collection of FW. However, if local authorities can maximise the separate collection they can reduce collection costs per tonne.

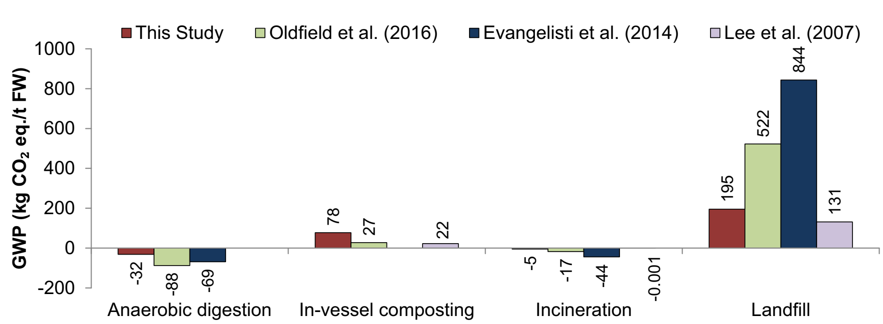


Fig. 6. Comparison of the global warming potential (GWP) with the literature.

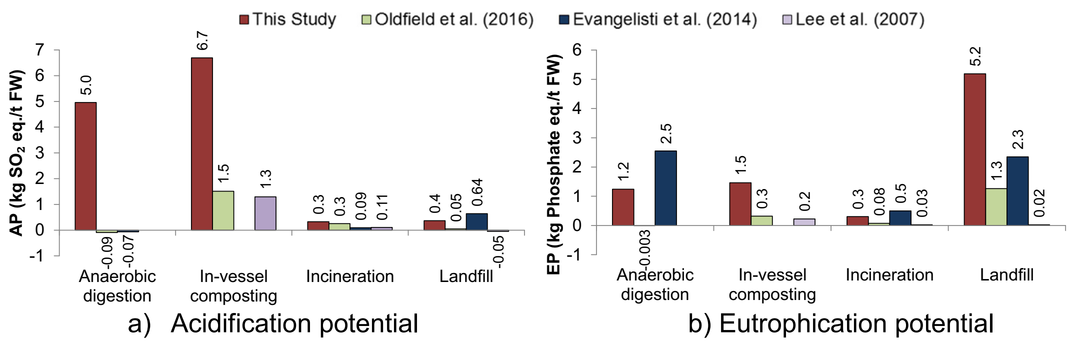


Fig. 7. Comparison of the acidification potential (AP) and eutrophication potential (EP) with the literature.

3.2.2. Costs to operators

Operators bear the costs of constructing and operating treatment facilities, which are offset through the gate fees they receive from local authorities, any government incentives and sales of recovered energy and/or materials (Eq. (3)). As a result, operators make a profit across the treatment options, which is the highest for incineration (£37/t FW), followed by IVC (£29/t), AD (£27/t) and finally landfilling (£8). These are detailed below.

3.2.2.1. Capital and operating costs. Despite having the highest capacity to treat waste, the incineration plant has the highest (overnight) capital costs at £28/t FW due to the large amount and scale of process equipment required. The capital costs of the AD and IVC facilities are £7/t FW and £4/t FW, respectively. Both plants only require a simple agitated processing vessel with the addition of a gas collection and CHP system for the AD plant. Landfill has a capital cost of £12/t FW and requires the largest land area, land preparation and investment in the vehicles, but only limited process equipment is needed for the gas engine and leachate treatment.

The incineration plant also attracts the highest operating costs of £26/t FW, largely due to the costs of disposing the bottom and fly ashes, and maintenance, but also due to the consumables needed to treat the flue gas and the resulting wastewater. The operating costs of AD (£8/t FW) are limited to the maintenance and replacement of the parts as the electricity and heat demand are supplied internally. IVC has higher operating costs (£14/t FW), of which more than half is related to the electricity costs (£9/t FW). Landfilling is the cheapest to operate at £5/t FW. This is due to limited waste processing and large quantities of waste, leading to low per-tonne costs.

3.2.2.2. Revenue. The revenue for all operators predominantly consists of the gate fees received from the local authority. The incineration facility brings in £8/t FW for the electricity it exports, which represents 9% of their revenue. Hence, incineration creates the most revenue for the operators compared to the other three options. Landfill generates £3/t FW revenue from the sale of electricity, or 12% of the total revenue. The income for IVC from selling compost is negligible at 0.5% of the total revenue. AD is the only treatment option that brings a significant product revenue; the generation of electricity attracts a tariff equal to £4/t FW while the electricity exported to grid brings in an additional £13/t FW. On average, the digestate has no market value and the operators must pay users to take it away at a rate equivalent to £4/t FW. With incentives, the revenue from AD products makes up 35% of total revenue and 31% without incentives.

3.2.3. Overall environmental and economic sustainability

Integrating the environmental impacts and economic costs, different treatment options are compared in Fig. 9a to identify the most sustainable alternative. Assuming equal importance of all environmental impacts and LCC, it can be seen that incineration is ranked the best and landfill the worst option. Although AD has 13 lowest impacts, it has the third highest costs, which places it overall the second in the ranking order. By contrast, IVC has the highest impacts but the second best LCC, which puts it in the third place. Therefore, neither AD nor IVC emerge as the most sustainable options per tonne of waste treated, contrary to being promoted as sustainable options in the context of a circular economy.

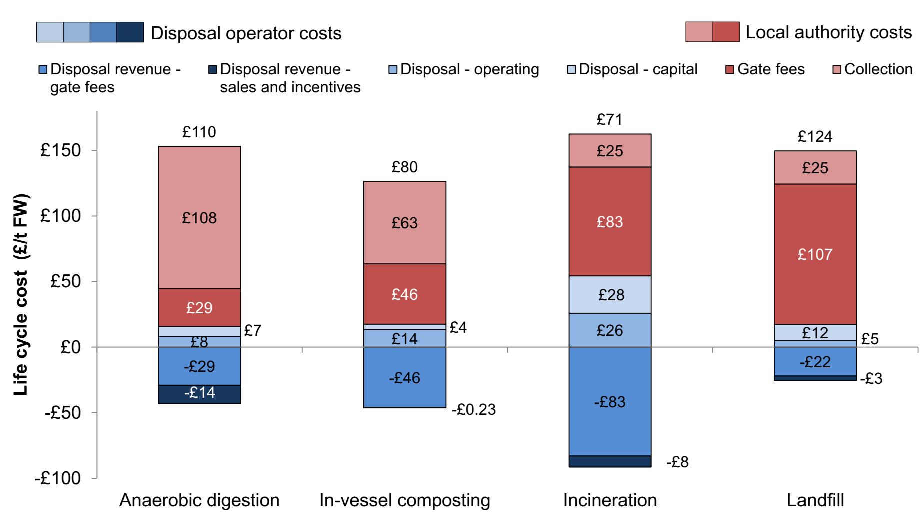


Fig. 8. Life cycle costs of collection and treatment of household food waste, showing the costs to local authorities and operators of treatment facilities.

3.3. Environmental impacts and costs at the UK level

This section presents the environmental impacts and costs of treating 4.9 Mt/year of FW collected by local authorities from households, separately from other MSW. These have been estimated by scaling up the impacts and costs of the individual treatment processes on a per-tonne basis, discussed in the previous sections, and taking into account the annual amounts of FW treated by each method (see Section 2.3.1). The results are discussed below.

3.3.1. Life cycle environmental impacts

As indicated in Fig. 10, the current FW treatment system is estimated to be a net energy producer with a primary energy demand of −1.93 PJ/year. IVC is the main energy consumer (0.65 PJ) as it is the only treatment process that produces no electricity; landfilling also has a net-positive consumption of energy (0.23 PJ) due to waste transportation and fuel needed for landfill machinery. The saving in PED does not translate to a saving in GWP, which amounts to 0.34 Mt CO_2 eq. This represents 0.07% of total UK emissions (BEIS, 2017a). The emissions of methane from landfills are the largest contributor (95%) to the total GWP.

Incineration treats the greatest share of FW (49.7%) and has the greatest influence on seven of the other impacts, resulting in net savings in FET, MET, IR and WD related to the displaced electricity. However, it is also the highest contributors to MD, FE and POF, with the first due to construction materials and operating consumables, and the last two due to emissions from waste combustion. Landfill has the second largest share of FW treated (33.7%). As a result, it has the greatest contributions to ME, OD, TET, FD and ULO but leads to net savings in NLT due to land restoration at the end of life. IVC and AD have small treatment shares (10.1% and 6.4%) but the biggest influence on TA and PMF due to ammonia released from digestate application and compost maturation.

For context, the results for the current situation are compared with a hypothetical case whereby all FW is assumed to be treated exclusively via AD, IVC, incineration or landfill. The FW collections remain the same as for the current situation, with the waste treated via AD collected separately and that treated via IVC mixed with garden waste; for incineration and landfilling, the waste is collected co-mingled with the rest of the MSW.

The results in Table S9 in the SI suggest that treating all the FW via AD or incineration would each result in savings across 15 of the impacts compared to the current situation, with AD having the highest savings in 13 categories. However, in the case of AD, acidification would increase by a factor of four and particulates by a factor of three. For incineration, summer smog would be higher by 30% and metal depletion by 56%. Treating FW exclusively via IVC or landfilling would lead to savings in ten and five impacts, respectively, compared to the current situation.

However, avoiding waste would lead to far greater savings in environmental impacts than the treatment by any of the methods. Focusing on GWP (due to a lack of data for the other impacts), the results of this work suggest that avoiding 2.9 Mt/year of avoidable and 0.8 Mt/year of potentially avoidable fractions would save 14.09 Mt CO_2 eq./year. This is based on the estimated GWP of avoidable and potentially avoidable waste of 3.74 t CO_2 eq./t FW (Quested et al., 2013a), equivalent to the total of 13.84 Mt CO_2 eq./year. Adding to this 0.26 Mt CO_2 eq./year generated during the treatment of this waste (based on 0.34 Mt CO_2 eq./year for 4.9 Mt FW (Fig. 10), scaled down to 3.7 Mt/year of avoidable waste) yields the total of 14.09 Mt CO_2 eq./year avoided annually by not generating this amount of waste. This is nearly two orders of magnitude higher than the saving of 0.15 Mt CO_2 eq./year in the best treatment case when all FW is treated via AD. Thus, in the food sector, the UK has a far greater opportunity to meet carbon reduction targets by preventing food waste than by changing treatment methods.

3.3.2. Life cycle costs

The estimated annual LCC of treating FW are summarised in Fig. 11 for the current situation. For context, an analysis similar to that for the environmental impacts in the previous section has also been performed for the LCC, considering the exclusive treatment of the waste by one of the four treatment methods. Given the assumption that the 4.9 Mt/year of household FW managed by local authorities is treated exclusively by each management route, the increase in FW collected separately or with garden waste reduces the collection costs per tonne compared to the current costs. AD requires separate food waste collections, with the provision extended to all UK households and all FW collected the collection costs have been estimated to reduce from the current value of £108/t to £39/t. For exclusive treatment via IVC, it has been assumed all food waste is collected co-mingled with garden

Indicator	Anaerobic digestion	In-vessel composting	Incineration	Landfill
PED	1	4	2	3
GWP	1	3	2	4
FD	1	4	2	3
MD	1	3	4	2
FET	1	4	2	3
MET	1	4	2	3
TET	1	4	2	3
HT	1	3	2	4
FE	1	2	4	3
ME	3	2	1	4
TA	3	4	2	1
PMF	3	4	2	1
POF	1	3	4	2
OD	1	4	2	3
ALO	2	4	1	3
ULO	2	1	3	4
NLT	2	4	3	1
IR	1	4	2	3
WD	1	4	2	3
Avg. env. score	1.5	3.4	2.3	2.8
LCC	3	2	1	4
Total score	4.5	5.4	3.3	6.8
Overall ranking	2	3	1	4

Legend	1	2	3	4
	Best			Worst

a) Comparison per tonne of food waste treated

Indicator	Current situation	Anaerobic digestion	In-vessel composting	Incineration	Landfill
PED	3	1	5	2	4
GWP	3	1	4	2	5
FD	3	1	5	2	4
MD	4	1	3	5	2
FET	3	1	5	2	4
MET	3	1	5	2	4
TET	3	1	5	2	4
HT	3	1	4	2	5
FE	3	1	2	5	4
ME	4	3	2	1	5
TA	3	4	5	2	1
PMF	3	4	5	2	1
POF	3	1	4	5	2
OD	3	1	5	2	4
ALO	2	3	5	1	4
ULO	4	2	1	3	5
NLT	2	3	5	4	1
IR	3	1	5	2	4
WD	3	1	5	2	4
Avg. env. score	3.1	1.7	4.2	2.5	3.5
LCC	4	1	2	3	5
Total score	7.1	2.7	6.2	5.5	8.5
Overall ranking	4	1	3	2	5

Legend	1	2	3	4	5
	Best				Worst

b) Comparison with current situation for the total annual food waste treated (4.9 Mt/yr)

Fig. 9. Heat maps comparing different treatment options. [Total score of an option is estimated by summing up its rankings in each environmental impact and life cycle costs (LCC). To avoid bias, the total score for the environmental impacts is divided by the number of impacts before adding it to the LCC score. For the impact acronyms, see Fig. 4.]

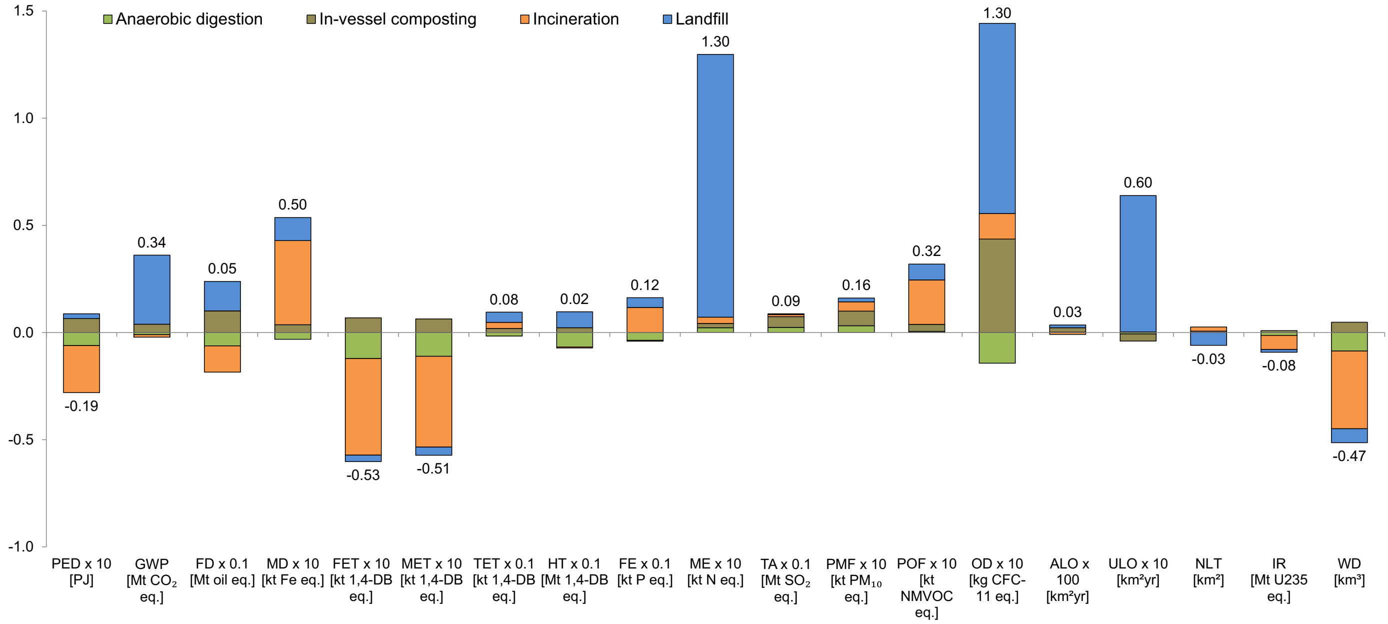


Fig. 10. Annual environmental impacts of treating 4.9 Mt of household food waste in the UK [2.44 Mt FW/year is incinerated, 1.65 Mt landfilled, 0.5 Mt treated via in-vessel composting and 0.31 Mt via anaerobic digestion. For impacts nomenclature, see Fig. 4. Some impacts have been scaled to fit and should be multiplied by the factor shown to obtain the original values].

waste; the increased collection of food waste has been estimated to reduce the current cost of £63/t to £40/t.

As can be seen, the total costs of FW treatment amounts to £452 m/year, including the operators' revenue of £302 m/year. Local authorities spend in total £579 m/year, or which £168 m/year is for waste collection and the rest for gate fees.

If instead all the FW was sent to AD, this would lead to the greatest annual LCC savings of £251 m compared to the current situation. This is followed by IVC with a saving of £172 m/year. AD would also lead to the lowest costs of £334 m/year for local authorities, a 42% reduction on the current costs of £579 m/year. Landfilling is the most expensive option overall (£604 m/year), followed by incineration (£349 m/year). This is in contrast with the trends seen for treatment routes with the current collections rates (see Fig. 8): if all FW can be collected either separately or co-mingled with garden waste, the collection costs fall sufficiently for both AD and IVC to have lower LCC than incineration.

Considering the avoidable FW, the results shows that the avoidable FW fraction collected by local authorities (2.9 Mt/year) has an annual shelf value of £9.71 bn and the potentially avoidable fraction (0.8 Mt/year) a value of £1 bn. No value is assigned to the unavoidable fraction. Therefore, the total LCC of managing 4.9 Mt/year of household FW are approximately 4% of the £10.71 bn shelf value of the avoidable and potentially avoidable FW. Hence, avoiding waste would achieve far greater cost savings (43 times) than the maximum saving potential of £251 m/year that could be attained by treating all the FW via AD.

3.3.3. Overall environmental and economic sustainability

Following the same approach as for the overall sustainability evaluation of different treatment methods per tonne of FW treated (Section 3.2.3), this section considers their sustainability in comparison to the current waste management situation based on the total amount

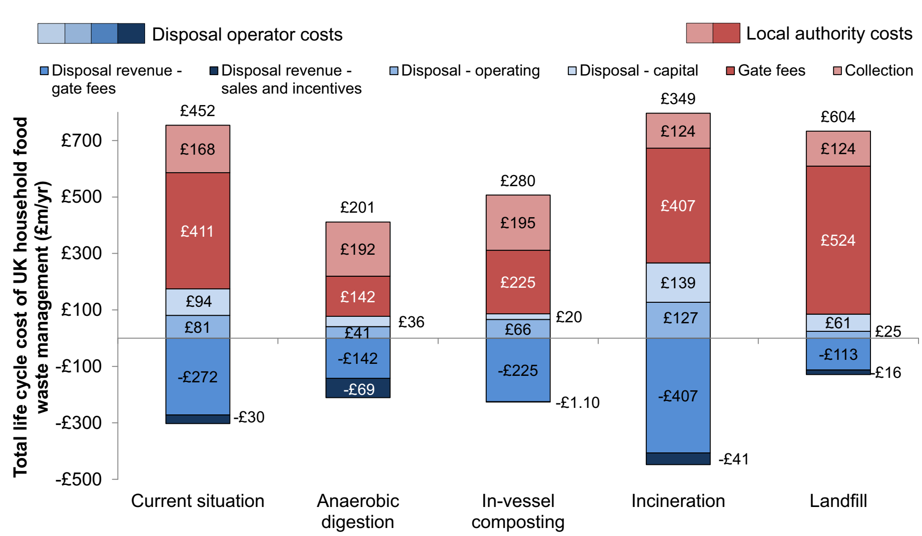


Fig. 11. Life cycle costs of the treatment of collected household food waste for different treatment options for the total annual collection of 4.9 Mt.

of FW treated annually. Contrary to the findings per tonne of FW treated, AD emerges as the best treatment option, while incineration is now ranked the second; IVC and landfill keep their previous rankings (Fig. 9b). The reason for this change in rankings is the LCC: while the LCA impacts exhibit the same trends as the per-tonne rankings shown in Fig. 9a, the LCC of AD and IVC are reduced significantly (by 64% and 35%) with higher volumes of waste they treat due to the economies of scale. However, it should be stressed that this increase in the volume of waste treated should only be at the expense of the volume treated by incineration and landfilling, rather than an absolute increase in the amount of FW.

It can also be seen from Fig. 9b that all options but landfilling are more sustainable than the current mix of treatment options, which is ranked fourth.

Therefore, the findings of this paper support UK policy proposals to mandate separate food collections from all households by 2023 (DEFRA, 2018b) and treat that FW via anaerobic digestion, as recommended by the National Infrastructure Commission (2018). However, as discussed earlier, much greater benefits would be achieved through FW prevention.

4. Conclusion and recommendations

This work has developed environmental and economic life cycle models for the treatment of FW via anaerobic digestion, in-vessel composting, incineration and landfill to provide new insights into the sustainability of FW management. The comparison of these treatment methods shows that, on average, anaerobic digestion has the lowest impacts for 13 out of 19 categories, including global warming potential. The latter is estimated to be net-negative at −32 kg CO_2 eq./t FW. Incineration also has a net-negative GWP (−5 kg CO_2 eq./t FW), along with the lowest marine eutrophication and agricultural land occupation, but it has the highest metal depletion, freshwater eutrophication and particulate matter formation. Landfill has the greatest global warming potential (195 kg CO_2 eq./t FW), human toxicity, marine eutrophication and urban land occupation but is the best option for terrestrial acidification, particulate matter and natural land transformation. In-vessel composting is overall the worst option, with 12 impacts being higher than for any other method. Therefore, although this option complies with the circular economy principles through the recycling of nutrients back to the start of the food supply chain, it is otherwise environmentally unsustainable. On the other hand, anaerobic digestion is both compliant with the circular economy principles and is also environmentally the most sustainable option.

At current food waste collection rates in the UK, incineration has the lowest life cycle costs of £71/t FW, attributable to the revenue from electricity generation and the lowest collection costs. It also offers the operators the highest profits in spite of the highest operating and capital costs; this is primarily due to a high gate fee. Despite making negligible revenue from compost, in-vessel composting has the second lowest LCC (£80/t FW), largely due to the lowest capital costs. Anaerobic digestion has the second highest overall costs (£110/t FW), attributable to the high collection costs per tonne of the relatively small share of FW currently collected separately. Without the landfill tax, landfilling has the lowest LCC of £39/t FW, but with the tax, it becomes the most expensive treatment method at £123/t FW.

Treating the total of 4.9 Mt of food waste collected annually from households by local authorities costs £452 m and generates 0.34 Mt CO_2 eq. However, the treatment system is a net provider of electricity, saving 1.9 PJ of primary energy. Despite that, it is a net contributor across 12 of the 17 other environmental impacts considered, including acidification, eutrophication, summer smog, ozone depletion and particulate matter.

Environmentally, the best option would be to treat all the food waste via anaerobic digestion, which would provide savings across 15 of the 19 impacts compared to the current situation, including primary energy demand (7.55 PJ/year) and global warming potential (0.49 Mt CO_2 eq./year). This would also lead to a cost saving of £251 m/year compared to the current situation. However, the environmental benefits of AD are primarily dependent on displacing the current grid electricity. With the increasing share of renewables in the mix, the relative advantage of anaerobic digestion would decrease and in-vessel composting could become more competitive.

Assuming equal importance of environmental impacts and costs, incineration is the most sustainable option per tonne of waste treated. For the annual volume currently treated in the UK, anaerobic digestion would be more sustainable than incineration; in-vessel composting is ranked third in both cases. However, they are all more sustainable than the current mix of treatment technologies. Landfilling is the least sustainable option overall.

Based on these results, the separate collection of household food waste should be encouraged if it can be treated via anaerobic digestion. However, if in-vessel composting remains the primary treatment route for separate collections, it would be more sustainable for the food waste to be incinerated with general MSW. Therefore, the most realistic scenario for improving the sustainability of household food waste is to continue to favour incineration over landfill for MSW, while increasing and promoting separate food waste collections for treatment via anaerobic digestion only. However, preventing avoidable food waste would achieve far greater benefits, both economically and environmentally, and should be favoured over any treatment option.

Acknowledgements

This work was carried out as part of the UK Centre for Sustainable Energy Use in Food Chains (CSEF), funded by the Research Councils UK (EP/K011820/1). The authors acknowledge gratefully this funding.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2019.07.322.

References

ADAS (Ed.), 2013. MANNER-NPK. v1.0.1 ed.: ADAS with AHDB, CSF, DARD, Defra. Environment Agency, Natural England, Scottish Government, Tried and Tested, and WRAP.

Amec Foster Wheeler, 2015. Addendum to Energy from Waste Business Case [Online]. Produced by Amec FW for Aberdeen City Council.Available. committees.aberdeencity.gov.uk/documents/s61677/Appendix%202%202015%20Energy%20from%20Waste%20Business%20Case%20Review.pdf (Accessed August 2018).

Amlinger, F., Peyr, S., Cuhls, C., 2008. Green house gas emissions from composting and mechanical biological treatment. Waste Manag. Res. 26 (1), 47–60.

Baky, A., Eriksson, O., 2003. Systems Analysis of Organic Waste Management in Denmark. Danish Ministry of the Environment, Danish Environmental Protection Agency.

Bank of England, 2018. Inflation calculator.[Online]. Available:. www.bankofengland.co.uk/monetary-policy/inflation (Accessed August 2018).

Banks, C.J., Chesshire, M., Heaven, S., Arnold, R., 2011. Anaerobic digestion of source-segregated domestic food waste: performance assessment by mass and energy balance. Bioresour. Technol. 102 (2), 612–620.

BEIS, 2017a. 2016 UK greenhouse gas emissions: provisional figures - data tables [Online]. Department for Business, Energy & Industrial Strategy Available:. www.gov.uk/government/statistics/provisional-uk-greenhouse-gas-emissions-national-statistics-2016, Accessed date: August 2018.

BEIS, 2017b. Periodic review of the feed-in tariffs 2015 phase 2: anaerobic digestion and micro combined heat and power. [Online]. Available:. Department for Business Energy & Industrial Strategywww.gov.uk/government/uploads/system/uploads/attachment_data/file/590331/FITs_AD_mCHP_impact_assessment.pdf, Accessed date: August 2018.

BEIS, 2017c. Prices of fuels purchased by non-domestic consumers in the United Kingdom excluding/including CCL (QEP 3.4.1 and 3.4.2) [Online].Available:. Department for Business, Energy & Industrial Strategy (BEIS)www.gov.uk/government/statistical-data-sets/gas-and-electricity-prices-in-the-non-domestic-sector (Accessed August 2018).

BEIS, 2018. Energy Trends March 2018 [Online].Available:. Department for Business Energy & Industrial Strategy (BEIS) Accessed May 2019. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/712458/Energy_Trends_March_2018.pdf.

Beretta, C., Hellweg, S., 2019. Potential environmental benefits from food waste prevention in the food service sector. Resour. Conserv. Recycl. 147, 169–178.

Bernstad, A., la Cour Jansen, J., 2011. A life cycle approach to the management of household food waste – a Swedish full-scale case study. Waste Manag. 31 (8), 1879–1896.

Bernstad, A., la Cour Jansen, J., 2012. Review of comparative LCAs of food waste management systems – current status and potential improvements. Waste Manag. 32 (12), 2439–2455.

Bhogal, A., Taylor, M., Nicholson, F., Rollett, A., Williams, J., Newell Price, P., Chambers (ADAS), B., Litterick (Earthcare), A. & Whittingham (Newcastle University), M. (2016). Field Experiments for Quality Digestate and Compost in Agriculture: Work Package 1 Report – Effect of Repeated Digestate and Compost Applications on Soil and Crop Quality. WRAP Project Code: OMK001-001/WR1212.

Breitenbeck, G.A., Schellinger, D., 2004. Calculating the reduction in material mass and volume during composting. Compost Sci. Util. 12 (4), 365–371.

Burns, C., Hopwood, L., More, O., Hindle, M., 2017. Organics recycling industry status report 2015. [Online]. Available:. WRAPwww.wrap.org.uk/sites/files/wrap/asori%202015.pdf, Accessed date: August 2018.

Cabaraban, M.T.I., Khire, M.V., Alocilja, E.C., 2008. Aerobic in-vessel composting versus bioreactor landfilling using life cycle inventory models. Clean Techn. Environ. Policy 10 (1), 39–52.

Cadena, E., Colón, J., Sánchez, A., Font, X., Artola, A., 2009. A methodology to determine gaseous emissions in a composting plant. Waste Manag. 29 (11), 2799–2807.

Carlsson Reich, M., 2005. Economic assessment of municipal waste management systems—case studies using a combination of life cycle assessment (LCA) and life cycle costing (LCC). J. Clean. Prod. 13 (3), 253–263.

DAERA-NI, 2017. Annual NILAS reports [Online].Available:. Department of Agriculture, Environment and Rural Affairswww.daera-ni.gov.uk/articles/published-waste-data (Accessed November 2018).

De Klein, C., Novoa, R., Ogle, S., Smith, K., Rochette, P., Wirth, T., McConkey, B., Mosier, A., Rypdal, K., 2006. IPCC guidelines for national greenhouse gas inventories, volume 4, chapter 11: N2O emissions from managed soils, and CO2 emissions from lime and urea application. Technical Report 4-88788-032-4. Intergovernmental Panel on Climate Change.

DEFRA. (2004). Defra Guidance for Successful Reclamation of Mineral and Waste Sites [Online]. Department for Environment, Food & Rural Affairs. Available: webarchive.nationalarchives.gov.uk/20090318074725/http://www.defra.gov.uk/farm/environment/land-use/reclamation/guidance-full.pdf [(Accessed August 2018)].

DEFRA, 2006a. NT26 Research Programme. [Online]. Available:. Department for Environment, Food & Rural Affairsrandd.defra.gov.uk/Document.aspx?Document=NT2605_4075_EXE.doc, Accessed date: August 2018.

DEFRA, 2006b. WP4 nitrogen losses to surface and ground waters. [Online]. Available:. Department for Environment, Agriculture & Rural Affairsrandd.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&Completed=0&ProjectID=11983, Accessed date: December 2017.

DEFRA, 2017a. ENV18 - local authority collected waste: annual results tables [Online]. Available:. Department for Environment, Food and Rural Affairswww.gov.uk/government/statistical-data-sets/env18-local-authority-collected-waste-annual-results-tables (Accessed August 2018).

DEFRA, 2017b. Family food datasets [Online]. Department for Environment, Food & Rural Affairs Available. www.gov.uk/government/statistical-data-sets/family-food-datasets, Accessed date: August 2018.

DEFRA, 2018a. Food and drink waste hierarchy: deal with surplus and waste. [Online]. Available:. Department for Environment, Food & Rural Affairs (DEFRA)www.gov.uk/government/publications/food-and-drink-waste-hierarchy-deal-with-surplus-and-waste/food-and-drink-waste-hierarchy-deal-with-surplus-and-waste, Accessed date: June 2019.

DEFRA, 2018b. Our waste, our resources: a strategy for England [Online]. Department for Environment, Agriculture & Rural Affairs Available. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/765914/resources-waste-strategy-dec-2018.pdf, Accessed date: December 2018.

Doka, G., 2009. Life cycle inventories of waste treatment services. Ecoinvent Report No. 13. Swiss Centre for Life Cycle Inventories (Dubendorf).

Eades, P., Banks, C., Heaven, S., Walker, M., 2011. Mass and energy balance for a rotating-drum composting plant. Proc. Inst. Civ. Eng. Waste Resour. Manag. 164 (3), 151–159.

EC, 1999. Council Directive 1999/31/EC of 26 April 1999 on the landfill of waste. L. Official Journal L 182, 0001–0019 16/07/1999. (European Commission).

EC, 2014. Communication From the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions A Policy Framework for Climate and Energy in the Period From 2020 to 2030. European Commission.

EC, 2015. Closing the Loop-An EU Action Plan for the Circular Economy. Communication From the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions, COM(2015), 614 (2). European Commission (EC), Brussels.

Ecoinvent, 2008. Calculation Tool for Waste Disposal in Municipal Sanitary Waste Landfill and Incineration for Ecoinvent v2.1. Ecoinvent Centre, Switzerland.

Ecoinvent, 2016. Ecoinvent Database v3.3. Zurich and Lausanne, Switzerland.

EMF, 2015. Delivering the Circular Economy - A Toolkit for Policy Makers. Ellen MacArthur Foundation (EMF), Isle of Wight.

Environment Agency, 2009. How to comply with your environmental permit. Additional guidance for: landfill (EPR 5.02) [Online]. Environment Agency. Available. www.gov.uk/government/uploads/system/uploads/attachment_data/file/296861/geho0409bput-e-e.pdf, Accessed date: August 2018.

Environment Agency, 2016. Municipal waste incinerator annual reports for 2016 [Online]. Environment Agency. Available:. ukwin.org.uk/resources/incinerator-reports, Accessed date: July 2018.

Esteves, S., Devlin, D., 2010. Food waste chemical analysis [Online]. WRAP. Available. www.wrapcymru.org.uk/sites/files/wrap/Technical_report_food_waste_characterisation_Wales_2009x2.9086.pdf, Accessed date: July 2018.

Eurostat. (2018). Gas prices for household consumers - bi-annual data (from 2007 onwards) [Online]. European Commission. Available - appsso.eurostat.ec.europa.eu/nui/submitViewTableAction.do [Accessed August 2018].

Evangelisti, S., Lettieri, P., Borello, D., Clift, R., 2014. Life cycle assessment of energy from waste via anaerobic digestion: a UK case study. Waste Manag. 34 (1), 226–237.

FAO, 2014. Food wastage footprint - full-cost accounting - final report. http://www.fao.org/3/a-i3991e.pdf: Food and Agriculture Organisation of the United Nations - FAO.

FAO, 2015. Food wastage footprint & climate change. www.fao.org/3/a-bb144e.pdf: Food and Agriculture Organisation of the United Nations - FAO.

Favoino, E., Hogg, D., 2008. The potential role of compost in reducing greenhouse gases. Waste Manag. Res. 26 (1), 61–69.

Fisher, K., 2006. Impact of energy from waste and recycling policy on UK greenhouse gas emissions–final report [online]. Prepared by Environment Resource Management (ERM) for Department for Environment. Food & Rural Affairs (DEFRA) Available:. randd.defra.gov.uk/Document.aspx?Document=WR0609_5737_FRP.pdf (Accessed August 2018).

Fletcher, L., Jones, N., Warren, L., Stentiford, E., 2014. Understanding Biofilter Performance and Determining Emission Concentrations Under Operational Conditions. The University of Leeds; Scotland and Northern Ireland Forum For Environmental Research (SNIFFER).

Goedkoop, M., Heijungs, R., Huijbregts, M., De Schryver, A., Struijs, J., van Zelm, R., 2013. ReCiPe 2008. A Life Cycle Impact Assessment Method Which Comprises Harmonised Category Indicators at the Midpoint and the Endpoint Level, First Edition (revised). PRé Consultants CML, University of Leiden RUN, Radboud University Nijmegen, RIVM.

Green Investment Group, 2016. GIB commits capital to Edinburgh energy-from-waste plant. [Online]. Available. greeninvestmentgroup.com/news-and-insights/2016/gib-commits-capital-to-edinburgh-energy-from-waste-plant, Accessed date: July 2018.

Gregory, R., Stalleicken, J., Lane, R., Arnold, S., Hall, D., 2014. WR1908 review of methane emissions modelling [Online]. Department for Environment, Agriculture & Rural Affairs Available:. randd.defra.gov.uk/Document.aspx?Document=12439_WR1908ReviewofMethaneEmissionsModelling.pdf, Accessed date: September 2018.

Guinée, J.B., 2002. Handbook on life cycle assessment operational guide to the ISO standards. Int. J. Life Cycle Assess. 7 (5), 311–313.

Gustavsson, J., Cederberg, C., Sonesson, U., Van Otterdijk, R., Meybeck, A., 2011. Global food losses and food waste. www.fao.org/docrep/014/mb060e/mb060e00.pdf: The Food and Agriculture Organization of the United Nations.

Harty, M.A., Forrestal, P.J., Watson, C.J., McGeough, K.L., Carolan, R., Elliot, C., Krol, D., Laughlin, R.J., Richards, K.G., Lanigan, G.J., 2016. Reducing nitrous oxide emissions by changing N fertiliser use from calcium ammonium nitrate (CAN) to urea based formulations. Sci. Total Environ. 563–564, 576–586.

HMRC, 2017. Landfill Tax rates [Online].Available:. HM Revenue & Customs, UK Governmentwww.gov.uk/government/publications/rates-and-allowances-landfill-tax/landfill-tax-rates-from-1-april-2013 (Accessed August 2018).

Hogg, D. (2002). Costs for municipal waste management in the EU [Online]. Eunomia Research & Consulting for the European Commission. Available: ec.europa.eu/environment/waste/studies/pdf/eucostwaste.pdf [Accessed August 2018].

Horne, J., Scholes, P., Areikin, E., Brown, B., 2013. A survey of the UK organics recycling industry in 2012 [Online]. WRAP. Available. www.wrap.org.uk/sites/files/wrap/ASORI%202012.pdf, Accessed date: August 2018.

Hunkeler, D., Lichtenvort, K., Rebitzer, G., 2008. Environmental Life Cycle Costing. CRC Press.

ICIS, 2017. World sulphuric acid weekly [Online]. ICIS. Available:. www.icis.com/globalassets/global/icis/pdfs/sample-reports/fertilizers-sulphuric-acid.pdf, Accessed date: September 2018.

ISO, 2006a. ISO 14040-Environmental Management—Life Cycle Assessment—Principles and Framework (Geneva).

ISO, 2006b. ISO 14044-Environmental Management—Life Cycle Assessment—Requirements and Guidelines (Geneva).

Jeswani, H.K., Azapagic, A., 2016. Assessing the environmental sustainability of energy recovery from municipal solid waste in the UK. Waste Manag. 50, 346–363.

Kim, M.-H., Kim, J.-W., 2010. Comparison through a LCA evaluation analysis of food waste disposal options from the perspective of global warming and resource recovery. Sci. Total Environ. 408 (19), 3998–4006.

Kim, M.-H., Song, Y.-E., Song, H.-B., Kim, J.-W., Hwang, S.-J., 2011. Evaluation of food waste disposal options by LCC analysis from the perspective of global warming: Jungnang case, South Korea. Waste Manag. 31 (9–10), 2112–2120.

Lee, S.-H., Choi, K.-I., Osako, M., Dong, J.-I., 2007. Evaluation of environmental burdens caused by changes of food waste management systems in Seoul, Korea. Sci. Total Environ. 387 (1), 42–53.

Leinemann, B., 1998. The Formation of Greenhouse Gases During Composting as a Result of Process Management and Input Materials. Dissertation an der Gemeinsamen Naturwissenschaftlichen Fakultät der Technischen Universität Braunschweig.

Letsrecycle, 2006. New 8m composting plant opens in Derbyshire. [Online]. Available:. www.letsrecycle.com/news/latest-news/new-8m-composting-plant-opens-in-derbyshire/, Accessed date: September 2018.

Letsrecycle, 2009. TEG buys 3.1m Norfolk composting facility [Online].Available:. www.letsrecycle.com/news/latest-news/teg-buys-3-1m-norfolk-composting-facility/ (Accessed September 2018).

Letsrecycle, 2010. Agrivert opens 5m Oxfordshire IVC facility [Online]. Available. www.letsrecycle.com/news/latest-news/agrivert-opens-5m-oxfordshire-ivc-facility/, Accessed date: September 2018.

Letsrecycle, 2015. Biowise opens £4.5 million IVC near Hull. [Online]. Available. www.letsrecycle.com/news/latest-news/biowise-opens-4-5-million-ivc-near-hull/, Accessed date: September 2018.
Letsrecycle, 2018. Biowise awarded Cheshire East organics contract [Online]. Available. www.letsrecycle.com/news/latest-news/biowise-cheshire-east-contract/, Accessed date: September 2018.
Levis, J.W., Barlaz, M.A., 2011. What is the most environmentally beneficial way to treat commercial food waste? Environ. Sci. Technol. 45 (17), 7438–7444.
Levis, J.W., Barlaz, M.A., 2013. Composting process model documentation (Retrieved May, 16, 2015).
LoRe, A.M., Harder, S., 2012. The Effect of Food Waste Diversion on Waste Heating Value and WTE Capacity. 20th Annual North American Waste-to-Energy Conference, 2012. American Society of Mechanical Engineers, pp. 89–99.
Martínez-Blanco, J., Muñoz, P., Antón, A., Rieradevall, J., 2009. Life cycle assessment of the use of compost from municipal organic waste for fertilization of tomato crops. Resour. Conserv. Recycl. 53 (6), 340–351.
Martinez-Sanchez, V., Kromann, M.A., Astrup, T.F., 2015. Life cycle costing of waste management systems: overview, calculation principles and case studies. Waste Manag. 36, 343–355.
Martinez-Sanchez, V., Tonini, D., Møller, F., Astrup, T.F., 2016. Life-cycle costing of food waste management in Denmark: importance of indirect effects. Environ. Sci. Technol. 50 (8), 4513–4523.
Møller, J., Boldrin, A., Christensen, T.H., 2009. Anaerobic digestion and digestate use: accounting of greenhouse gases and global warming contribution. Waste Manag. Res. 27 (8), 813–824.
Monson, K., Esteves, S., Guwy, A., Dinsdale, R., 2007. In: University of Glamorgan ISBN (Ed.), Anaerobic Digestion of Biodegradable Municipal Wastes: A Review, p. 978-1.
MRWA, 2010. Flagship waste processor gets the green light. [Online]. Available:. Merseyside Recycling & Waste Authoritywww.merseysidewda.gov.uk/2010/01/flagship-waste-processor-gets-the-green-light/, Accessed date: August 2018.
Naroznova, I., Møller, J., Scheutz, C., 2016. Global warming potential of material fractions occurring in source-separated organic household waste treated by anaerobic digestion or incineration under different framework conditions. Waste Manag. 58, 397–407.
National Infrastructure Commission. (2018). National Infrastructure Assessment [Online]. UK Government. Available: www.nic.org.uk/wp-content/uploads/CCS001_CCS0618917350-001_NIC-NIA_Accessible.pdf#page=33 [(Accessed July 2019)].
Nicholson, F.A., Bhogal, A., Chadwick, D., Gill, E., Gooday, R.D., Lord, E., Misselbrook, T., Rollett, A.J., Sagoo, E., Smith, K.A., Thorman, R.E., Williams, J.R., Chambers, B.J., 2013. An enhanced software tool to support better use of manure nutrients: MANNER-NPK. Soil Use Manag. 29 (4), 473–484.
Nicholson, F., Taylor, M., Bhogal, A., Rollett, A., Williams, J., Newell Price, P., Chambers (ADAS), B., Becvar, A., Wood, M., Litterick (Earthcare), A., Crooks, B., Knox, O., Walker (SRUC), R., Misselbrook, T., Cardenass, L., Chadwick (North Wyke Research), D., Lewis (Harper Adams), P. & Mark, E. E. M. (2016). Field Experiments for Quality Digestate and Compost in Agriculture: Work Package 2 Report – Digestate Nitrogen Supply and Environmental Emissions. WRAP Project Code: OMK001-001/WR1212. WRAP.
Nicholson, F., Bhogal, A., Cardenas, L., Chadwick, D., Misselbrook, T., Rollett, A., Taylor, M., Thorman, R., Williams, J., 2017. Nitrogen losses to the environment following food-based digestate and compost applications to agricultural land. Environ. Pollut. 228, 504–516.
Ofgem, 2018. Feed-In Tariff (FIT) rates [Online]. Office of Gas and Electricity Markets, UK Government. Available. www.ofgem.gov.uk/environmental-programmes/fit/fit-tariff-rates, Accessed date: August 2018.
Oldfield, T.L., White, E., Holden, N.M., 2016. An environmental analysis of options for utilising wasted food and food residue. J. Environ. Manag. 183 (Part 3), 826–835.
Quested, T., Parry, A., 2017. Household food waste in the UK, 2015. [Online]. Available. WRAPwww.wrap.org.uk/sites/files/wrap/Household_food_waste_in_the_UK_2015_Report.pdf, Accessed date: August 2017.
Quested, T., Easteal, S., Ingle, R., 2013a. Methods used for household food and drink waste in the UK 2012 [Online]. WRAP. Available. www.wrap.org.uk/sites/files/wrap/Methods%20Annex%20Report%20v2.pdf, Accessed date: April 2019.
Quested, T., Ingle, R., Parry, A., 2013b. Household Food and Drink Waste in the United Kingdom 2012. WRAP, London.
Rigby, H., Smith, S., 2011. New markets for digestate from anaerobic digestion. WRAP Report: ISS001-001.
Roberts, D., 2015. Characterisation of chemical composition and energy content of green waste and municipal solid waste from Greater Brisbane, Australia. Waste Management 41, 12–19.
Ryan, R., 2013. Harnessing the power. [Online]. Available:. Viridorwww.aston.ac.uk/EasysiteWeb/getresource.axd?AssetID=148030&type=Full&servicetype=Attachment, Accessed date: August 2018.
S&P Global Platts. (2016). Solventswire [Online]. S&P Global Platts,. Available: www.platts.com/IM.Platts.Content/ProductsServices/Products/solventswire.pdf [Accessed August 2018].
Scholes, P., Areikin, E., 2014. A survey of the UK anaerobic digestion industry in 2013 [Online]. WRAP. Available. www.wrap.org.uk/content/survey-uk-anaerobic-digestion-industry-2013, Accessed date: July 2018.
SEPA, 2017. Household waste – summary data 2016. [Online]. Available:. Scottish Environment Protection Agencywww.sepa.org.uk/media/320744/household-waste-summary-data-and-commentary-2016.pdf, Accessed date: July 2018.
Slorach, P.C., Jeswani, H.K., Cuéllar-Franca, R., Azapagic, A., 2019. Environmental sustainability of anaerobic digestion of household food waste. J. Environ. Manag. 236, 798–814.
Smith, P., Powlson, D., Glendining, M., Smith, J.O., 1997. Potential for carbon sequestration in European soils: preliminary estimates for five scenarios using results from long-term experiments. Glob. Chang. Biol. 3 (1), 67–79.
Stamford, L., Azapagic, A., 2012. Life cycle sustainability assessment of electricity options for the UK. Int. J. Energy Res. 36 (14), 1263–1290.
Statista, 2018. Annual average price of red diesel in the United Kingdom (UK) from 2003 to 2016* (British pence per liter) [Online]. Scottish Government. Available. www.statista.com/statistics/527997/annual-average-price-of-red-diesel-in-the-united-kingdom-uk/, Accessed date: August 2018.
StatsWales. (2017). Waste managed (tonnes) by management method and year [Online]. Welsh Goverment. Available: statswales.gov.wales/Catalogue/Environment-and-Countryside/Waste-Management/Local-Authority-Municipal-Waste/wastemanaged-by-management-year [Accessed August 2018].
Stenmarck, A.s., Jensen, C., Quested, T., Moates, G., Buksti, M., Cseh, B.z., Juul, S., Parry, A., Politano, A., Redlingshofer, B., 2016. Estimates of European Food Waste Levels. FUSIONS - European Commission.
Swarr, T.E., Hunkeler, D., Klöpffer, W., Pesonen, H.-L., Ciroth, A., Brent, A.C., Pagan, R., 2011. Environmental life-cycle costing: a code of practice. Int. J. Life Cycle Assess. 16 (5), 389–391.
The Andersons Centre, 2010. A detailed economic assessment of anaerobic digestion technology and its suitability to UK farming and waste systems [Online]. Funded by DECC and managed by the NNFCC. Available:. www.organics-recycling.org.uk/uploads/category1060/10-010%20FINAL_Andersons_NNFCC_AD2010.pdf, Accessed date: August 2018.
The World Bank, 2016. Total greenhouse gas emissions (kt of CO2 equivalent). [Online]. Available:. data.worldbank.org/indicator/EN.ATM.GHGT.KT.CE, Accessed date: July 2018.
Thinkstep, 2017. GaBi Software-System and Database for Life Cycle Engineering (Leinfelden-Echterdingen).
Towler, G., Sinnott, R., 2013. Chapter 8 - estimating revenues and production costs. Chemical Engineering Design. Second edition. Butterworth-Heinemann, Boston.
Turner, D.A., Williams, I.D., Kemp, S., 2016. Combined material flow analysis and life cycle assessment as a support tool for solid waste management decision making. J. Clean. Prod. 129, 234–248.
UK Government, 2014. Using animal by-products at compost and biogas sites [Online]. Available. www.gov.uk/guidance/using-animal-by-products-at-compost-and-biogas-sites, Accessed date: July 2018.
UK Government, 2019. UK becomes first major economy to pass net zero emissions law. [Online]. Available:. UK Governmentwww.gov.uk/government/news/uk-becomes-first-major-economy-to-pass-net-zero-emissions-law, Accessed date: July 2019.
UN, 2015. Resolution adopted by the General Assembly on 25 September 2015. Transforming our world: the 2030 Agenda for Sustainable Development [Online]. Available. sustainabledevelopment.un.org/post2015/transformingourworld, Accessed date: August 2018.
WasteDataFlow, 2018. . Available:. wastedataflow.org/, Accessed date: July 2018.
WRAP, 2008. ICP2 – online tool modelling assumptions technical annex [Online]. WRAP. Available:. laportal.wrap.org.uk/Documents/ICP%20online%20tool%20assumptions.pdf, Accessed date: August 2018.
WRAP, 2016a. Household food waste collections guide [Online]. Waste & Resources Action Programme (WRAP). Available:. www.wrap.org.uk/sites/files/wrap/HH_food_waste_collections_guide_section_1_context_%20and_background.pdf, Accessed date: August 2018.
WRAP, 2016b. In vessel composting (IVC) [Online]. Available:. http://www.wrap.org.uk/content/vessel-composting-ivc, Accessed date: May 2019.
WRAP, 2017a. Gate fees report 2017 [Online]. Waste & Resources Action Programme (WRAP). Available:. www.wrap.org.uk/sites/files/wrap/Gate%20Fees%20report%202017_FINAL_clean.pdf, Accessed date: August 2018.
WRAP, 2017b. What is Courtauld 2025? [Online]. Waste & Resources Action Programme (WRAP). Available:. www.wrap.org.uk/content/what-courtauld-2025, Accessed date: August 2018.
WRAP, 2018. Kerbside Costing Tool [Online]. Waste & Resources Action Programme (WRAP). Available. laportal.wrap.org.uk/ICPToolHome.aspx, Accessed date: August 2018.