

Research article

Environmental sustainability of anaerobic digestion of household food waste

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ABSTRACT

Consumers are the leading producers of food waste (FW) in developed countries and the majority of household FW is still embedded in general waste where it is incinerated or landfilled. There is increasing awareness in the value of collecting FW as a separate waste stream for production of compost or recovery of energy through anaerobic digestion (AD). This study focuses on AD to evaluate the life cycle environmental sustainability of recovering energy and fertilisers from household FW in the UK. The analysis is carried out for two different functional units: i) treatment of 1 tonne of FW, which is compared to incineration and landfilling; and ii) generation of 1 MWh of electricity, which is compared to other electricity generation options. The former results in net negative greenhouse gas (GHG) emissions ($-39 \text{ kg CO}_2\text{-eq./t}$) and primary energy demand (-2 GJ/t) due to the displacement of grid electricity and mineral fertilisers. AD has lower impacts than both incineration and landfilling across 15 of the 19 impacts. However, the application of digestate to land and the release of ammonia and nitrates lead to higher marine eutrophication (ME), terrestrial acidification (TA) and particulate matter formation (PMF). For the second functional unit, AD electricity emits $203 \text{ kg CO}_2\text{-eq./MWh}$, compared to $357 \text{ kg CO}_2\text{-eq./MWh}$ for the UK grid mix. Compared to renewables, such as wind and solar, AD electricity has lower energy demand, toxicity potentials and metal depletion. However, it has higher global warming potential, ME, TA and PMF. At the UK level, treating 4.9 Mt of kerbside FW collected annually could provide 0.37% of the national electricity demand and save $190,000 \text{ t CO}_2\text{-eq./yr}$ compared to the grid electricity. The digestate produced could displace 1% of industrial nitrogen fertilisers. Although small fractions of the national demands, they represent a valuable return from a largely unutilised waste stream and help towards implementation of a circular economy.

1. Introduction

Food waste remains an inevitable product of a modern society. Although there are no definitive figures for global food waste, it has been reported that approximately a third of food intended for human consumption is lost or wasted through the food supply chain (Gustavsson et al., 2011); this quantity has a production value of \$750 billion (FAO, 2013) and accounts for 6.8% of global annual greenhouse gas (GHG) emissions (FAO, 2015; The World Bank, 2016). Embedded in this waste are the huge areas of land, volumes of water and quantities of fertiliser used to produce food that will never be eaten. This is occurring concurrently with 821 million people living without adequate nourishment (FAO, 2018) and with a drive to increase food supply to meet the demands of a growing population (Tilman et al., 2011).

The United Nations have set a target to halve the per-capita food waste at the retail and consumer levels by 2030 (UN, 2015). The prevention of food waste is the best means to eliminate its burden (Papargyropoulou et al., 2014) as demonstrated by several life cycle

assessments studies (e.g. Oldfield et al., 2016; Eriksson and Spangberg, 2017). However, the production of food wastes across the supply chain is inevitable and it is desirable to reclaim value from this waste stream. There is no universal definition of food waste because different organisations disagree on the supply chain boundaries considered and on the type of foodstuffs included. For example, according to the Food and Agriculture Organization of the United Nations (FAO), food losses refer to the decrease in edible food for human consumption up to the point of the final product and food waste is the loss of edible food during retail and consumption (Parfitt et al., 2010). On the other hand, a food-waste project funded by the European Commission expands this definition to also include the inedible parts of food and the entire food supply chain (Östergren et al., 2014). Whilst monitoring the loss of edible food waste is important in achieving waste reduction targets, there is no meaningful difference between the edible and inedible components when utilising it as a valuable waste stream. In this study, the term “household food” waste encompasses both edible and inedible components.

Food waste is generated across the supply chain, including crops

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rejected at harvest, processing and manufacturing losses, unsold food at retail and disposals by the consumer. Developed countries tend to have major losses related to the consumption stage, whereas developing nations have significant post-harvest losses associated with poor storage and access to refrigeration (Parfitt et al., 2010). North America and Oceania have the highest consumer food waste in the world (Gustavsson et al., 2011), with the US generating 99 kg per person annually (EPA, 2017). Within the EU, more than half of all food waste is produced in households by the consumer, with the average EU citizen disposing of 71 kg of food waste per year in their municipal waste (Stenmarck et al., 2016). Europe's most populous country, Germany, produces 5.1 Mt of household food waste per year or 63 kg per person per year (ISWA University of Stuttgart, 2012). Austria has the lowest recorded waste per person in Europe: 44 kg per year (Schneider et al., 2012). The UK produces an estimated 4.9 Mt of locally collected household food waste per year, equivalent to 73 kg per person; a further 1.6 Mt is disposed to the sewer and 0.8 Mt home-composted or fed to domestic animals (Quested and Parry, 2017).

The majority of food waste is collected alongside general municipal waste and sent to landfill or incineration; however, there is increasing awareness of the environmental damage this can cause. In 2005, Sweden introduced a landfill ban on organic waste (EEA, 2013b) and Germany has a ban on landfilling any un-pretreated organic wastes (EEA, 2013a). Waste disposed of in landfills generate methane and, although many sites employ landfill gas capture, its efficiency is estimated at 68% (Gregory et al., 2014). The incineration of waste prevents the release of methane to the atmosphere and allows for the recovery of heat to generate electricity. However, the high moisture content of household food waste results in a low heating value and even a negative lower heating value (LHV) in the case of some fruits and vegetables (Eriksson et al., 2015). Neither landfill nor incineration allow for the recovery of nutrients from food waste, which could be used to facilitate the cultivation of new crops. Thus, there is an apparent need to replace incineration and landfilling with other treatment methods that enable sustainable harnessing of the value of waste food streams.

The most common treatment methods currently used in the UK for food waste collected separately from other waste are composting and anaerobic digestion (AD) (WasteDataFlow, 2018). Both of these routes comply with the biological principles of circular economy through the production of compost and digestate, respectively, which can be used to return nutrients to agricultural land (Ellen MacArthur Foundation, 2013). In addition, AD generates biogas which can be used to recover energy. In the UK, about 10% of household food waste is currently collected separately or alongside garden waste (WasteDataFlow, 2018) and is managed via centralised composting or AD. Presently, 44% of UK households are provided with a separate collection for food waste and the recommendation by the government-supported Waste and Resources Action Plan (WRAP) is to increase the provision to all households (UK Parliament, 2017). However, the environmental impacts and benefits of treating UK household food waste via either composting or AD must be fully understood prior to their wider deployment. This paper focuses on treating food waste via AD as a waste disposal route and an electricity generation option.

AD decomposes organic matter into biogas and digestate in the absence of oxygen. It is a proven technology, with 17,376 biogas plants operating in Europe in 2015 (European Biogas Association, 2016). The majority of these are agricultural or related to waste water treatment, including plants using energy crops and operating at sewage plants. AD facilities treating specifically municipal food waste are typically stand-alone commercial AD plants (Scholes and Areikin, 2014), where the process can be controlled to meet regulations associated with the treatment of the animal by-products contained in the waste (UK Government, 2014).

The composition of biogas produced in AD plants is dependent on feedstock. In a typical plant using food waste as a feedstock, biogas

consists of approximately 60% methane with the balance made up predominantly of carbon dioxide (Banks and Zhang, 2010). By comparison, mains natural gas is typically 95% methane and also contains some heavier hydrocarbons. Biogas can be burned directly to generate electricity and heat, which can be exported to the grid and used for industrial or district heating, respectively. Alternatively, the biogas may be upgraded to biomethane (Tippayawong and Thanompongchart, 2010) to be exported to mains network or used as a vehicle fuel. In addition to biogas, AD also produces the digestate – nutrient-rich solids and liquids – which is typically used as an agricultural fertiliser.

Fugitive emissions of methane produced during AD vary from one plant to another and are estimated to be in the range of 1–3% of total produced methane (Bernstad and la Cour Jansen, 2012). The AD process also produces ammonia in an intermediate step; this may be volatilised during the handling and storage of digestate (Bell et al., 2016). The digestate also has the potential to cause further methane emissions in storage from pockets of continuing anaerobic digestion (Zeshan and Visvanathan, 2014). Furthermore, the application of digestate on the land can release large amounts of ammonia to the air and nitrates to the water, causing high levels of acidification and eutrophication (Whiting and Azapagic, 2014).

There are numerous life cycle assessment (LCA) studies of AD, but most focus on agricultural rather than household food waste; for a review, see e.g. Muench and Guenther (2013) and Bernstad and la Cour Jansen (2012). Some LCA studies of AD of food waste have also been carried out, each location-specific. For example, an Irish study (Oldfield et al., 2016) assessed the global warming, acidification and eutrophication potentials of nationwide scenarios for food waste treatment via landfill, incineration, AD and composting as well as waste minimisation. The same four treatment routes were the focus of a study based in a Swedish town (Bernstad and la Cour Jansen, 2011), in which five environmental impacts were assessed: global warming, ozone depletion, photochemical oxidants formation, acidification and nutrient enrichment. The study by Eriksson et al. (2015) was also based in Sweden but focused on food waste from a specific supermarket. In addition to the four aforementioned treatment options, it considered conversion of waste to animal feed and donations for human consumption; only global warming potential was considered.

For the UK, only two studies have been found in the literature. In the first, Patterson et al. (2011) presented the endpoint impacts for the AD treatment of Welsh food waste in a range of centralised and distributed systems. In the second, Evangelisti et al. (2014) focused on multiple waste treatment options for a single London borough with midpoint results for global warming, acidification, summer smog and nutrient enrichment potentials.

As far as the authors are aware, this is the first LCA study to consider the AD treatment of all household food waste in the UK using a comprehensive set of environmental categories. Furthermore, the analysis is carried out from two perspectives, reflecting different motivations for using AD: i) as an option for treating food waste; and ii) as an option for recovery of electricity. In the first case, the impacts are compared to incineration and landfilling, currently the predominant disposal routes for food waste. For the second perspective, the impacts of electricity from AD are compared to the UK electricity mix as well as individual generation sources contributing to the mix. A detailed uncertainty analysis of the data and results has also been carried out. The results of the study are aimed at local authorities and policy makers.

2. Methods

The LCA methodology applied in this study follows the guidelines in ISO 14040/14044 (ISO, 2006a; ISO, 2006b). The following sections detail the goal and scope of the study and the inventory data used to complete the assessment.

2.1. Goal and scope

The main goal of this study is to estimate the environmental impacts of treating UK household food waste by AD and to compare it to two alternative methods used currently – incineration and landfilling – to determine if AD offers environmental benefits. Given that in addition to treating food waste AD provides electricity and fertiliser, a further aim is to compare AD to the alternative ways of supplying these outputs: UK electricity grid and the production of mineral fertilisers. Although the biogas produced by AD is used in a combined heat and power (CHP) plant to co-generate heat and electricity, only the latter is exported from the system while the former is used to heat the AD reactor. Unlike many Northern European countries, such as Sweden (Bernstad and la Cour Jansen, 2011), the UK lacks the infrastructure to utilise excess heat for district heating (Scholes and Areikin, 2014). It is therefore assumed that all excess heat is wasted and hence a comparison with alternative sources of heat is not relevant.

To enable these comparisons, two functional units are considered, reflecting different motivations for using AD:

1. *Treatment of 1 tonne of household food waste:* In this case, AD is compared to incineration with energy recovery and landfilling. System expansion has been applied to credit the AD system for the displacement of grid electricity and mineral fertiliser. No credits for heat generation have been considered as the heat is not exported from the system. The incineration and landfill systems are only credited for the electricity exported.
2. *Generation of 1 MWh of net electricity:* The impacts of electricity from AD are compared with the UK generation mix as well as with the individual electricity technologies in the mix. The system has been credited for the displacement of mineral fertiliser. As above, no credits for heat generation have been applied. Credits related to the redirection of food waste from other waste management routes are not considered as the focus for this functional unit is the valorisation of waste for electricity generation rather than comparison of treatment routes.

The scope of the study, shown in Fig. 1, is from the point of waste generation in the household to end-of-life of resources and materials produced by AD, including:

- collection and transport of food waste to the AD facility;
- construction of AD and CHP plants;
- anaerobic digestion of waste;

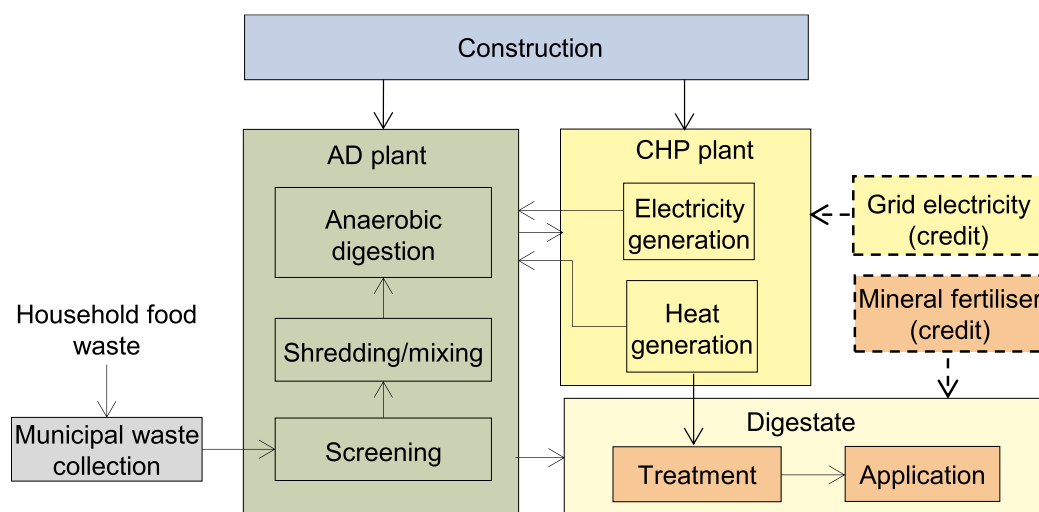


Fig. 1. Life cycle stages considered for anaerobic digestion of food waste (—— System credits. AD: anaerobic digestion. CHP: combined heat and power).

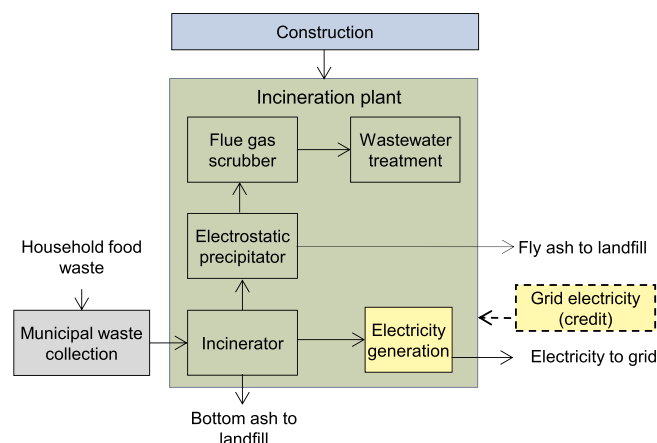


Fig. 2. Life cycle stages considered for food waste incineration with energy recovery (—— System credits. Removal of NO_x carried out during combustion in the incinerator.).

- combustion of biogas in the CHP plant to generate electricity and heat; and
- transportation and application of digestate for use as an agricultural fertiliser.

The scope for incineration and landfilling is analogous as shown in Figs. 2 and 3 and discussed further in Sections 2.2.3 and 2.2.4. De-commissioning of the facilities and recycling of waste materials are not considered.

2.2. Inventory data

2.2.1. Anaerobic digestion

Inventory data have been sourced from real AD plants, literature and the Ecoinvent database V3.3 (Ecoinvent, 2016). Due to the variability in the capacity and operating conditions of the AD plants, as well as in various other parameters, a range of their values has been considered. The mean or otherwise representative values have been selected to define a base case as detailed next. The effect of parameter ranges on the results has been explored through an uncertainty analysis as discussed in Section 2.2.1.3. A sensitivity analysis has also been carried out to test some of the assumptions made in the study (Section 2.2.1.4).

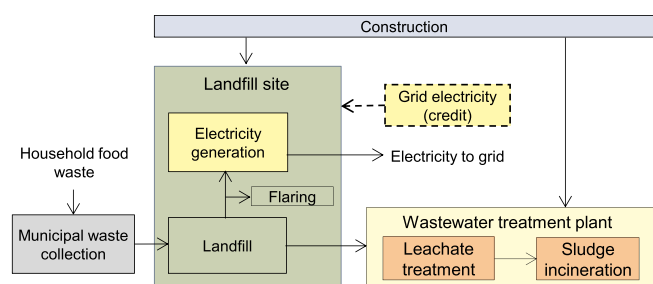


Fig. 3. Life cycle stages considered for food waste landfilling with landfill gas utilisation (— System credits. LFG: landfill gas).

Table 1

UK household food waste compositional analysis, including summer and winter variations (Esteves and Devlin, 2010).

Parameter	Unit	Summer	Winter	Mean
Total solids	%	24.2	27.7	26.0
Volatile solids	%	21.0	25.7	23.4
Elemental analysis:				
Nitrogen	%TS	3.4	3.2	3.3
Carbon	%TS	45.8	49.3	47.5
Sulphur	%TS	0.7	0.4	0.5
Hydrogen	%TS	6.3	6.5	6.4
Oxygen	%TS	35.1	37.1	36.1

2.2.1.1. Household food waste. Household food waste is assumed free of environmental burdens at the point of generation and therefore no environmental impacts are allocated to it. The waste is collected by a conventional 21 tonne municipal waste lorry, which has been modelled using Ecoinvent (2016) data. The assumed distance in the base case is 20 km, with a range of 5–30 km used in the uncertainty analysis. This transport assumption is also explored in more detail in the sensitivity analysis in Section 3.1.14.

Detailed chemical composition of typical UK food waste is shown in Table 1, showing how the summer and winter diets change its composition. However, AD plants tend to be flexible and will treat a range of organic feedstocks; therefore, a mean biogas generation rate is used in the base case, based on the mean composition of the waste shown in Table 1. With a total solids content of 26%, the AD process will operate as a dry high-solids system (Monson et al., 2007).

2.2.1.2. Anaerobic digestion facility. Municipal food waste is treated in the UK at large commercial facilities, as opposed to small farm-based plants. Therefore, the AD plant considered in this study has a 2500 m³ single-stage mesophilic reactor with a capacity to treat 25,000 t of waste per year over a lifespan of 25 years. This is representative of a typical commercial stand-alone plant in the UK treating municipal organic waste and it also represents a mean capacity of AD plants in Europe (Scholes and Areikin, 2014). Process data have been sourced from a range of European plants, as summarised in Table 2; for site-specific data, see Table S1 in the Supplementary Information (SI). The mean values of the parameters in Table 2 have been used in the base case, while the maximum and minimum values have been considered in the uncertainty analysis.

All facilities treat mixed food wastes as part of their dynamic feedstock but none is exclusive in their treatment of municipal food waste. This is typical as operators are often flexible in treating mixtures of agricultural, garden and municipal/commercial food wastes. Banks and Zhang (2010) have shown that food-waste-only feed can lead to unstable digester operation due to the accumulation of volatile fatty acids. It is therefore reasonable to assume that taking average biogas production parameters from a range of plants, which include municipal food waste in their feed mix, is representative of how food waste is treated. As shown in Fig. 1, the waste is first screened to remove

inorganic contaminants, often carried out manually (Scholes and Areikin, 2014). It is then mixed and shredded using a rotating drum pulveriser before being added to the digester using a screw press (Monson et al., 2007).

The majority of AD facilities in the UK (86%) operate under mesophilic conditions (Scholes and Areikin, 2014), also considered in this study. However, as shown in Table 2, there is no significant difference between the biogas generation rates between the mesophilic and thermophilic operating regimes. The higher temperatures of thermophilic reactors speed up the process, allowing for shorter retention time and higher conversion efficiency to biogas, but potentially at the cost of reactor stability (Guo et al., 2014; Labatut et al., 2014). Also, the average energy consumption of thermophilic plants is twice as high as that of mesophilic installations (Table 2).

Fugitive emissions of methane from the AD facility range from 1 to 3% of the amount of biogas produced (Bernstad and la Cour Jansen, 2011; Evangelisti et al., 2014; Naroznova et al., 2016); the mean value of 2% has been assumed in the base case (Table 3). Emissions of ammonia, hydrogen sulphide and nitrous oxide have been modelled using Ecoinvent data for biowaste AD (Ecoinvent, 2016).

The biogas can be burned directly without further treatment (DEFRA, 2013a; Salter and Banks, 2008) and this is assumed in the base case. However, large commercial AD plants clean the biogas to remove H₂S which can damage the CHP engine (Monson et al., 2007); the effect on the results of the biogas clean-up is considered in the sensitivity analysis. A 1 MW CHP reciprocating internal combustion engine is considered in the study based on the capacity of the AD plant (25,000 t/yr or 3.13 t/h, assuming 8000 operating hours) and the amount of electricity generated (0.277 MWh/t food waste or 0.865 MWh/h). This has also been modelled using data from Ecoinvent (2016) which are available for this CHP capacity. However, the data for construction of the 2500 m³ AD reactor are not available and had to be scaled up from the digester capacity of 500 m³ available in Ecoinvent. This has been carried out in accordance with standard chemical engineering practice for scaling up process plants, adapted for estimation of environmental impacts as follows (Towler and Sinnott, 2013; Whiting and Azapagic, 2014):

$$E_2 = E_1 \left(\frac{C_2}{C_1} \right)^{0.6} \quad (1)$$

where:

E_1 environment impact of the smaller plant

E_2 environment impact of the larger plant

C_1 plant capacity of the smaller plant

C_2 plant capacity of the larger plant.

The digestate from the AD facility must be heated to 70 °C for 1 h to ensure all pathogens related to the animal by-products in food waste are killed (UK Government, 2014) before it can be used as a fertiliser in agriculture. This heat is sourced from the CHP unit. The composition of different food-derived digestate with regard to major plant nutrients is shown in Table 3. It can be seen that there is significant variation between the minimum and maximum values for all three major plant macronutrients – nitrogen, phosphorus and potassium. Based on an extensive literature review, Bernstad and la Cour Jansen (2011) found that potassium and phosphorus have a 100% uptake ratio, suggesting that they are as effective as mineral fertilisers. The reported substitution ratios for nitrogen vary from 30% to 146%, but are typically below 100% due to the lower bioavailability. As these values are highly dependent on location and digestate composition, data for the uptake of nutrients and their emissions to air and soil have been sourced from UK field experiments on the application of food-based digestate as fertiliser (Nicholson et al., 2016). Table 3 presents these data for the emissions of nitrogen species. It can be seen that there are significant differences in

Table 2Summary of process and operating data for industrial anaerobic digestion plants treating food waste within the European Union.^a

Parameter	Unit	Type	Mean ^b	Minimum	Maximum	Source
Biogas production ^c	Nm ³ biogas/t waste	Mesophilic	137	85	187	Bernstad and la Cour Jansen (2011); Banks et al. (2011); Monson et al. (2007)
Electricity consumption (for facility) ^d	kWh/t waste	Thermophilic	122	108	138	Monson et al. (2007); Wagner et al. (2014)
		Mesophilic	23	10	44	Bernstad and la Cour Jansen (2011); Banks et al. (2011); Monson et al. (2007)
Heat consumption ^e	kWh/t waste	Thermophilic	46	26	80	Monson et al. (2007); Jungbluth et al. (2007)
		Mesophilic	82	36	113	Bernstad and la Cour Jansen (2011); Banks et al. (2011); Monson et al. (2007)
Retention time	days	Thermophilic	128	90	165	Jungbluth et al. (2007)
			17	14	25	Monson et al. (2007); Naroznova et al. (2016)

^a For a full breakdown of the site-specific data, see Table S1 in the Supplementary information.^b The mean values have been derived based on the range of data (minimum and maximum) reported in sources given in the last column in the table.^c Gas at normal conditions: 20 °C and 1 atm.^d Used in the handling, screening, shredding, mixing and controlling the processes and materials on site.^e Sites cover heat requirements internally, so do not require external heat supply.**Table 3**

Emissions from the AD plant and digestate used in LCA modelling of food waste treatment via anaerobic digestion.

Parameter	Unit	Base case	Minimum ^a	Maximum ^a	Source
Methane emissions at AD plant	% of total production	2%	1%	3%	Bernstad and la Cour Jansen (2011); Evangelisti et al. (2014); Naroznova et al. (2016)
Digestate dry solid (DS) fraction	%	4.5%	2.7%	6.8%	Rigby and Smith (2011)
Total N in digestate	% of DS	15.0%	11.9%	20.5%	Rigby and Smith (2011)
Total P in digestate	% of DS	0.7%	0.3%	2%	Rigby and Smith (2011)
Total K in digestate	% of DS	4.7%	1.4%	9.3%	Rigby and Smith (2011)
Nitrogen fertiliser displaced ^b	Equivalent % of the mass of N in the digestate	40%	18%	65%	Derived from Nicholson et al. (2016)
Potassium fertiliser displaced	equiv. % of the mass of K in the digestate		100%		Bernstad and la Cour Jansen (2011); Möller et al. (2009)
Phosphate fertiliser displaced	equiv. % of the mass of P in the digestate		100%		Bernstad and la Cour Jansen (2011); Möller et al. (2009)
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)
Nitrous oxide emission air from: digestate application	% total N emitted as N ₂ O-N	0.45%	0.20%	0.90%	Nicholson et al. (2016)
ammonium nitrate application	% total N emitted as N ₂ O-N	1.5%	0.6%	3.8%	Harty et al. (2016)
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%	5.0%	10%	Assumption based on DEFRA (2006b)

^a Used as the upper and lower limits in the uncertainty analysis.^b The nitrogen in the digestate has a lower bioavailability compared to nitrogen in the ammonium nitrate and cannot displace it on a one-to-one basis (Bernstad and la Cour Jansen, 2011).

emissions of ammonia and nitrous oxide from different sites and under different seasonal conditions. For this reason, the mean values have been used in the base case but the range of data have been considered in the uncertainty analysis.

As indicated in Table 3, for the application of digestate, on average 42% of the total applied nitrogen is emitted to air as ammonia and 0.5% as nitrous oxide, with 15% leached to water as nitrate. The remaining 42.5% of the nitrogen is retained in the soil for plant uptake. By comparison, UK-based experimental research (van der Weerden and Jarvis, 1997) shows that the average percentage of applied nitrogen lost through ammonia volatilisation after application of mineral fertilisers ammonium nitrate and calcium nitrate is below 1%. The application of urea results in approximately 29% volatilisation of total nitrogen as ammonia (van der Weerden and Jarvis, 1997). Similar results were found by DEFRA (2006a), with the average ammonia volatilisation of 2% from the application of ammonium nitrate, 27% for urea applied on grassland and 22% for arable land. However, urea showed a very large range of ammonia emissions (2–58% of applied nitrogen), dependent largely on weather factors. The same study also found that the most commonly used fertiliser was ammonium nitrate at 85% of total nitrogen, with the remaining 15% being urea-based products. Hence, it is

assumed here that AD digestate would displace ammonium nitrate, meaning that ammonia emissions from the former will be higher from ammonium nitrate (42% vs 2% of total nitrogen applied, respectively). The effects of the displacement of urea are explored in the sensitivity analysis in Section 3.1.14.

The nitrous dioxide emissions following the application of fertiliser are set by the Intergovernmental Panel on Climate Change (IPCC) at a default value of 1% of total nitrogen applied (De Klein et al., 2006). Table 3 shows that even in the worst case, the emissions from the application of food-derived digestate are below 1%. Harty et al. (2016) found that the average emissions of nitrous oxide after mineral fertiliser application in six sites across Ireland were equivalent to 0.49% of the total nitrogen applied. However, these values for ammonium nitrate ranged from 0.58% to 3.81%, with an average value of 1.49%, whereas for urea they ranged from 0.1 to 0.49%, averaging at 0.25%. As it is assumed that the digestate displaces ammonium nitrate, it is possible that this would result in lower nitrous dioxide emissions. However, due to the uncertainty in the data, the emissions are assumed to be equal for both the digestate and ammonium nitrate and no credit has been applied.

DEFRA (2006b) also studied the loss of nitrogen to ground water at

eight UK sites and found that the total emissions ranged from 0.7% to 23% of the total applied mineral nitrogen. The upper value represents a worst case scenario as irrigation water was applied soon after fertiliser application. Emissions to soil also range widely, depending on soil type and weather conditions, but the nitrate losses to the soil for food-derived digestate presented in Table 3 are within the upper range of the values found for ammonium nitrate. Therefore, it is assumed in the base case that 50% of the nitrate emissions released from the application of the digestate would otherwise have been released if ammonium nitrate was used. The uncertainty in this assumption is addressed by varying the emission of nitrates as shown in Table 3. A summary of the other emissions data used in the study can also be found in this table.

The influence on the results of the above assumptions regarding nitrate and nitrous oxide emissions is explored further in the sensitivity analysis.

2.2.1.3. Uncertainty analysis. The uncertainty analysis has been carried out to explore the effect of the data ranges on the environmental impacts. The Monte Carlo (MC) method, as implemented in GaBi V8 (Thinkstep, 2017), has been used for this purpose. The data in Tables 2 and 3 represent the upper and lower limits used as the boundaries in the MC simulations. A uniform distribution has been chosen as the source data are not extensive enough to determine a normal distribution reliably. The minimum and maximum values are all from real plants or from full scale field tests and therefore represent realistic scenarios rather than statistical extremes. A value for each variable parameter in Tables 2 and 3 has been chosen randomly in each of the 10,000 simulation runs to determine the variations in the impacts. The exception to this are the retention time, as it is dependent on the variations in the electricity and heat values and the P/K substitution rates which are fixed at 100%.

2.2.1.4. Sensitivity analysis. In addition to the uncertainty analysis, a sensitivity analysis has been performed to test the influence of the some of the assumptions that could affect the results. These are:

- transport of food waste;
- emissions of nitrate and nitrous oxide from digestate;
- displacement of urea instead of ammonium nitrate; and
- biogas clean-up to remove H_2S .

2.2.2. UK electricity mix

The environmental impacts of UK grid electricity have been estimated using 2016 as the base year; the generation mix is shown in Table 4. Data for the individual electricity technologies present in the mix have been sourced from Ecoinvent (2016). The estimated impacts of the grid electricity can be found in Table S2 in the SI.

2.2.3. Incineration with energy recovery

The life cycle diagram of the incineration system is shown in Fig. 2. The following stages are considered:

- collection and transport of food waste;
- construction of the incineration plant;
- operation of the incinerator to generate electricity; and
- treatment of waste streams.

Moving grate furnace, the most common type used in incineration facilities in the UK (DEFRA, 2013b), has been considered. The

Table 4
UK electricity generation mix for 2016 (BEIS, 2017b).

Coal	Oil	Gas	Nuclear	Hydro (natural flow)	Wind and solar	Bioenergy	Pumped storage	Other fuels
9%	1%	42%	21%	2%	14%	9%	1%	1%

Table 5
Inventory data for incineration of UK household food waste.

Parameter	Unit	Value	Source
Gross electricity generation	kWh/t ^a	255	Environment Agency (2016)
Parasitic electricity demand	kWh/t	81	Environment Agency (2016)
Imported electricity	kWh/t	6	Environment Agency (2016)
Auxiliary fuel (natural gas)	kWh/t	11.7	Ecoinvent (2008)
Slag sent to landfill	kg/t	6.2	~ II ~
Residual sent to landfill	kg/t	2.7	~ II ~
DeNox process consumables			
Ammonia	kg/t	0.4	~ II ~
Chromium oxide	g/t	0.25	~ II ~
Scrubber columns consumables			
Hydrochloric acid	g/t	0.59	~ II ~
Sodium hydroxide	kg/t	0.5	~ II ~
Wastewater treatment consumables			
Quicklime (CaO)	g/t	4.2	~ II ~
Iron (III) chloride	g/t	0.33	~ II ~

^a FW: food waste.

combustion gases are passed through a steam turbine to produce electricity, after which they are treated before release to the atmosphere. This includes removal of nitrogen oxides, particulates and acid gases; the waste water generated from the treatment of the latter is treated before being discharged into the environment. Based on UK operating plants, the modelled facility treats 300,000 tonnes of waste per year (Environment Agency, 2016). For the inventory data, see Table 5.

The mean composition of food waste is assumed in the modelling (Table 1) with the lower heating value (LHV) of 3.8 MJ/kg (LoRe and Harder, 2012; Roberts, 2015). This is lower than the LHV of general MSW of 8.9 MJ/kg (Environment Agency, 2016) in which the food waste is embedded. Therefore, the ratio of the two LHVs has been used to estimate the electricity generation rates from food waste. The resultant mean electricity generated from 1 tonne of food waste is 255 kWh, of which 81 kWh are used on site, with an additional 6 kWh imported from the grid (Environment Agency, 2016). The exported electricity is credited for displacing the UK generation mix (BEIS, 2017b).

The environmental impacts of incineration have been estimated following the approach in Doka (2009) and using an Ecoinvent tool for modelling the incineration of waste (Ecoinvent, 2008), based on the mean composition in Table 1. As for the AD system, decommissioning of the plant and recycling of waste materials are not considered.

2.2.4. Landfilling with gas utilisation

As shown in Fig. 3, the landfill system comprises the following life cycle stages:

- collection and transport of the food waste;
- construction of the landfill site;
- operation of the landfill;
- landfill gas (LFG) collection and utilisation; and
- treatment of leachate.

UK sanitary landfill sites are designed to be closed systems with collection of methane-rich landfill gas (LFG) for electricity generation. Large-scale modern UK landfill sites are estimated to capture 68% of the LFG with 9.1% of the captured gas being flared rather than utilised for energy (Gregory et al., 2014). The LFG is burned in a gas engine to generate electricity with a net efficiency of 36%, resulting in an average

Table 6

Inventory data for the treatment of UK household food waste at a landfill facility.

Parameter	Unit	Value	Source
Gross electricity generated	kWh/t ^a	68	Gregory et al. (2014)
Parasitic electricity demand	kWh/t	6.7	Gregory et al. (2014)
LFG capture efficiency	%	68%	Gregory et al. (2014)
Fraction of captured LFG ^b flared	%	9.1%	Gregory et al. (2014)
Auxiliary fuel (light fuel oil)	kWh/t	0.14	Ecoinvent (2008)
Leachate production	m ³ /t	2.5	~ II ~
Leachate treatment consumables			
Iron sulphate	g/t	16	~ II ~
Aluminium sulphate	g/t	4.4	~ II ~
Titanium dioxide	g/t	0.23	~ II ~
Sludge incineration consumables			
Chromium oxide	g/t	0.0048	~ II ~
Hydrochloric acid	g/t	0.00053	~ II ~
Auxiliary fuel (natural gas)	kWh/t	0.39	~ II ~
Sodium hydroxide	g/t	0.41	~ II ~
Iron (III) chloride	g/t	22	~ II ~
Quicklime (CaO)	g/t	0.074	~ II ~

^a FW: food waste.

^b Landfill gas.

61 kWh exported to the grid per tonne of food waste; this takes into account the parasitic electricity of 6.7 kWh/t. The system has been credited for the grid exports. The leachate from the landfill is collected in the sewer and treated at a wastewater treatment plant, with the activated sludge being incinerated.

The environmental impacts of the process have been modelled using the same Ecoinvent tool as for incineration (Ecoinvent, 2008), for the mean waste composition specified in Table 1. The landfill process used in the model assumes a watertight excavation area of 90,000 m² at a depth of 20 m (Doka, 2009). The inventory data for landfilling are summarised in Table 6.

2.3. Impact assessment

The modelling has been carried out using GaBi V8 software (Thinkstep, 2017). The ISO 14040/14044 (ISO, 2006a; ISO, 2006b) does not recommend any specific impact assessment method to be used. The International Reference Life Cycle Data System (ILCD) handbook (Wolf et al., 2012) and the Product Environmental (PEF) Guide (Manfredi et al., 2012) both recommend a differing set of impact methodologies, including ReCiPe. The latter is the most up-to-date and widely used by LCA practitioners and is therefore also used in this study. The following mid-point impact categories are considered, estimated by applying the “hierarchist” perspective (Goedkoop et al., 2013): global warming potential; fossil depletion; metal depletion; freshwater ecotoxicity; marine ecotoxicity; terrestrial ecotoxicity; human toxicity; marine eutrophication; freshwater eutrophication; terrestrial acidification; particulate matter formation; photochemical oxidants formation; ozone depletion; agricultural land occupation; urban land occupation; natural land transformation; ionising radiation; and water depletion.

In addition, primary energy demand has also been estimated (gross calorific value), considering both renewable and non-renewable energy resources (Thinkstep, 2017).

3. Results and discussion

3.1. Functional unit: 1 tonne of food waste

The total impacts from the treatment of 1 tonne of food waste via AD are presented in Fig. 4 with credits applied for the displacement of electricity and mineral fertilisers; the contribution of different life cycle stages are detailed in Fig. 5. It can be observed from Fig. 4, that 13 out

of the 19 categories have negative values, indicating net savings in these impacts. These are attributable to the displacement of grid electricity and mineral fertiliser. The following sections discuss results for each impact category in turn, with maximum and minimum values for AD referring to the 10th and 90th percentiles. These are compared to the impacts of incineration with energy recovery and landfilling in Section 3.1.11. All the impacts below are expressed per tonne of food waste.

3.1.1. Primary energy demand

The primary energy demand of food waste treatment via AD ranges from −1.76 to −2.27 GJ/t, with a base case value of −1.99 GJ/t, indicating a net saving in energy. This is predominantly attributable to the credit of −2.30 GJ/t for displacing grid electricity. The primary energy use in the AD system is principally related to the transportation of waste and digestate (Fig. 5). The anaerobic digestion facility has its heat and electricity demand covered internally by the CHP engine which is fuelled with the produced biogas; therefore, no external energy is required.

3.1.2. Global warming potential

The AD also shows net savings for this impact, which ranges from −17 to −70 to kg CO₂-eq./t, with the base case value of −39 kg CO₂-eq./t. The GHG emissions are due to CH₄ losses from AD (41 kg CO₂-eq./t) and CHP flue gases (3 kg CO₂-eq./t), CO₂ emissions from transportation of the waste and digestate (30 kg CO₂-eq./t) and N₂O emissions from the application of digestate on the land (12 kg CO₂-eq./t). The emissions of CO₂ from the combustion of biogas have not been considered because of their biogenic nature. The displaced grid electricity provides the majority of the credits (−91 kg CO₂-eq./t); the credits for avoiding the manufacture of ammonium nitrate are also significant (−21 kg CO₂-eq./t). The N₂O emissions from digestate application are matched by N₂O emissions from the application of mineral nitrogen fertiliser (see Section 2.2.1.2).

3.1.3. Depletion of fossil and metal resources

Depletion of fossil resources in the base case is estimated at −22 kg oil-eq./t, ranging from −18 to −28 kg oil-eq./t. The only significant contributor is the transportation of food waste (10 kg oil-eq./t) and distribution of digestate for use on the land (1.2 kg oil-eq./t). The displacement of grid electricity, which contains 52% of fossil fuels in the mix (BEIS, 2017b), provides the majority of the credits to the system (−30 kg oil-eq./t).

Between 0.5 and 1.8 kg Fe-eq./t of metals can be saved by using AD, with the saving in the base case of 0.9 kg Fe-eq./t. The CHP engine is the largest contributor (0.6 kg Fe-eq./t) to the metals depletion, with transport (0.4 kg Fe-eq./t) and the AD equipment (0.2 kg Fe-eq./t) being the only other significant life cycle stages. As mentioned earlier, the decommissioning of the facility is not considered, due to the relatively small impacts of the plant so no credits are included for recycling of the construction materials. The credits to the system are shared between the displacement of grid electricity (−1.1 kg Fe-eq./t) and the mineral fertiliser (−1.0 kg Fe-eq./t). It can be seen in Fig. 6 that the majority of the credits are associated with the displacement of nitrogen in industrially-produced ammonium nitrate.

3.1.4. Human toxicity and ecotoxicity

The human toxicity potential ranges from −20 to −28 kg 1,4-dichlorobenzene (DB) eq./t, with the base case impact of −22 kg 1,4-DB eq./t. Transportation (3.1 kg 1,4-DB eq./t) and the production of metal components of the CHP plant (2.1 kg 1,4-DB eq./t) are the main contributors to this impact. The majority of human toxicity from transport (2.0 kg 1,4-DB eq./t) is due to the emissions from waste collection lorries. Most of the net savings in the impact are related to the displaced electricity (−22 kg 1,4-DB eq./t).

The base-case values for freshwater, marine and terrestrial ecotoxicities are −3.3, −3.0 and −0.005 kg 1,4-DB eq./t, respectively.

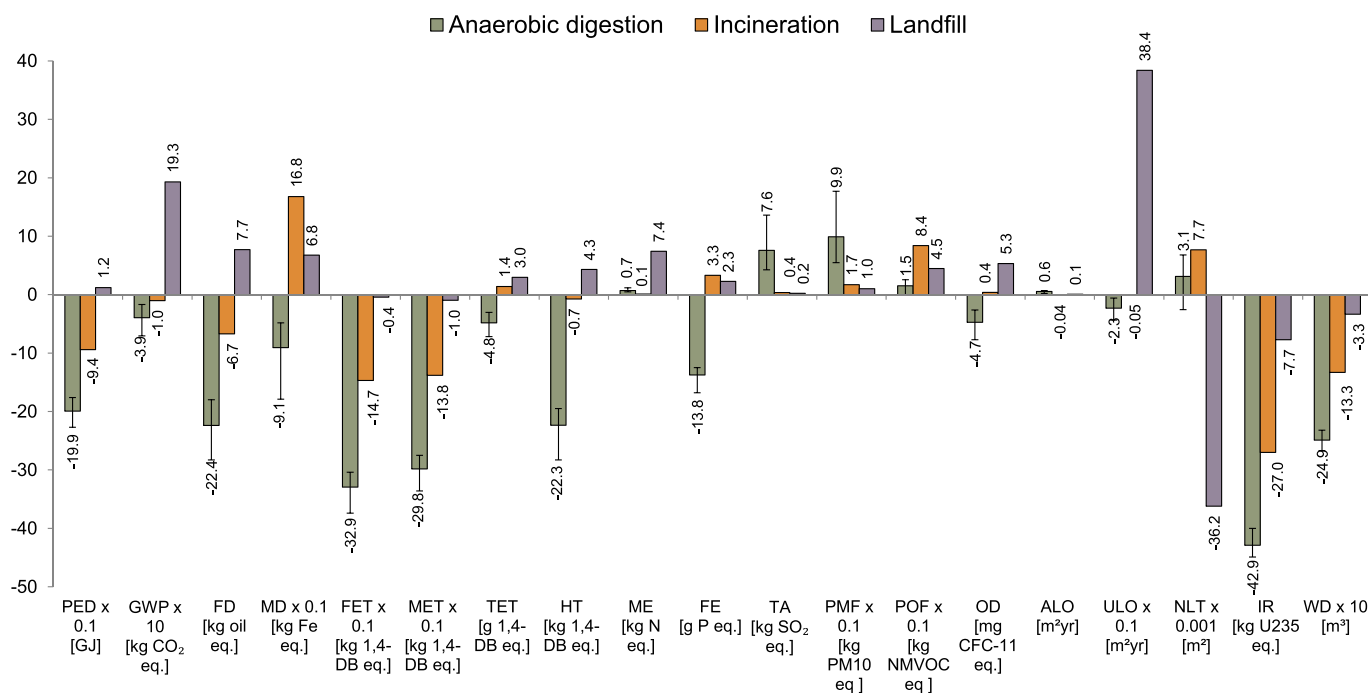


Fig. 4. Environmental impacts of treating food waste by anaerobic digestion (All impacts expressed per tonne of food waste treated. The solid bars and the data labels for the AD refer to the base case. The error bars represent the 90th and 10th percentile impacts obtained through Monte Carlo analysis using the inventory ranges in Tables 2 and 3. The values for incineration and landfill correspond to the average annual composition of waste (Table 1). Some impacts have been scaled – to obtain the original value, multiply with the factor shown on x-axis where relevant. 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; 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.).

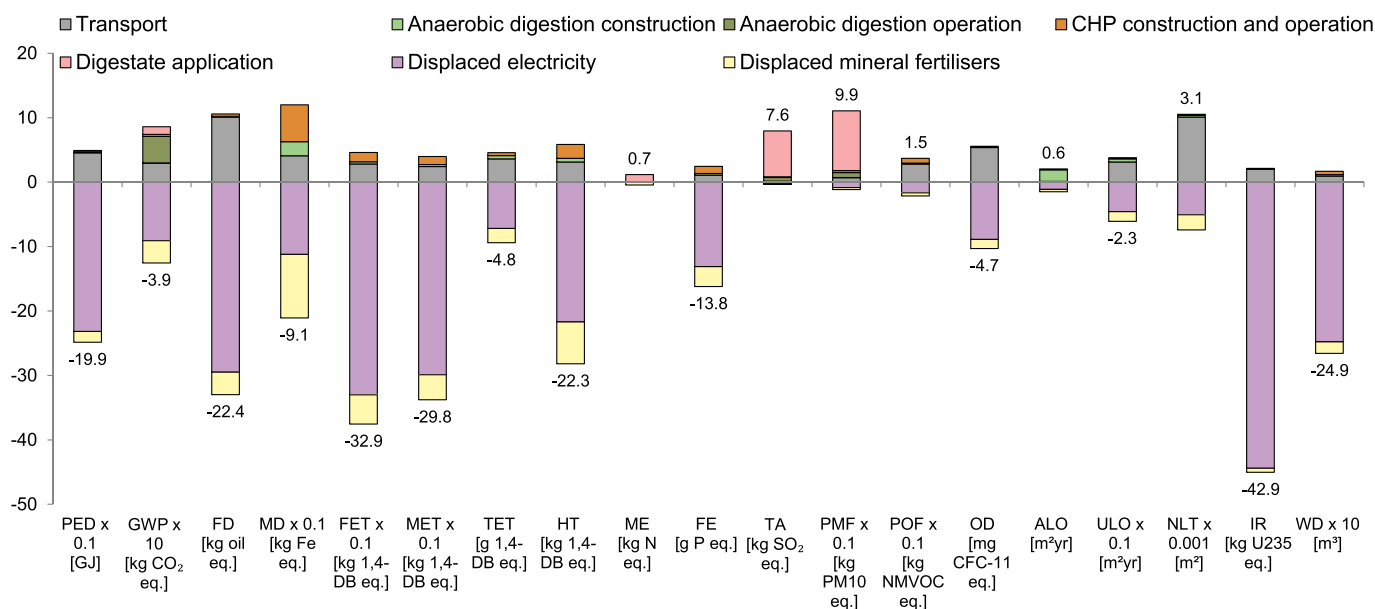


Fig. 5. Contribution of different life cycle stages to the impacts from treatment of food waste by anaerobic digestion (The results shown only for the base case. All impacts expressed per tonne of food waste treated. Some impacts have been scaled – to obtain the original value, multiply with the factor shown on x-axis where relevant. For impacts nomenclature, see Fig. 4).

Similarly to human toxicity, the majority of these is from the diesel lorries and the production of metal components for the CHP plant (Fig. 5). The credits are due primarily to displaced electricity from the hard coal and solar PV present in the grid (see Fig. 12).

3.1.5. Eutrophication and acidification

Marine eutrophication in the base case has a value of 0.72 kg N-eq./t, with a range of 0.48–1.19 kg N-eq./t. At 1.13 kg N-eq./t, the application of the digestate on the land is the greatest source of this impact

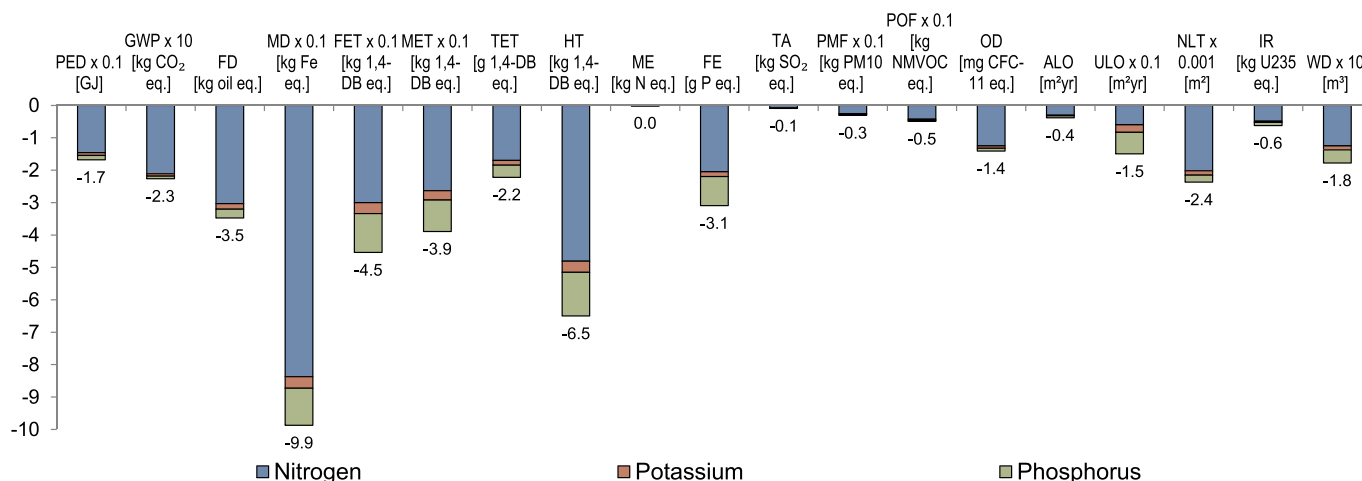


Fig. 6. Environmental credits for the displacement of mineral fertiliser (ammonium nitrate) by digestate (The results shown only for the base case. All impacts expressed per tonne of food waste treated. Some impacts have been scaled – to obtain the original value, multiply with the factor shown on x-axis where relevant. For impacts nomenclature, see Fig. 4.).

due to the emissions of ammonia and nitrates, in excess of the emissions from the application of ammonium nitrate. The displacement of grid electricity and production of mineral fertilisers provide negligible savings. Freshwater eutrophication varies from -13 to -17 g P eq./t (-14 g P eq./t in the base case). Phosphates are a major contributor to this category while the displacement of grid electricity provides the majority of the net savings (-13 g P eq./t).

The terrestrial acidification potential in the base case is equal to 7.6 kg SO₂ eq./t, ranging from 4.3 to 13.6 kg SO₂ eq./t. The gaseous emission of ammonia following the application of digestate on the land is the only significant contributor to acidification.

3.1.6. Particulate matter and photochemical oxidants

The emissions of ammonia from the application of digestate on the land cause the majority of particulate matter (PM₁₀) formation. In the base case, this amounts to 0.93 kg PM₁₀ eq./t, out of 0.99 kg PM₁₀ eq./t in total (Fig. 5). Across the range of the parameters considered, the PM₁₀ formation varies from 0.5 to 1.8 kg PM₁₀ eq./t (Fig. 4).

The photochemical oxidants creation potential, or summer smog, has a base-case value of 0.15 kg non-methane volatile organic compounds (NMVOC) eq./t, with a range from -0.002 – 0.26 kg NMVOC eq./t. Nitrous oxides and NMVOC produced in the combustion of fuel in the transport vehicles (0.28 kg NMVOC eq./t) and of the biogas in the CHP engine (0.07 kg NMVOC eq./t) are the two major contributors to summer smog. The impact is reduced by the savings achieved from displacing grid electricity (-0.17 kg NMVOC eq./t) and mineral fertiliser (-0.05 kg NMVOC eq./t).

3.1.7. Ozone depletion

The transportation of the waste and digestate is the major contributor to ozone depletion at 5.4 mg chlorofluorocarbons (CFC)-11 eq./t. With credits to the system for the displaced grid electricity and mineral fertiliser, the overall impact in the base case is -4.7 mg CFC-11 eq./t, ranging from -2.6 to -7.7 mg CFC-11 eq./t across the parameter values.

3.1.8. Land occupation and transformation

The major land occupation related to AD is typically the agricultural land used to grow the feedstock; however, in this case the food waste is considered free of environmental burdens. Hence, the main contributor to agricultural land occupation is the AD plant itself (1.8 m²/t), assuming its location in a rural setting; this impact would be directly transferrable to urban land occupation if the plant was located in an urban environment. With the system credits, the total occupation of

agricultural land reduces to 0.55 m²/t in the base case, ranging from 0.21 to 0.71 m²/t.

There is a net saving for urban land occupation, ranging from -0.06 to -0.43 m²/t (-0.23 m²/t in the base case). Transport is the largest contributor (0.31 m²/t) primarily due to the road use (0.25 m²/t), but the displaced electricity (-0.46 m²/t) and mineral fertiliser (-0.15 m²/t) provide overall net savings. Natural land transformation is negligible (0.003 m²/t in the base case).

3.1.9. Irradiation

The electricity credits lead to the net saving in ionising radiation, which ranges from -40 to -44.9 kg ²³⁵U eq./t and averages -42.9 kg in the base case. The reason for these savings is the displacement of nuclear energy which contributes 21% to the grid electricity. Transport is the only significant contributor at 2 kg ²³⁵U eq./t.

3.1.10. Water depletion

In the base case, 249 m³/t of water is saved, with a range from -232 to -267 m³/t. With no significant external water use in the AD process, the water savings are primarily due to the electricity credits (-248 m³/t in the base case).

3.1.11. Comparison with incineration and landfill

Fig. 4 shows that AD has lower impacts than incineration for 15 out of the 19 environmental categories considered. The releases of ammonia and nitrates, predominantly from the application of digestate, lead to higher marine eutrophication, terrestrial acidification and particulate matter formation. Agricultural land occupation is also higher for AD due to the facility processing a relatively small amount of waste compared to an incineration plant, resulting in a greater plant footprint per tonne of waste treated.

Incineration provides net savings across eight of the impacts (although these are smaller than for AD), including primary energy demand and global warming potential. However, it is a significant net contributor to particulate matter formation due to flue gas emissions, as well as to freshwater eutrophication due to the ash disposal.

Compared to the landfilling, 15 out of the 19 impacts are lower for AD. Although AD remains the worst alternative for terrestrial acidification and particulate matter formation, landfill has higher marine eutrophication due to the leachate management. Landfill also outperforms AD for natural land transformation due to a requirement to regenerate landfills at the end of their life. The displacement of a small quantity of grid electricity from the combustion of landfill gas leads to savings in freshwater and marine ecotoxicity, irradiation and water

depletion, but landfill is a net contributor to primary energy demand and global warming potential.

3.1.12. Comparison of results with other studies

As discussed in the introduction, other LCA studies of treating food waste by AD are available but a direct quantitative comparison is not possible due to different system boundaries, assumptions and life cycle impact assessment methodologies. For example, Oldfield et al. (2016) and Evangelisti et al. (2014) considered the treatment of 1 t of food waste in Ireland and the UK, respectively, but used CML 2001 (Guinée, 2002) and EDIP97 (Wenzel et al., 1997) methodologies. They also estimated only four impacts: global warming, acidification, eutrophication and photochemical oxidants.

In an attempt to enable comparison, the above four impacts obtained in this study through ReCiPe have been recalculated using the CML 2001 method. For the global warming potential, the resulting value is $-33.9 \text{ kg CO}_2 \text{ eq./t}$ (compared to $-39.5 \text{ kg CO}_2 \text{ eq./t}$ obtained with ReCiPe). Oldfield et al. (2016) and Evangelisti et al. (2014) determined this impact to be -64.7 and $-73.4 \text{ kg CO}_2 \text{ eq./t}$, respectively. The major cause of these differences is the assumption on the global warming potential of the UK grid and the related system credits. In this study, this value is $358 \text{ kg CO}_2 \text{ eq./MWh}$, compared to $567 \text{ kg CO}_2 \text{ eq./MWh}$ assumed by Evangelisti et al. (2014). The former is lower as it is more recent, reflecting the falling share of coal and an increase in solar and wind energy in the mix (BEIS, 2017b).

The recalculated (CML) acidification potential is $4.9 \text{ kg SO}_2 \text{ eq./t}$ in the current study, whereas Oldfield et al. (2016) and Evangelisti et al. (2014) reported net savings (-0.092 and $-0.017 \text{ kg SO}_2 \text{ eq./t}$, respectively). A similar discrepancy is found for the CML eutrophication potential: $1.24 \text{ kg PO}_4^{3-} \text{ eq./t}$ in this study, compared to $-0.003 \text{ kg PO}_4^{3-} \text{ eq./t}$ found by Oldfield et al. (2016). Evangelisti et al. (2014) used EDIP97 to determine this impact, reporting a value of $0.003 \text{ kg NO}_3 \text{ eq./t}$. When the results of this study are converted using this method, the value is $12.8 \text{ kg NO}_3 \text{ eq./t}$. The emission of ammonia, following application of food digestate on the land is the primary cause of both these impacts. In this work, field results from a UK study into these emissions have been used (Nicholson et al., 2016), whereas the other two studies have made assumptions or not clearly stated the emissions from application. Further to this, the larger coal fraction in the avoided electricity mixes in the literature mitigates better the eutrophication potential due to the larger system credits, hence yielding a lower overall impact than estimated here.

Evangelisti et al. (2014) also reported photochemical oxidants creation potential estimated through the CML method. The equivalent recalculated value in this study is $0.037 \text{ kg C}_2\text{H}_4 \text{ eq./t}$, compared to $0.102 \text{ kg C}_2\text{H}_4 \text{ eq./t}$ found by Evangelisti and co-workers. It has been shown in this paper that the combustion of biogas in the CHP engine and the transport of waste are the major contributors to the oxidants formation. Evangelisti et al. (2014) show that this impact is primarily attributable to NMVOC emissions from the CHP engine; however, their CHP emissions model is based on natural gas as opposed to the biogas used in this paper, which has lower NMVOC emissions.

Thus, the limited comparisons possible show that there is a correlation between the results in this study and in the literature and clear reasons for the differences.

3.1.13. Uncertainty analysis

The results of the uncertainty analysis for the functional unit of 1 tonne of food waste are presented in Fig. 7 and Table S3 in the SI. Note that the Monte Carlo mean values differ from those of the base case as a uniform distribution of the inventory ranges has been simulated while the base case considers the arithmetic means or other assumptions as outlined in Section 2.2.

It can be seen in Fig. 7 that the variation in the impacts is relatively small and all environmental impacts with net savings in the base case remain net savers for the 90th percentile value of impacts. The ranges

for the impacts with net saving, such as primary energy demand, global warming potential, depletion of metals and toxicities, are primarily driven by the variation in the amount of biogas and the associated electricity that can be produced. The ranges in eutrophication, acidification, particulate matter and photochemical oxidants are influenced primarily by the uncertainty in the emissions from digestate application and credits for displacing mineral fertilisers.

Therefore, the outcomes of the uncertainty analysis, reflecting the effects of inherent variability in AD performance, suggest that the results are robust across the range of the values considered. The confidence in the results is strengthened further by the low average coefficient of variation across the impacts of 0.37 (Table S3). Therefore, the base case results are arguably a good representative model of the data ranges available and used in the study.

3.1.14. Sensitivity analysis

The results of the sensitivity analysis for the four parameters considered are discussed in the following sections, based on the functional unit of 1 tonne of food waste.

i) Transport of food waste

The assumption in the base case is that the waste transport involves 20 km of stop/start driving related to street collections (Ecoinvent, 2016). However, this may overestimate some impacts as, after collection, the transportation to the treatment facility would be a continuous journey. To explore the effect of the latter, it is assumed here that the waste collection involves 10 km of stop/start driving, while the remaining 10 km is a continuous journey to the AD facility. The results are shown in Fig. 8 for the impacts for which a deviation from the base case is $> 5\%$ (the average deviation for the remaining impacts is 2.2%). For the mean values, the impacts in the base case are higher by 8% (primary energy demand) to 123% (natural land transformation); global warming potential is 29% greater. This is due to the higher fuel use in the base case. However, these differences are within the uncertainty ranges estimated for the base case, except for primary energy demand.

ii) Emissions of nitrate and nitrous oxide from digestate

A UK field study used in the base case found that nitrate emissions were 50% higher from digestate than from ammonium nitrate (Nicholson et al., 2016). However, the findings of another study (Walsh et al., 2012) suggest digestate to have lower nitrate emissions. Given this lack of agreement in the literature, it is assumed in the sensitivity analysis that the emissions from ammonium nitrate are 50% higher than those from digestate. The results in Fig. 9a show that in that case the system has a net savings for marine eutrophication, suggesting that digestate is a much better option for this impact than ammonium nitrate. The other impacts are not affected.

Due to data uncertainty, nitrous oxide emissions from digestate and ammonium nitrate have been assumed equal in the base case. Fig. 9b presents the results if the average emissions given in Table 3 are considered instead. The slightly higher emission rate from ammonium nitrate leads to greater credits for its avoidance through the application of digestate, which doubles the savings in greenhouse gas emissions to $-81.9 \text{ kg CO}_2 \text{ eq./t}$. This means that the global warming potential of the whole AD system is 40% lower than in the base case. If future studies can consistently confirm the lower nitrous oxide emissions from digestate, this will significantly strengthen the case for using digestate instead of mineral fertilisers in terms of global warming potential. No other impacts are affected.

iii) Displacement of urea by digestate

It has been assumed in the base case that digestate replaces

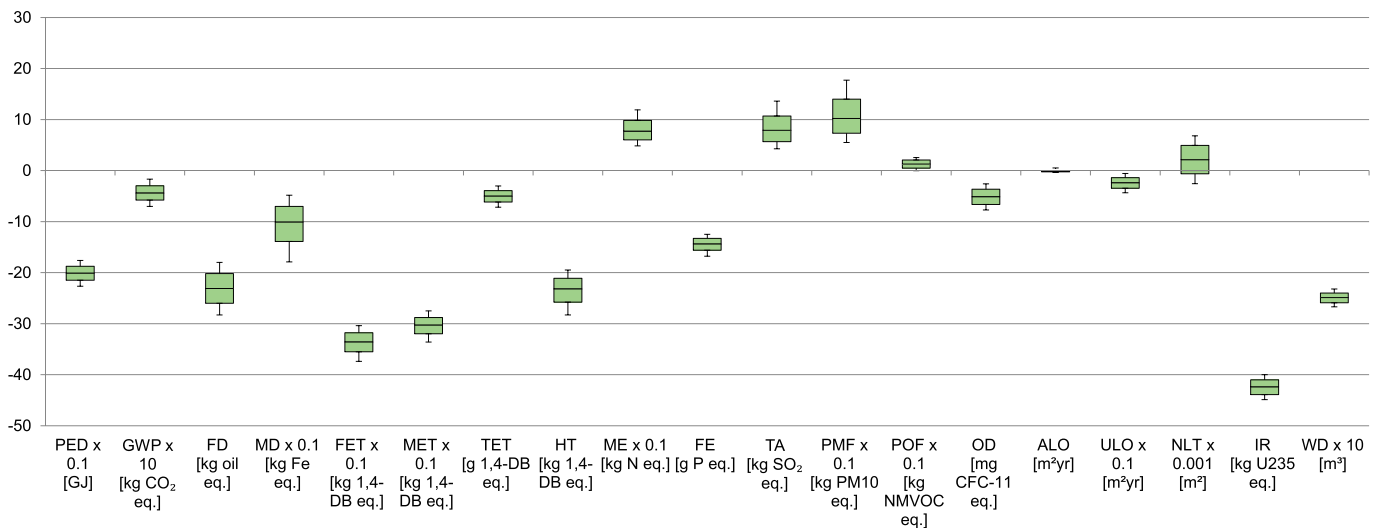


Fig. 7. Monte Carlo simulation results for the treatment of food waste by anaerobic digestion (All impacts expressed per tonne of waste treated. The inventory ranges explored over 10,000 simulation runs using uniform distribution. The boxes represent the 25th percentile, median and 75th percentile values while the whiskers represent the 10th and 90th percentile. Some impacts have been scaled – to obtain the original value, multiply with the factor shown on x-axis where relevant. For the impacts nomenclature, see Fig. 4.).

ammonium nitrate as the latter represents 85% of the UK market, with the rest occupied by urea (DEFRA, 2006a). To explore the effect of this assumption, it is assumed here that urea is displaced by digestate instead. As for ammonium nitrate, emissions of nitrous oxide from urea are assumed to be equal to those from digestate.

As can be seen in Fig. 10, the effect on the affected impacts is mixed, with some increasing and others decreasing. Urea has higher ammonia emissions and therefore the credits for its displacement reduce terrestrial acidification and particulate matter formation by more than 50%. On the other hand, urea has lower nitrate emissions than digestate and hence the marine eutrophication increases but remains within the uncertainty ranges of the base case. Furthermore, the production of urea has lower resource demands and global warming potential than ammonium nitrate and so the credits for digestate use are reduced. As a result, the net savings for metal depletion and global warming potential reduce towards the 10th percentile range of values of the base case.

iv) Biogas desulphurisation

The biogas can be used in a CHP plant directly without any treatment, which has been assumed in the base case. However, large scale plants often include desulphurisation to remove hydrogen sulphide and protect the gas engine from damage. Various options can be used for these purposes. As assessing them all is beyond the scope of this work, the sensitivity analysis focuses on the two simplest and most common methods (Allegue and Hinge, 2014):

- use of ferric chloride to precipitate hydrogen sulphide from the biogas into an insoluble iron sulphide salt; and
- adsorption of hydrogen sulphide on activate carbon impregnated with sodium hydroxide.

The description of these methods can be found in the SI. This simplified analysis considers only the consumables required for H₂S removal; energy consumption and infrastructure are excluded. The LCA data for the consumables have been sourced from Ecoinvent.

The results in Fig. S1 suggest that using ferric chloride increases all impacts from AD on average by 22%. The highest increase of 77% is found for metal depletion, related to the iron ore used in the production of ferric chloride. The treatment with activated carbon also increases all impacts, but by a much smaller margin (3% on average) and well within the uncertainty ranges of the base case. Therefore, the results suggest that activated carbon can be added to the AD system with little additional environmental penalty while the treatment with ferric chloride is much less environmentally attractive.

3.2. Functional unit: 1 MWh of electricity generated

As can be seen in Fig. 11, electricity generation from anaerobic digestion of food waste has net savings for freshwater eutrophication, human toxicity and water depletion as the system is only credited for the displacement of mineral fertiliser. The contribution analysis shows a similar trend to that discussed for the functional unit of 1 tonne of waste, with the main contributors to most impacts being transport and

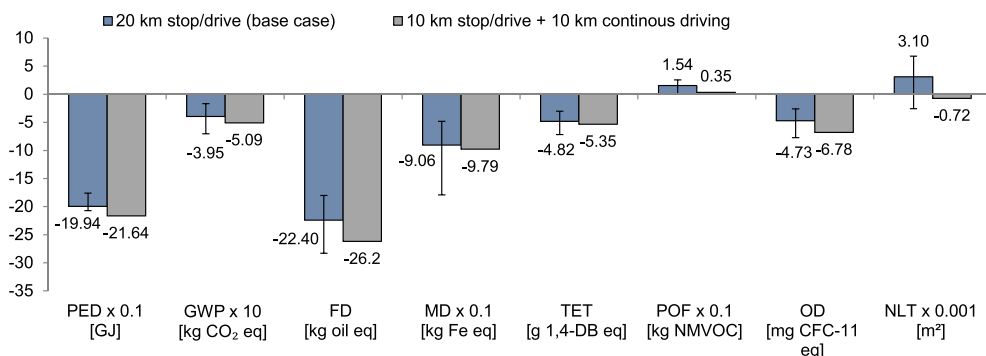


Fig. 8. Effect of the truck-driving mode during waste collection on the impacts of anaerobic digestion (All impacts expressed per tonne of food waste treated. The values on top of the graph bars represent the mean values and the error bars represent impact ranges (90th and 10th percentile) for the base case. Some impacts have been scaled – to obtain the original value, multiply with the factor shown on x-axis where relevant. For impacts nomenclature, see Fig. 4.).

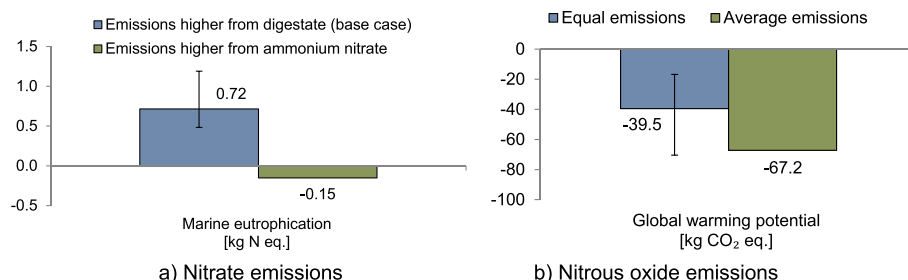


Fig. 9. Effects of nitrate and nitrous oxide emissions on the impacts of anaerobic digestion. (The impacts expressed per tonne of food waste treated. For impacts nomenclature, see Fig. 4.)

digestate application. Therefore, this discussion is not repeated but instead the focus is on the comparison of electricity from the AD system with the grid and the individual electricity technologies present on the grid; these results are presented in Fig. 12.

Overall, electricity generated from the anaerobic digestion of food waste has lower impacts than the 2016 UK grid mix for 13 of the 19 impacts, including primary energy demand, global warming potential, fossil and metal depletion, as well as all toxicity impacts. Against the individual generation options, AD performs the best in four categories: primary energy demand, human toxicity, freshwater eutrophication and water depletion.

The primary energy demand of AD electricity is the lowest of all compared alternatives. The reason for this is that, with the exception of transport and the construction of the AD facility, the AD system does not require external energy sources. Furthermore, the food waste is regarded as burden free; in contrast, energy is required for the processing of fossil fuels and the materials used in the construction of wind turbines and solar panels.

The global warming potential of AD electricity is 203 kg CO₂-eq./MWh, less than half that of a conventional natural gas power plant (499 kg CO₂-eq./MWh) but significantly greater than the impact of nuclear and renewables, which range from 12 to 97 kg CO₂-eq./MWh. Compared to the grid electricity, the global warming potential of AD is 43% lower (203 vs 358 kg CO₂-eq./MWh).

Across all toxicity categories and for metal depletion, AD electricity has lower impacts as there is limited use of scarce or toxic materials within the system. The materials needed to construct wind turbines and, especially, solar panels result in significantly higher metal depletion for both technologies and high toxicities for solar panel construction. The metal depletion by AD electricity is also less than a fifth of that of the UK grid, as all major constituent generation sources have a higher impact, with the exception of biomass.

However, due to the emissions from digestate application, AD has the highest marine eutrophication (2.9 kg N eq./MWh) and terrestrial acidification (30.1 kg SO₂ eq./MWh), which are two orders of magnitude greater than the impacts from the other renewable sources and the grid. The particulate matter formation (4.2 kg PM₁₀ eq./MWh) is also more than double that of coal electricity (1.8 kg PM₁₀ eq./MWh) and almost 13 times greater than the grid (0.33 kg PM₁₀ eq./MWh) due to ammonia emissions to air.

In summary, the generation of electricity from food waste via AD is more environmentally sustainable than the current grid electricity, with the exception of impacts affected by ammonia emissions related to digestate application. In terms of global warming potential, AD is not competitive with other renewable energy sources. However, it has low primary energy demand and metal depletion, meaning that the energetic and material investment to generate electricity from the food waste is relatively small compared to other generation technologies.

3.2.1. Comparison of results with other studies

No LCA studies have been found for the anaerobic digestion of food waste that used a functional unit related to electricity generation. However, several existing studies assessed the life cycle impacts of using energy crops, agricultural residues and animal slurry (Whiting and Azapagic, 2014; Fantin et al., 2015; Blengini et al., 2011; Fusi et al., 2016; Böhle et al., 2011). The system boundaries in these differ significantly from that in this study as they do not include municipal waste collection and some include the cultivation of energy crops.

Whiting and Azapagic (2014) considered AD of agricultural wastes (manure, maize silage, fodder beet and cheese whey) in the UK, with excess electricity exported to the grid and digestate used as a fertiliser. The system boundaries are similar in the two studies to allow for a rough comparison of results, albeit the feedstocks are different. As Whiting and Azapagic used the CML method, the impacts in this study have been recalculated using the same method. As can be seen in Table S5 in the SI, the agreement in global warming potential is very good, with 202 kg CO₂ eq./MWh estimated in this study compared to 222 kg CO₂ eq./MWh in Whiting and Azapagic (2014). However, the transportation of waste results in higher abiotic depletion of fossil fuels and elements, ozone depletion and photochemical oxidants creation. The difference is particularly high for ozone depletion (16.1 vs 0.1 mg CFC-11 eq.) but that is due to different Ecoinvent models used in the two studies for emissions from waste-collection trucks. Furthermore, Whiting and Azapagic (2014) assumed the emissions from the application of digestate to be equivalent to those of mineral fertilisers and therefore have significantly lower impacts for acidification and eutrophication.

Fantin et al. (2015) considered an agricultural AD which treats energy crops, slurry and crop residues; however, results are reported in a way that allows the cultivation impacts to be excluded from the

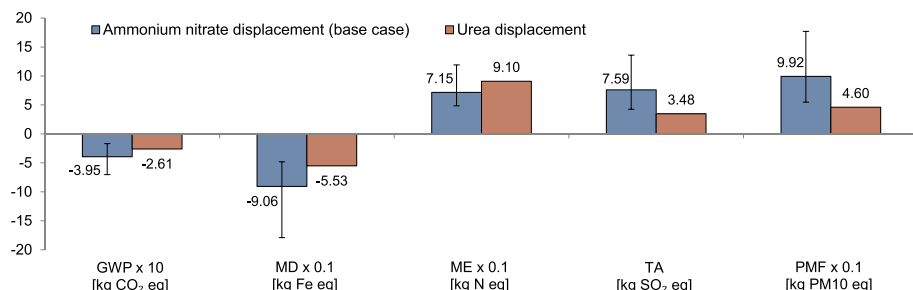


Fig. 10. Effect of the digestate displacing urea instead of ammonium nitrate on the impacts of anaerobic digestion. (All impacts expressed per tonne of food waste treated. The values on top of the graph bars represent the mean values and the error bars represent impact ranges (90th and 10th percentile) for the base case. Some impacts have been scaled – to obtain the original value, multiply with the factor shown on x-axis where relevant. For impacts nomenclature, see Fig. 4.)

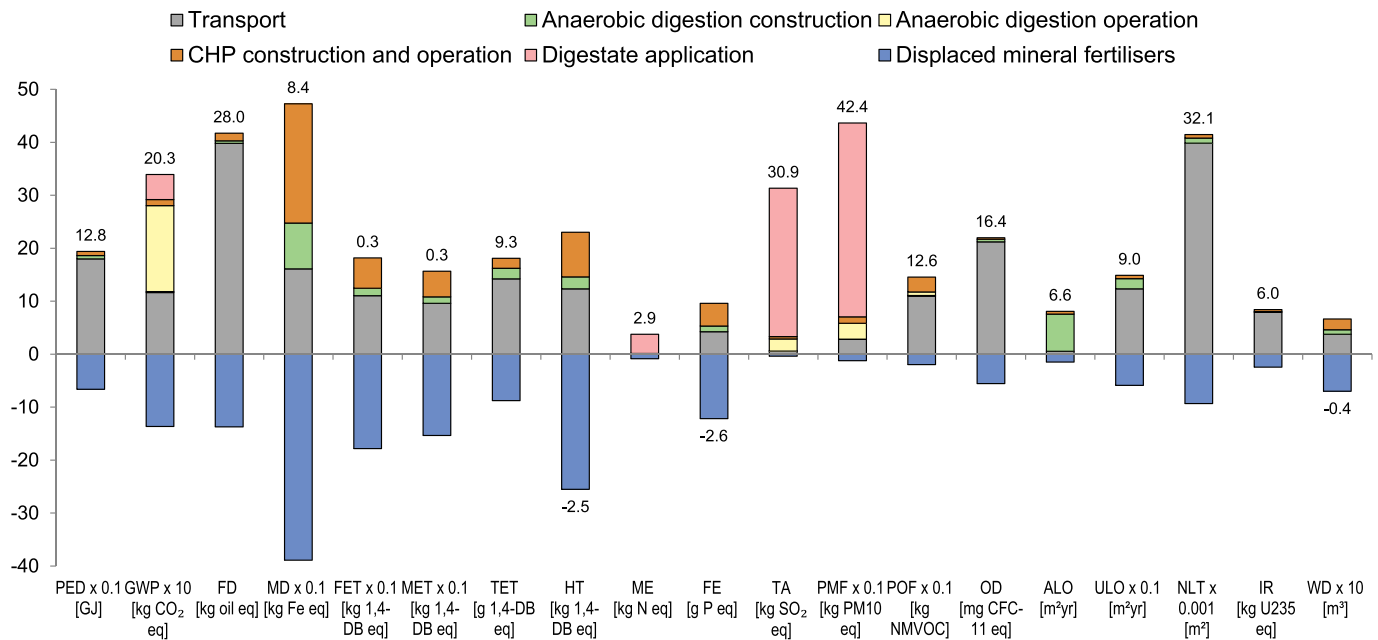


Fig. 11. Environmental impacts of electricity from anaerobic digestion of food waste and contribution of different life cycle stages. (The results shown for the base case. All impacts expressed per 1 MWh of electricity generated. Some impacts have been scaled – to obtain the original value, multiply with the factor shown on x-axis where relevant. For impacts nomenclature, see Fig. 4.)

results. As the authors used the International Reference Life Cycle Data System (ILCD) method (Jungbluth et al., 2007), the results from this study have been converted using this method, which can be seen in Table S6 in the SI. Fantin et al. (2015) modelled the emissions from digestate application using a combination of theory and experimental data and found similar values as this study for marine eutrophication, acidification, photochemical oxidants creation. The values for global warming potential are also very similar (202 kg vs 195 kg CO₂ eq./MWh). Therefore, despite the different feedstocks, there is good agreement with the literature.

3.2.2. Uncertainty analysis

The results of the uncertainty analysis for the functional unit of 1 MWh electricity can be found in Fig. S2 and Table S4 in the SI. As explained in Section 3.1.13, the Monte Carlo mean values differ from the base case due to the uniform distribution used in the uncertainty analysis. The uncertainty is greater for this functional unit as many of the impacts, particularly metal depletion (coefficient of variation (CV = 13.1)), water depletion (CV = 1.48) and human toxicity (CV = 1.42), are more reliant on the fertiliser displacement (see Fig. 11). The latter has wide inventory ranges due to the natural variability in digestate nutrient content and performance in agricultural application. Transportation also becomes more influential for this functional unit and the wide inventory range for this adds to the uncertainty.

3.3. Impacts at the UK level

Based on the results discussed in the previous sections, this section considers the impacts of anaerobic digestion of food waste at the UK level, taking into account the total amount generated in the country annually. The potential of AD treatment to contribute to national electricity generation and displacement of mineral fertilisers is also considered.

The estimated total household food waste collected by local authorities in 2015 was 4.9 million tonnes (Quested and Parry, 2017) of which almost 15% (0.72 Mt) was collected as a separate waste stream. Fig. 13 presents selected impacts of treating the total amount of food

waste via AD in comparison to incineration, landfilling and UK grid electricity; for all impacts see, Table S7 in the SI. These results have been obtained by scaling up the impacts for the functional unit of 1 tonne to the total annual amount of food waste of 4.9 Mt. It is assumed that all food waste would be treated at waste facilities within each local authority.

If all UK household food waste was treated via AD, it could generate 1.24 TWh/yr of electricity or 0.37% of the national electricity demand. This would result in net-negative GHG emissions, saving 190,000 t CO₂-eq. per year by displacing the equivalent amount of UK grid electricity (Fig. 13). Although the electricity potential is limited, it would still supply the annual demand of 401,500 households (Table 7). It also offers greater GHG savings than incineration (50,000 t CO₂-eq./yr, see Fig. 13), which would also generate less electricity (0.82 TWh, supplying 264,400 households). Landfilling the same amount of waste, on the other hand, would increase the GHG emissions to 960,000 t CO₂-eq./yr and would satisfy the electricity demand for only 96,500 households per annum (0.3 TWh). If only the food waste that is collected separately is considered for AD treatment (0.72 Mt), the potential for electricity generation is much smaller (0.18 TWh) but it would nevertheless supply the annual demand of 59,000 households, with the net savings of 28,500 t CO₂-eq./yr.

Looking at the results from the electricity generation point of view (rather than the amount of waste treated), gives the same net GHG savings of 190,000 t CO₂-eq. for the AD relative to the grid electricity. This can be inferred from Fig. 14, which shows that, if the functional unit is the generation of 1.24 TWh (instead of 4.9 Mt of waste treated), the net emissions from AD are now positive (250,000 t CO₂-eq.). However, given the emissions from the grid of 440,000 t CO₂-eq., the difference of 190,000 t is equal to the net savings estimated for the first functional unit.

While increasing the amount of food waste treated by AD is desirable for electricity generation, it would lead to an increase in the amount of digestate produced. At present, more than half of digestate in the UK is either given away free of charge or a fee is paid for its removal (Scholes and Areikin, 2014). However, this could change if digestate could be valorised as a replacement for mineral fertilisers in agriculture. The total potential for this valorisation is presented in Table 8.

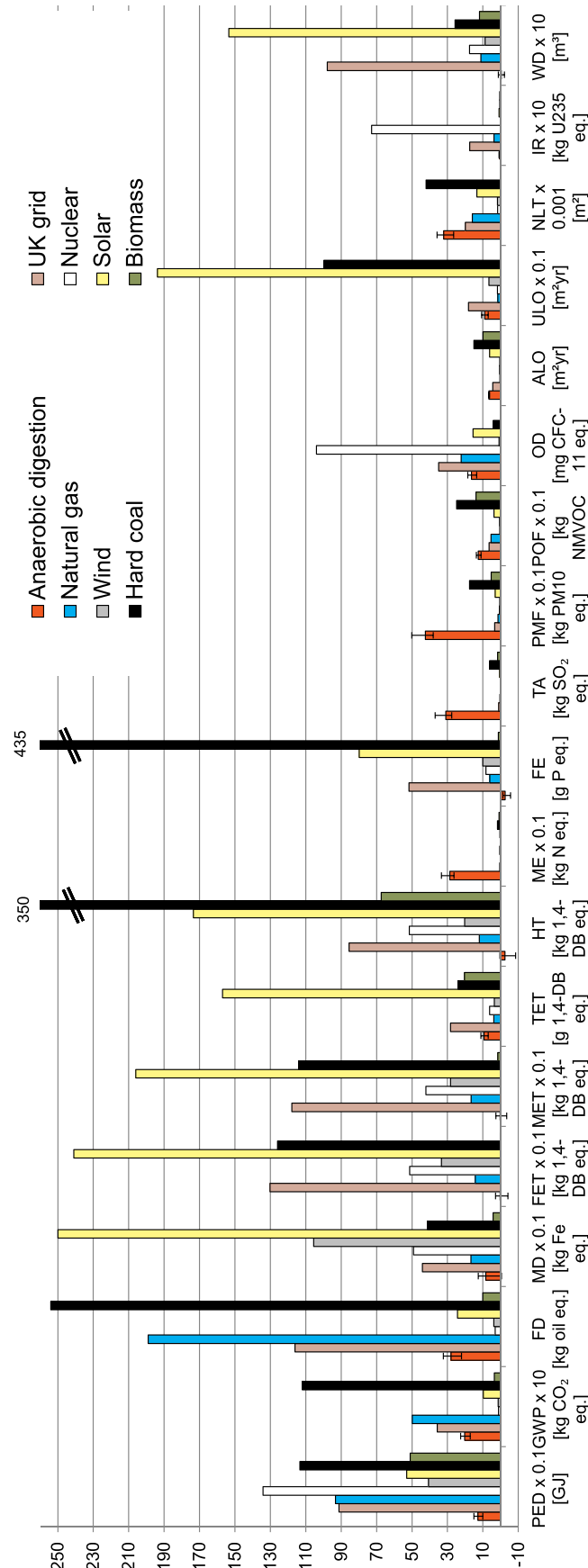


Fig. 12. Environmental impacts of electricity from anaerobic digestion of food waste compared to the UK grid electricity and electricity sources in the mix. (All impacts expressed per 1 MWh electricity generated. Data for electricity sources are from Ecoinvent. Some impacts have been scaled – to obtain the original value, multiply with the factor shown on x-axis where relevant. For impacts nomenclature, see Fig. 4.)

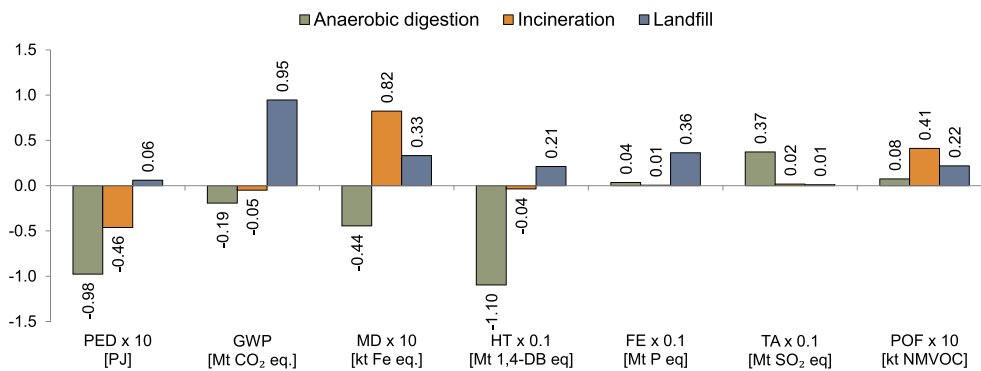


Fig. 13. Selective environmental impacts for the treatment of household food waste collected annually by local authorities in the UK. (Annual impacts from treating 4.9 Mt/yr food waste. The system has been credited for electricity and fertiliser production. Some impacts have been scaled – to obtain the original value, multiply with the factor shown on x-axis where relevant. For impacts nomenclature see Fig. 4.)

Table 7

Annual potential for electricity generation from UK household food waste treated via anaerobic digestion (base case data).

Parameter	Unit	Current collection rate	Total household food waste in municipal solid waste
Electricity exported	MWh/t	0.254	0.254
Mass of food waste	Mt/yr	0.72	4.90
Generation potential	TWh/yr	0.18	1.24
Share of total UK generation ^a	%	0.05	0.37
Share of UK domestic demand ^b	%	0.17	1.15
Equivalent number of homes ^c	–	59,000	401,500
GHG saving compared to UK grid	t CO ₂ -eq./yr	28,500	190,000
GHG saving compared to coal power	t CO ₂ -eq./yr	170,000	1,130,000

^a Total UK electricity generation in 2016: 339 TWh/yr (BEIS, 2017b).

^b The UK domestic electricity use in 2016: 108 TWh/yr (BEIS, 2017a).

^c Average electricity consumption by the UK households: 3100 kWh/yr per household (Ofgem, 2015).

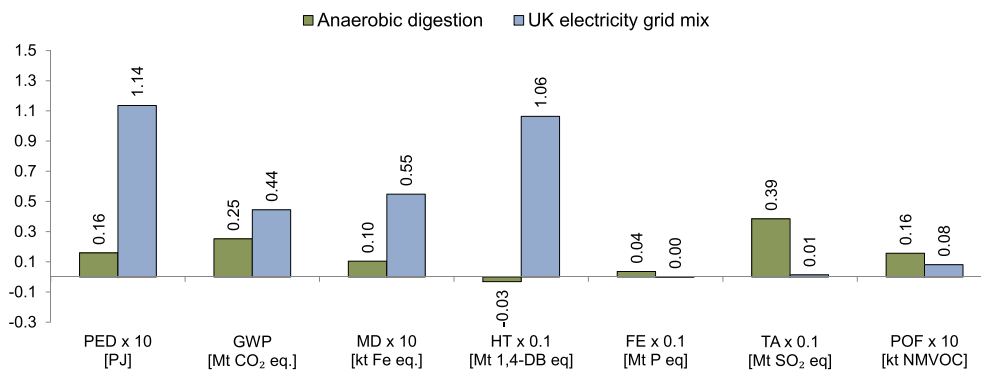


Fig. 14. Selective environmental impacts for generation of electricity from the food waste collected annually by local authorities in the UK, compared to grid electricity. (Annual impacts from 1.24 TWh/yr electricity generated from 4.9 Mt of food waste treated. The system has been credited for electricity and fertiliser production. Some impacts have been scaled – to obtain the original value, multiply with the factor shown on x-axis where relevant. For impacts nomenclature see Fig. 4.)

Table 8

Annual potential for displacement of mineral fertilisers by digestate (base case data).

Parameter	Unit	Nitrogen (NH ₄ NO ₃)	Phosphate (P ₂ O ₅)	Potassium (K ₂ O)
Total use of mineral fertiliser in the UK (DEFRA, 2017)	Mt	1.05	0.20	0.27
Displaced fertilisers	kg/t	2.42	0.41	2.15
Fertilisers displaced annually with current collection rate	t/yr	1742	293	1548
Share of the UK total	%	0.17	0.15	0.57
Fertilisers displaced annually with total household food waste in municipal solid waste	t/yr	11,858	1994	10,535
Share of UK total	%	1.13	0.99	3.90

Digestate from the food waste collected as a separate waste stream could displace 0.17% of the mineral nitrogen fertiliser currently in use in the UK. With all the UK food waste treated by AD, this figure would rise to 1.13%. Similar values are observed for phosphate fertiliser. The replacement potential is almost four times higher for mineral potassium fertiliser (3.9%), due to the high content of this nutrient in food waste. However, these figures may be difficult to achieve as the digestate needs to be made available at the correct times and locations and there is significant variation in its composition depending on feedstock (Rigby and Smith, 2011).

4. Conclusions and recommendations

The treatment of household food waste via anaerobic digestion with production of electricity and fertilisers has net negative GHG emissions of – 39 kg CO₂-eq./t and saves 2 GJ/t in primary energy demand (PED). The AD system also displays net savings for human and ecotoxicity potentials as well as depletion of metals, water, fossil fuels and ozone layer. As a waste treatment technology, AD has lower environmental impacts than incineration with energy recovery across all categories except marine eutrophication (ME), particulate matter formation

(PMF), terrestrial acidification (TA) and agricultural land occupation (ALO). AD also has lower impacts than landfilling across all categories except TA, PMF, ALO and natural land transformation (NLT). This indicates that AD can bring significant environmental benefits compared to incineration and, particularly, landfilling, which should be phased out.

Against the UK grid, AD electricity has lower impacts in 13 of the 19 categories, including PED, GWP and metal depletion (MD). It has the lowest human and eco-toxicities, PED and MD of the electricity options considered. Its GHG emissions are 46% lower than for natural gas (203 vs 499 kg CO₂-eq./MWh), the leading generation method in the UK, but significantly greater than that of solar (97 kg CO₂-eq./MWh) and wind energy (14 kg CO₂-eq./MWh).

However, when compared to other renewables, its global warming, eutrophication and acidification potentials would need to be improved. The release of ammonia from the handling and application of the digestate as a fertiliser is the greatest cause of acidification and, alongside nitrate leaching, of eutrophication. The impacts deriving from digestate application vary significantly, depending on soil type, application method and weather conditions. This is particularly relevant to nitrate emissions where there is lack of scientific consensus, with the sensitivity analysis suggesting that marine eutrophication could be significantly lower under certain conditions. Best practice and regulations should be applied to minimise the potential for emissions.

Collection of food waste and distribution of the digestate contribute significantly to the environmental impacts (before applying credits for displaced electricity and fertilisers), causing 34% of GWP, 34% of MD and 75% of POF. Therefore, it is important that the transportation distances for waste collection be optimised and that digestate can be applied locally. Furthermore, as the plant produces biogas and electricity, it is possible that these could be used to fuel alternative collection vehicles. However, this would require upgrading the biogas and would lower system credits for electricity and would hence increase the impacts; these implications could be explored in future work.

If all UK household food waste was treated via AD, the biogas could cover approximately 0.37% of the national UK electricity demand and save 190,000 t CO₂-eq./yr compared to the grid electricity. At the same time, the digestate could displace 1.1% of the mineral nitrogen fertilisers in the UK. Although small, these recovered resources represent a valuable return from a largely unutilised waste stream, some of which is unavoidable, and could contribute towards a circular economy. However, the use of digestate must be carefully managed to reduce the potential to cause significant acidification and eutrophication impacts.

Considering that in-vessel composting is the other major biological treatment of food waste in the UK, future studies should assess its impacts in comparison to AD. Also, as the UK Renewable Heat Incentive rewards the injection of biogas into the grid and the export of heat from biogas combustion, this could encourage more plants to upgrade to biomethane or utilise waste. These changes should be explored through LCA, especially as heat use could significantly increase the credits to the system through the displacement of natural gas.

At the same time, efforts must continue to reduce avoidable waste as that is always going to be more sustainable than any recovery or treatment methods, however “circular” they may be.

Acknowledgements

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.jenvman.2019.02.001>.

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