Life cycle assessment and net present worth analysis of a community-based food waste treatment system

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ABSTRACT

Food waste management has been a global challenge with significant economic and environmental impacts. A community-based food waste treatment scheme for Glasgow, UK is proposed. The food waste was treated by small-scale wet, mesophilic anaerobic digestion. Biogas was combusted in a combined heat and power plant to generate heat and electricity for each community. 201.39 kWh of electricity and 246.09 kWh of thermal energy could be provided to local communities per tonne of food waste treated. A total of 52,762 tonnes of food waste were produced each year in the city. Net-present worth analysis was employed to evaluate the scheme's economic feasibility. The scheme's environmental impacts were evaluated using life cycle assessment. The entire system saved 92.27 kg $CO_{2\text{-eq.}}$ per tonne of food waste treated and had a net-present worth of £ 3.187 million with a carbon tax of 50 £ $tonne^{-1}$ and a biogas yield of 190 m^3 $tonne^{-1}$.

Keywords: Food waste; Anaerobic digestion; Renewable energy; Carbon saving; Life cycle assessment

1. INTRODUCTION

Governments all over the world identified food waste (FW) as a high priority waste steam over the past years (Defra, 2011). The European parliament identified prevention as the most important step in the waste hierarchy. However, it is impossible to avoid all FW resulting in a need to find sustainable ways to treat FW and recover energy from it (Papargyropoulou et al., 2014). This is where waste-to-energy (WtE) technologies represent a unique opportunity. Recently much interested has been shown in generating renewable energy using distributed systems to supply local communities and WtE ties in well with this (Castaldi and Themelis, 2010).

FW can be utilised by a number of different, either biological or thermochemical, waste-to-energy conversion technologies. Anaerobic digestion (AD) established itself as a valid option for the treatment of organic wastes and will most likely play a significant role in future FW treatment systems. It has the potential to aid greenhouse gas emission reduction efforts, generate decentralised renewable electricity and thermal energy, and produce fertiliser of a lower carbon impact than e.g. mineral or chemical fertilisers (Zglobisz et al., 2010).

The technology has seen a rapid growth in recent years, especially in plants treating the organic fraction of municipal solid waste (MSW) and agricultural plants. Angelonidi and Smith compared wet and dry systems over a wide range of factors, including technical performance, energy balance, and economic performance. Dry AD systems were found to offer a number of benefits, such as shorter retention times and a greater flexibility in feedstock. However, wet AD plants were shown to have a better energy balance which also resulted in an improved economic performance (Angelonidi and Smith, 2015).

The main end-product of AD is biogas, which generally consists of 55-80% CH_4 and the remainder being mostly CO_2 and trace amounts of other gases such as hydrogen, nitrogen, and water vapor (Ali et al., 2018). Digestate is a potentially valuable by-product of the AD process. The use of digestate as fertiliser has been identified as beneficial to farmland with positive effects such as reducing the necessity of plant protection products and the destruction of possible pathogens (Koszel and Lorencowicz, 2015). Dalemo and Sonnesson identified that the use of digestate from AD may result in lower global warming potential (GWP) and acidification potential (AP) in comparison to mineral fertilisers (Dalemo et al., 1998).

The potential energy in municipal waste has been identified in literature and small-scale systems represent an interesting option to harness this available energy close to its source (Di Matteo et al., 2017). However, the small-scale distributed treatment of FW or other organic wastes on a city-wide scale, in the form of a case study, has to the authors' knowledge not been considered in scientific literature. The goal of this study is to evaluate the environmental and economic impacts of a community-based distributed FW treatment scheme on local communities in Glasgow, UK. Distributed systems have received attention in the recent past and may provide a unique opportunity for the treatment of FW. They offer several benefits, including lower transport related costs and emissions, and a reduced risk of pathophoresis. The environmental impacts of the proposed system are evaluated using LCA methodology as further explained in section 2.4. Net-present worth (NPW) analysis is employed to evaluate the economic feasibility of the proposed system as detailed in section 2.5.

2. MATERIAL AND METHODS

2.1 Scheme description

Glasgow is split into 23 electoral districts called wards. The wards have a population ranging from 21,000 to 31,000 inhabitants. These wards are adopted as the local communities of this study and it is proposed that one AD plant is installed at a central location in each of the wards for the treatment of local FW.

FW is separately collected every fortnight by refuse collection trucks and transported to the closest treatment facility. The collected FW is treated using wet AD at mesophilic operating conditions which is the most commonly employed type of AD in the UK (Angelonidi and Smith, 2015). The created biogas is locally combusted in a combined heat and power (CHP) unit to generate heat and electricity. All key parameters describing the AD and CHP process may be found in Table 1. Electricity is fed into the local electricity grid, whereas heat is sold for local utilisation. Digestate, a by-product of the AD process, is transported to local farms and used as fertiliser.

Table 1. Summary of input parameters of the AD & CHP system

Input parameter	Value	Unit	Reference
Total annual feedstock input	52762	$t\ y^{-1}$	calculated
Biogas yield	105	$Nm^3\ (t\ FI\ ww)^{-1}$	(Curry and Pillay, 2012)
CH4 content	65	%	(Curry and Pillay, 2012)
CO2 content	35	%	(Curry and Pillay, 2012)
Biogas energy density	6.25	$kWh\ m^{-3}$	(Curry and Pillay, 2012)
Digestate production rate	0.5	$t\ (t\ FI\ ww)^{-1}$	(Evangelisti et al., 2014; Møller et al., 2009)
Annual operating hours	8200	h	(Renda et al., 2016)
Electrical conversion efficiency	33	%	(Pöschl et al., 2010)
Thermal conversion efficiency	50	%	(Pöschl et al., 2010)
Auxiliary electricity demand	7	%	(Pöschl et al., 2010)
Auxiliary heat demand	25	%	(Pöschl et al., 2010)

The research scope of this study is to study the feasibility of the proposed scheme from an environmental and economic viewpoint. Existing literature data is used to model the sub-processes making up the entire scheme. This will ultimately aid policymakers and investors with making informed decisions about novel green waste treatment options.

2.2 Food waste (FW) treatment using anaerobic digestion (AD)

AD describes the decomposition of biodegradable feedstocks by bacteria in an oxygen free environment. The feedstock is ultimately converted into biogas, which is mostly composed of CH_4 and CO_2, and digestate.

This study assumes local combustion of the biogas to cover the plants auxiliary electricity and heat demands. Excess electricity is fed into the national grid and excess heat is sold locally to residents.

Food waste production is adopted from one of the authors' previous study (Ascher et al., 2019). In this study a municipal solid waste production of 480 kg per capita per year is used with a FW content of 17.7%. The amount of FW available is calculated based on Glasgow's population of 621,020 in 2017. The population of the individual wards is obtained from the local city council.

A biogas yield from FW of 105 m^3 $tonne^{-1}$ wet weight (ww) is used in this work based on Curry and Pilley (Curry and Pillay, 2012). This is a rather conservative value, as e.g. Banks *et al.* (Banks et al., 2011) found yields as high as 156 m^3 $tonne^{-1}$ ww and a British wet AD plant considered in Angelonidi and Smith (Angelonidi and Smith, 2015) was quoted to have a biogas yield of 190 m^3 $tonne^{-1}$ ww. The effects of higher biogas yields will be considered by sensitivity analysis.

2.3 Waste collection and transportation modelling

Studies agree that the environmental impact of waste collection is relatively small compared to other parts of a waste treatment scheme (Ascher et al., 2019; Hupponen et al., 2015). However, they represent a major part of the cost of waste handling systems. For example, waste collection and transportation were estimated to make up 60-80 % of all costs related to the Swedish solid waste handling system (Sonesson, 1996). Hence, it is of great importance to apply an accurate model to estimate the economics and environmental effects of a given waste collection scheme.

Not many models exist in literature to model a waste collection process. The Swedish model Organic Waste Research (ORWARE) was developed in the mid-1990s to simulate the handling of organic wastes (Dalemo et al., 1997). One of its sub-models is the transport sub-model which is outlined in more detail in three papers by one of the co-creators of the model (Sonesson, 2000, 1996). The Waste Recycling and Cost Model (WRCM) is an Australian model developed in the late 1990s. The WRCM model is a generalised model which requires minimal input data due to using assumptions about input parameters such as average road speed, set-out rate of bins, truck capacity, etc. These parameters can be replaced with case data if available (Edwards et al., 2016).

MSW-collect is an alternative model predicting the energy and time requirements of a waste collection scheme. This model has been compared to the ORWARE model and WRCM model and it was found that the MSW-collect model was more accurate compared to the other two models (Edwards et al., 2016). Hence, the MSW-collect model is adopted for this study. Initially the various sub-systems were modelled using MATLAB and all required input data was collected. Then, the main model was created, and interim results were calculated. The interim results were summed and further converted into a diesel and truck time requirement per tonne of waste collected. A detailed description of the model may be found in Edwards *et al* (Edwards et al., 2016).

The results used for the LCA and NPV analysis were a diesel requirement of 10.95 l $tonne^{-1}$ ww of waste collected, a truck time requirement of 0.8275 h $tonne^{-1}$ ww, and a requirement of 25 trucks to collect all FW for the 23 wards in Glasgow. All relevant assumptions made, and input parameters used may be found in Appendix A.

2.4 Life cycle assessment (LCA)

Life cycle assessment (LCA) is a widely used method to assess the environmental impacts of a product, process or system throughout its complete life cycle. The entire LCA is conducted in accordance with ISO 14040.

In this work LCA was carried out with GaBi and MATLAB. GaBi is a designated LCA software, which was used to evaluate the environmental impact of some of the subprocesses. This was done by creating models in the software using some of the existing processes provided by the software's database. More specifically, the avoided environmental impacts from displacing electricity and heat otherwise generated by natural gas and the environmental impact of waste collection were modelled using GaBi. The impact categories considered in GaBi follow ReCiPe 1.08 Midpoint methodology. MATLAB was used for all other aspects of the LCA by using the conversion values given in Table 2. The convention that the emission of biogenic carbon has a GWP of 0 was adopted (Møller et al., 2009). To show the effect of this convention, results are always shown as GWP excluding and including biogenic CO_2.

Table 2 – LCA equivalency factors

Emissions	Equivalency factors	Source
	GWP equivalency factors relative to CO_2	
CO_2	1	(IPCC, 2016)
CO_2 (biogenic)	0	(Møller et al., 2009)
CH_4	28	(IPCC, 2016)
N_2O	265	(IPCC, 2016)
	AP equivalency factors relative to SO_2	
SO_x	1	(GHK, 2019)
NO_x	0.70	(GHK, 2019)
NH_3	0.93	(GHK, 2019)
	PMF equivalency factors relative to PM_{10}	
PM_{10}	1	(Ravina and Genon, 2015)
NO_x	0.88	(Ravina and Genon, 2015)
VOC	0.02	(Ravina and Genon, 2015)
SO_2	0.54	(Ravina and Genon, 2015)
NH_3	0.64	(Ravina and Genon, 2015)

The goal of the LCA is defined as: Evaluating a community-based distributed food waste treatment scheme regarding three different impact categories, namely global warming potential (GWP) for a 100-year time horizon, terrestrial acidification potential (AP), and particulate matter formation (PMF).

The proposed system fulfils two main purposes which are (i) the treatment of local FW and (ii) the generation of energy in the forms of electricity and heat. Hence, the functional unit (FU) was selected to be the treatment of 1 tonne ww of FW.

The system flow chart is shown in Fig. 1. All processes comprised in the system boundary are shown in the figure, as well as the process of FW generation which is excluded from the system boundary. The system's main flows are indicated.

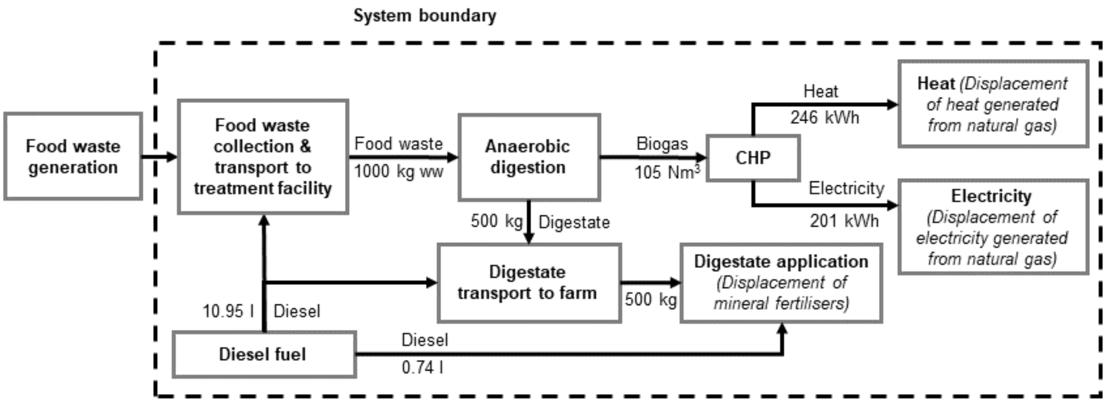

Fig. 1 System flow chart including system boundary

Table 3 summarises the emissions due to biogas utilisation in the CHP unit and the direct emissions from the AD plant in the form of fugitive emission. These emission values are further used to compute the impact on the three considered impact categories using the equivalency factors given in Table 2. Direct emissions from other processes, such as waste collection, are not shown in Table 3. This is so because their impacts on the three impact categories are directly computed by modelling the process in GaBi. Further explanations on how the environmental impacts of these processes are found are given in the subsections of section 2.4.

Table 3. Direct emissions from CHP and AD (fugitive emissions)

Emissions [g m^{-3} biogas]	Process	
	CHP	AD (fugitive emissions)
CO	2.588 [a]	-
NO_x	3.330 [a]	-
CH_4	10.463 [a]	13.026 [c]
NMVOC	2.363 [a]	-
PM_{10}	0.860 [b]	-
CO_2 (biogenic)	1838.750 [c]	19.341 [c]
N_2O	-	-

a – (Evangelisti et al., 2014)
b – (Leme et al., 2014)
c – calculated

2.4.1 Biogas utilisation in CHP unit

The total emissions resulting from the combustion of biogas in the CHP unit are shown in Table 3. Biogenic CO_2 emission are calculated, assuming complete combustion of CH_4, according to

$$CH_4+2O_2 \rightarrow CO_2+2H_2O \tag{1}$$

with the exception of a small fraction of CH_4, which is emitted unburnt (10.463 g m^{-3} biogas). Other emissions resulting from the biogas combustion, such as CO, NO_x, and particulate matter emissions, are taken from existing studies as indicated in Table 3.

2.4.2 Direct emissions from AD – fugitive emissions

Fugitive emissions (leakage) of biogas are highly variable and usually range from 0-10% of the produced biogas. Where unintentional emissions are flared this is generally closer to 0% (Bernstad and la Cour Jansen, 2012). Bernstad and la Cour Jansen reviewed various LCAs for the management of waste systems. They found that many studies did not take fugitive emission into account at all, whereas other studies found that fugitive emissions are a key parameter in assessing the GWP of an AD system

(Bernstad and la Cour Jansen, 2012). The studies that identified AD as a key parameter used fugitive emissions of 0-3%. Two other studies used values of 1% (Edwards et al., 2017) and 2% (Evangelisti et al., 2014). Based on the review of these literature sources, fugitive emissions of 3% of the produced biogas are used for this study. This is rather conservative. The emissions due to biogas leakage are summarised in Table 3.

2.4.3 Waste collection

Emissions due to waste collection are calculated using the diesel requirements per tonne of waste collected found using the model described in section 2.3. A model was created in GaBi to model the environmental impacts of the required diesel fuel.

To create the model, two inbuilt GaBi processes were used. Firstly, the process "Diesel mix at refinery" was used to model the environmental impacts of diesel for the UK. Secondly, the process "Truck - Dump Truck / 52,000 lb payload" was used to model the combustion of diesel in the waste collection vehicle. It is to be noted, that this process is US specific. However, no alternative processes were available to model diesel combustion in waste collection vehicles. The results were checked using a EURO 6 truck with an 14-20 t payload and only very minor differences were found. For this reason, it was judged that the US process "Truck - Dump Truck / 52,000 lb payload" was valid for this work and represented the real-life process most closely.

The total environmental impact of this process on the considered impact categories is shown in the results section (section 3.2). Since the process is modelled using GaBi, individual emissions (e.g. CO_2, CH_4, etc.) are not quoted, but rather the resulting equivalent impacts (e.g. $CO_{2\text{-eq.}}$).

The model only considers emissions resulting from diesel production and combustion. Hence, emissions due to tire wear, vehicle production, lubricating oil, etc. are not considered. The same approach is for example used in the ORWARE simulation model (Sonesson, 1996). Taking factors like these into account may improve the accuracy of the LCA and therefore indicate one possible area of improvement in the future.

2.4.4 Electricity and heat displacement

As previously mentioned, avoided emissions due to displacing electricity and heat otherwise generated by natural gas were modelled using GaBi. The inbuilt GaBi processes “Electricity from natural gas” and “Thermal energy from natural gas” were used. Both processes are country specific to the UK with a reference year of 2016. It is stated that the data is valid until 2021.

The electricity and heat displaced were calculated from the data shown in Table 1. Initially, the energy of the biogas produced from one tonne of FW was calculated using the biogas yield and energy density of the biogas. The total amount of electricity and heat ready for sale was calculated based on the CHP conversion efficiencies and auxiliary demands of the system. It was found that, 201.4 kWh of electricity and 246.1 kWh of heat are displaced for every tonne of FW treated. The effect of 177 kWh of waste heat emitted to the environment for every tonne of FW treated is also considered in this process.

2.4.5 Use of digestate as fertiliser

Digestate application and its use as fertiliser affects the LCA results in a number of ways. Firstly, there are emissions related to the diesel requirements for the transport and spreading of digestate. In this work a similar approach to the one described in Berglund and Börjesson was adopted (Berglund and Börjesson, 2006). Initially, a truck transports the digestate to the end-user e.g. a nearby farm. A truck with a capacity of 16 t and an average distance from AD plant to farm of 20 km were used. The spreading of the digestate was modelled using a tractor with a load capacity of 15 t and an average transport distance of 2 km from farm to field. Energy requirements for digestate transport from AD plant to farm including the empty return trip, loading, transport from farm to field, and spreading are 1.6 MJ tonne^{-1} km^{-1}, 2.5 MJ tonne^{-1}, 5 MJ tonne^{-1}, 17 MJ tonne^{-1}, respectively (Berglund and Börjesson, 2006). The required diesel fuel can be calculated based on the total energy requirement for these processes and the calorific value of diesel fuel per litre. GaBi was used to find the environmental impacts of these processes. The inbuilt models “Truck, Euro 6, 20 - 26t gross weight /17.3t payload capacity” and “Universal Tractor” were used to model both forms of transport. Similar to section 2.4.3, the process “Diesel mix at refinery” was used again to model the impacts of British diesel.

Secondly, there are emissions and avoided emissions related to using digestate as a fertiliser. These are calculated using the method outlined in Møller *et al.* (Møller et al., 2009). Initially the total amount of C, N, P, and K in the final digestate were calculated using the British FW composition reported by Tampio *et al.* (Tampio et al., 2015). A total solids (TS) content of FW of 24.86 g kgFW^{-1} was used in combination with 469.1 g kgTS^{-1} of C, 37.0 g kgTS^{-1} of N, 3.8 g kgTS^{-1} of P, and 11.4 g kgTS^{-1} of K for FW. Hence this results in the following C, N, P, and K contents in the digestate resulting from the treatment of 1 tonne of FW: 55.57 kg of C, 9.20 kg of N, 0.94 kg of P, and 2.83 kg of K. For this calculation it was assumed that the nutrient content of the digestate is the same as the one in the feed (i.e. FW), since nutrients are not lost during

AD. The potential loss of nutrients due to storage or post-processing of the digestate was not considered. The loss of carbon in the form of biogas during the AD process was accounted for, and the final amount of carbon was found to be 55.57 kg C $tonne^{-1}$ ww received at the facility. This is equivalent to 203.8 kg biogenic CO_2 $tonne^{-1}$ ww received at the facility. Avoided emissions due to carbon storage were calculated using an emission factor of 0.09 of the digestates carbon content. This resulted in a GWP of -8.34 kg $CO_{2\text{-eq.}}$ $tonne^{-1}$ ww.

N_2O emissions were calculated from the digestate's nitrogen content and a factor for N_2O–N of 0.015 of the N applied to the soil resulting in a total of 0.2168 kg N_2O $tonne^{-1}$ ww (Møller et al., 2009). N_2O emissions were further converted to CO_2-eq. emissions in the LCA using the factors given in Table 2.

Thirdly, there are avoided emissions by displacing N, P, and K fertiliser with the produced digestate. According to Møller *et al.*, the production of N fertiliser has an emission factor of 8.9 kg $CO_{2\text{-eq.}}$ kg^{-1} N, the production of P fertiliser has an emission factor of 1.8 $CO_{2\text{-eq.}}$ kg^{-1} P, and K fertiliser has an emission factor of 0.96 kg $CO_{2\text{-eq.}}$ kg^{-1} K (Møller et al., 2009). Hence, avoided emissions can be calculated from the N, P, and K content in the final digestate. Thus, the total GWP of displacing mineral fertiliser, in the form of N, P, and K fertiliser, was found to be -86.28 kg $CO_{2\text{-eq.}}$ $tonne^{-1}$ ww. Ultimately, the impacts of all three different components related to digestate use as a fertiliser were summed up to be used in the LCA.

2.5 Net present worth (NPW) analysis

The net present worth (NPW) method is an economic technique used to evaluate the economic desirability of a project. All cash flows of a project are examined over a chosen time period and resolved to their equivalent present date cash flow. Costs/expenses are taken as negative cash flows and revenues/incomes are taken as positive cash flows. A given project is regarded to be profitable for positive NPW values.

For this work the following expenses and revenues were considered: (1) the capital cost (CAPEX) is the initial investment cost of constructing the treatment plants; (2) the operation and maintenance (O&M) cost is the sum of all the costs required for the running of the plants; (3) the collection and transport (C&T) cost is the cost resulting from collecting all FW and transporting it to a nearby treatment facility; the revenues from the sale of (4) electricity (ES), (5) heat (HS), and (6) digestate (DS), are the revenues from selling the generated energy/digestate; (7) gate fees (GF) denote revenue due to the disposing of the waste; (8) revenues due to a carbon tax (CT) are considered as an additional potential source of income. Further explanation on each of these elements is given in the subsections of section 2.5. Based on this, the projects NPW is given by

$$\begin{aligned} \mathrm{NPW} = {} & \mathrm{CAPEX} + \mathrm{PW}(\mathrm{O\&M}) + \mathrm{PW}(\mathrm{C\&T}) \\ & - \big(\mathrm{PW}(\mathrm{ES}) + \mathrm{PW}(\mathrm{HS}) + \mathrm{PW}(\mathrm{DS}) + \mathrm{PW}(\mathrm{GF}) + \mathrm{PW}(\mathrm{CT})\big) \end{aligned} \tag{2}$$

where PW indicates that the element is converted from its annual worth (AW) to its present worth (PW) using Eq. (3) given by

$$\mathrm{PW} = \mathrm{AW}\,\frac{(1+i)^N - 1}{i\,(1+i)^N} \tag{3}$$

where i denotes the interest rate and N denotes the study period in years, i.e. the AD plants' lifetime of 20 years (Chang and Pires, 2015; Sullivan et al., 2014). An interest rate of 6 % is used based on literature (Ascher et al., 2019; Chang and Pires, 2015). The income due to CT is included in Eq. (2). However, this is seen as only a potential source of income and is hence not included in all scenarios. It will be indicated if CT is considered.

2.5.1 Capital cost (CAPEX) and operation and maintenance (O&M) cost

Renda *et al.* quotes a CAPEX of 7500 € kW^{-1} for AD plants with a power rating up to 100 kWe (Renda et al., 2016). It is calculated, that for this work a 100 kWe plant can treat 3786 t FW y^{-1}, based on the biogas yield, methane content in the biogas, annual operating hours, and electrical efficiency as shown in Table 1. The CAPEX value is updated to the year 2018 using Chemical Engineering Plant Cost Index (CEPCI) values and Eq. (4) given by

$$\mathrm{Cost_m} = \mathrm{Cost_n} \left(\frac{\mathrm{CEPCI_m}}{\mathrm{CEPCI_n}} \right) \qquad (4)$$

where m and n represent the reference and base year, respectively. CEPCI values of 556.8 and 603.1 were used for the years 2015 (base year) and 2018 (reference year) respectively. Finally, a capital cost of £195.7 t^{-1} FW y^{-1} is obtained using a Euro to British Pound exchange rate of 0.8823 for the year 2018.

Angelonidi & Smith compared various AD based treatment options for MSW and FW (Angelonidi and Smith, 2015). The two British plants considered have capacities of 50,000 and 80,000 t FW y^{-1} with capital costs of £10,000,000 and £20,000,000 respectively. A linear relationship between the plant size and capital cost was identified for the nine plants considered, with the two British plants having a below average CAPEX. According to these numbers, the CAPEX of the reference plant of 100 kWe, which is treating 3786 t FW y^{-1}, is calculated to be £757,200 or £200.0 t^{-1} FW y^{-1}. Hence, a CAPEX of £200 t^{-1} FW y^{-1} is used for plants with a power output of up to 100 kWe. Ascher *et al.* reviewed the ratios between CAPEX and annual O&M cost of AD plants and suggested an O&M Cost to CAPEX ratio of 7 % (Ascher et al., 2019).

2.5.2 Costs related to waste collection and transport

The model discussed in section 2.3 is used to find diesel requirements and truck time requirements per tonne ww of waste collected, as well as the total number of trucks required to collect all FW in Glasgow. This work calculated the total capital cost of purchasing all required trucks, the annual O&M cost of the trucks, the costs for hiring staff to operate the trucks, and the costs of diesel use.

The total capital cost of purchasing all required trucks, was found using information obtained from the waste collection model and using cost data and other parameters obtained from literature (Groot et al., 2014; Nakou et al., 2014). It was found that 24 trucks are required to operate the entire FW collection scheme.

This was used in combination with a CAPEX of £160,000 per truck and a life cycle of 10 years per truck (Groot et al., 2014; Nakou et al., 2014). Hence the initial CAPEX for all trucks was calculated and the future value of buying another 24 trucks was discounted to current time using

$$PW = FV \frac{1}{(1+i)^N} \tag{5}$$

where FV denotes future worth, i denotes the interest rate and N denotes the trucks lifecycle (Sullivan et al., 2014). Thus, by summing these up the NPW of buying 48 trucks in total was calculated.

An annual O&M cost of £2500 per truck per year was estimated based on (Groot et al., 2014; Nakou et al., 2014). This was used in combination with the total number of trucks required to find an annual O&M cost of operating all trucks. Hence, this value was converted from AW to PW using Eq. (3).

To calculate the costs related to wages for staff operating the trucks, the variable truck time required per tonne ww of waste collected based on the waste collection model was used. This was used in combination with the assumption that three staff are required to operate one truck and each staff is paid a wage of £9 per hour. From this, the total annual staff cost is calculated, which is further converted to its PW using Eq. (3).

Finally, the diesel required per tonne ww of waste collected is used in combination with a typical diesel price of 1.30 p l^{-1} to calculate the annual diesel cost for collecting all the FW. This is again converted from AW to PW.

2.5.3 Incomes due to the sale of electricity, heat, and digestate

Following the procedure previously described in section 2.4.4, it was found that 201.40 kWh of electricity and 246.09 kWh of heat can be sold for every tonne of FW treated.

The UK government has been providing Feed-in-Tariffs (FiT) for technologies promoting renewable and low-carbon electricity generation, such as AD, from 01.04.2010 – 01.04.2019. Even though the scheme is officially closed, projects may still receive tariffs according to the government agency ofgem, which is running the FiT scheme. Hence, the electricity tariff is taken as 4.5 p kWh^{-1} for AD plants with an installed capacity of less than 250 kW. The heat tariff of 1.4 p kWh^{-1} is not dependant on the size of the system. These are the most recent FiTs (ofgem, 2019).

Further economic benefits are due to the sale of the produced digestate. A digestate price of £13.1 t^{-1} is used for this work, based on the quoted value of 15 € t^{-1} in Renda *et* *al.* (Renda et al., 2016). This is assumed to be a gross profit which already considers costs incurred due to digestate transport, distribution, etc.

2.5.4. Incomes due to gate fees

Gate fees for the disposal of FW are based on WRAP's most recent gate fees report which was published in 2018 (Dick and Scholes, 2018). It considers gate fees charged for a number of different waste treatment, recovery and disposal options – one of which is AD. A UK wide gate fee range of -£5 t^{-1} to £68 t^{-1} was identified for food waste treated by AD. This is based on a total of 62 reported gate fees. The median value was £26 t^{-1} and the most frequently occurring value was in the range of £35 t^{-1} to £40 t^{-1}. Overall, a downward trend for FW gate fees was identified over the past few years. The report points out a high regional variability in gate fees. The gate fees in Scotland and Wales were found to be more stable and significantly higher, with a median gate fee of £49 t^{-1}, compared to the ones in England. Based on these considerations a FW gate fee value of £49 t^{-1} is adopted for this study.

2.5.5 Incomes due to carbon tax

Carbon tax can represent an efficient mean to reduce carbon emissions by adding an economic incentive to more environmentally friendly practices (Zhang et al., 2016). Allan *et al.* studied the economic and environmental impact of the introduction of a carbon on Scotland (Allan et al., 2014). According to their study, the Scottish CO_2 reduction targets could be met by imposing a carbon tax of £50 per tonne of CO_2, whilst simultaneously stimulating economic activity. The potential economic impacts of introducing a carbon tax will be explored following the methodology proposed by the authors (Ascher et al., 2019). Hence, it is assumed that a carbon tax does not apply to biogenic CO_2 emissions. Additionally, it is assumed that negative CO_2 emissions (i.e.

the displacement of CO_2) can result in revenue for the party causing the avoided emissions.

Ultimately, a carbon tax of £50 per tonne of CO_2 is used for some scenarios, as indicated in sections 3.3 and 3.4.2, as proposed by Allan *et al.* (Allan et al., 2014).

3. RESULTS AND DISCUSSION

3.1 Overall generation

All 23 AD plants combined were found to generate a total of 11,426 MWh y^{-1} of electricity and 17,313 MWh y^{-1} of thermal energy. Of this 10,626 MWh of electricity and 12,984 MWh of thermal energy can be sold to the national grid and local communities each year. This would be enough to cover the electricity demand of 3189 local households, based on an average annual household electricity demand of 3332 kWh (ofgem, 2017). The heat demand of 1082 households was satisfied when using a typical domestic heat demand of 12,000 kWh y^{-1} (Wilson et al., 2013). An increased biogas yield has the potential to substantially increase the number of households that can be covered by the system. For example, for a biogas yield of 156 m^3 $tonne^{-1}$ ww, the electricity and heat demands of 4738 and 1608 households respectively could be satisfied. For a further increase in the biogas yield to 190 m^3 $tonne^{-1}$ ww, the electricity and heat demands of 5771 and 1958 households could be satisfied. It can be clearly seen that biogas yield is a key factor and linearly relates to the energy generated. Hence, it is of great importance that the AD system operates at high efficiency to maximise biogas yield. Furthermore, biogas yield greatly influences the scheme's economics as it is directly related to the electricity and heat available for sale.

In a previous study by Ascher *et al.*, the treatment of MSW using a combined system utilising AD for organics and gasification for the waste categories Paper, Cardboard, Leather-Wood-Textiles-Rubber, and Plastics was analysed (Ascher et al., 2019). Overall the system generated a substantially larger amount of energy than the system considered in this study. This is due to the additional treatment of high heating value wastes using gasification and a greater amount of annual feedstock input for the AD system by also considering garden waste. This makes the analysis of this work and the previous one not directly comparable quantitatively. Importantly, however, the previous study found AD to be the preferred option for the treatment of organic wastes.

The scheme contributes to the coverage of the energy needs of Glasgow's residents and represents a suitable mean to safely treat FW, whilst recovering energy from the waste. The energy generated by the system can cover approximately 1.1% of households' electricity demand and 0.4% of households' heat demand in Glasgow. The environmental impacts and economics are of great importance in determining the feasibility of such a system. Policymakers and investors need to know concrete details on emissions resulting from the scheme and potential costs of installing and running the system to make educated decisions. These factors will be discussed in the following two subsections (3.2 and 3.3).

3.2 Environmental results

The LCA results for the three impact categories (GWP, AP and PMF) are shown in Fig. 2. The abbreviations used for the processes shown in the figure are: CHP – emissions resulting from burning the biogas in the CHP unit; LK – emissions due to biogas leakage in the AD plant; C&T – emissions due to the collection and transport of FW; EL – Avoided emissions due to displacing electricity otherwise generated by natural gas; HT – Avoided emission due to displacing heat otherwise generated by natural gas; DI – Emissions and avoided emissions resulting from the application of digestate as a biofertiliser. The GWP was considered for two cases: Fig. 2 (a) excludes the emissions of biogenic CO_2 (i.e. the equivalency factor of biogenic CO_2 is set to zero); Fig. 2 (b) includes the emissions of biogenic CO_2 (i.e. the equivalency factor of biogenic CO_2 is set to 1). All equivalent emission values given in this section and throughout the report are per FU (i.e. per tonne of FW treated) unless otherwise stated.

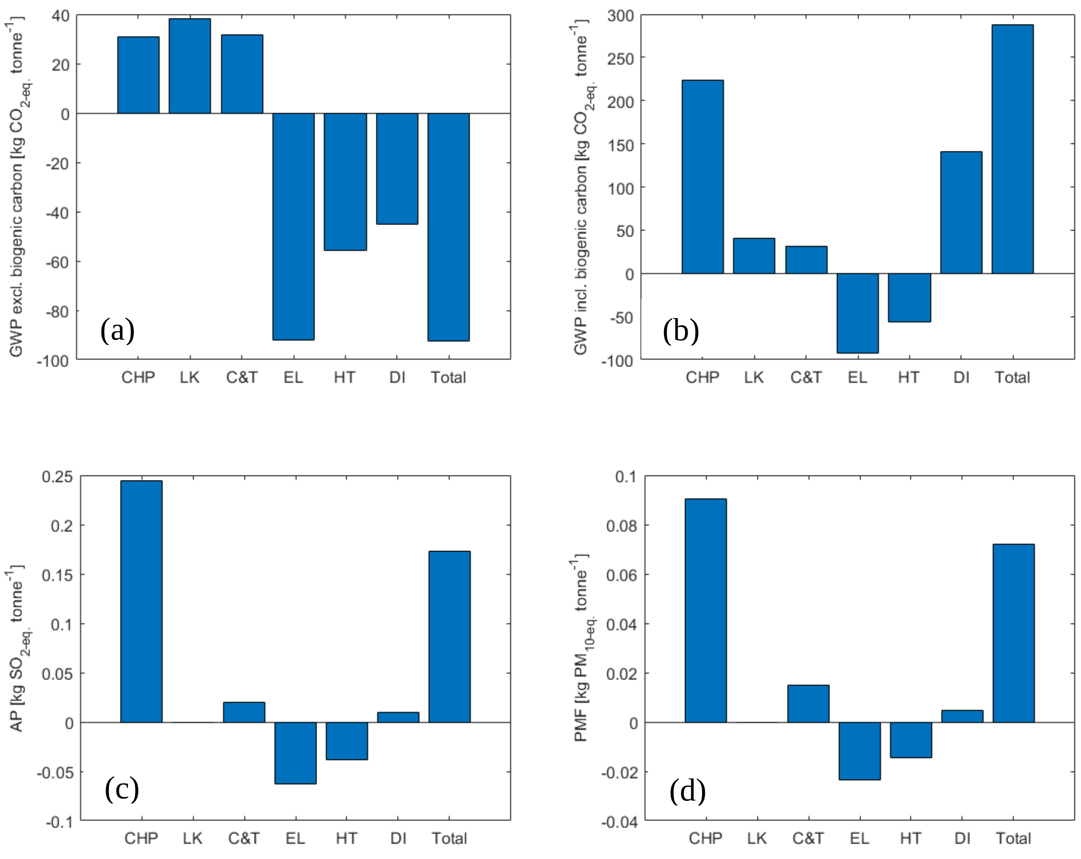

Fig. 2 LCA results for (a) GWP excluding biogenic carbon, (b) GWP including biogenic carbon, (c) AP, and (d) PMF.

The system's GWP excl. biogenic carbon, as shown in Fig. 2 (a), was found to be -92.27 kg $CO_{2\text{-eq.}}$. The negative value indicates avoided CO_2 emissions due to the system. This is mainly attributed to the avoided emissions resulting from electricity and heat displacement and the use of digestate as a fertiliser. Displacing electricity was the most significant negative contributor with a value of -90.00 kg $CO_{2\text{-eq.}}$. In comparison, emissions due to fugitive biogas losses (38.30 kg $CO_{2\text{-eq.}}$) resulted in the highest positive impact. However, both other positive contributors, namely emissions resulting from the CHP unit and emissions related to waste collection and transport, resulted in similar, lower emissions.

Changing the equivalency factor of biogenic CO_2 from 0 to 1, as shown in Fig. 2 (b), substantially changed the GWP impacts. The categories C&T, EL, and HT remained unchanged and LK increased insubstantially from 38.30 to 40.33 kg $CO_{2\text{-eq.}}$. Emissions due to CHP increased drastically from 30.76 to 223.80 kg $CO_{2\text{-eq.}}$. Interestingly, the emissions from DI changed from being negative (-44.93 kg $CO_{2\text{-eq.}}$) to positive (140.50 kg $CO_{2\text{-eq.}}$). Considering the impact categories GWP excl. biogenic carbon and GWP incl. biogenic carbon, it was found that three of the four contributors to DI remained unchanged. Namely emissions due to diesel needs for digestate transport and application (2.24 kg $CO_{2\text{-eq.}}$), carbon storage (-18.34 kg $CO_{2\text{-eq.}}$), and the displacement of mineral fertiliser (-86.29 kg $CO_{2\text{-eq.}}$). However, emissions due to digestate application changed from 57.51 kg $CO_{2\text{-eq.}}$ to 224.55 kg $CO_{2\text{-eq.}}$ upon including biogenic CO_2 in the analysis. This is due to the high quantity of biogenic carbon bound in the digestate, which was now considered in the analysis.

The system's impact on AP, as shown in Fig. 2 (c), was found to be 0.1732 kg $SO_{2\text{-eq.}}$. This was heavily dominated by the emissions from the category CHP (0.2448 kg $SO_{2\text{-eq.}}$). Only the categories EL (-0.0629 kg $SO_{2\text{-eq.}}$) and HT (-0.0382 kg $SO_{2\text{-eq.}}$) resulted in negative emissions; however, they were not large enough to outweigh the emissions resulting from CHP.

PMF is shown in Fig. 2 (d). Similar to the case of AP, the CHP process was the largest contributor to the overall impact. The total impact of the system on PMF was 0.0712 kg $PM_{10\text{-eq}}$. The CHP process contributed 0.0903 kg $PM_{10\text{-eq}}$., whereas C&T only contributed 0.0149 kg $PM_{10\text{-eq}}$. EL and HT resulted in negative emissions of -0.0235 and – 0.0143 kg $PM_{10\text{-eq.}}$ respectively. Again, avoided emissions due to displacing electricity and heat were not sufficient to outweigh the emissions mostly resulting from burning the biogas in the CHP unit.

As shown, the CHP process contributed significantly to all three impact categories considered. Carbon capture and storage systems may represent a valid way to reduce the CO_2 emission resulting from the CHP unit. Retrofitting existing units with post-combustion carbon capture and storage technology may be the easiest option. Advanced filtration systems may help in further cleaning the exhaust gases of the CHP unit, resulting in reduced PMF and AP impacts.

3.3 Economic results

The results of the NPW analysis are shown in Fig. 3. Fig. 3 (a) shows the baseline scenario excluding potential benefits from CT. The projects' overall NPW was found to be £ -6.645 million. Waste collection and transport represented the greatest cost factor over the system's life cycle with a total NPW of £ -20.02 million. The various components making up the C&T element are also shown in the figure. Diesel costs for operating the trucks represented the biggest cost factor with an NPW of £ 13.52 million. The CAPEX of the trucks was the second biggest cost factor within the C&T cost element with an NPW of £ 5.984 million. Staff costs and O&M costs for operating the trucks had lower NPWs of £ 0.6882 million and £ 8.612 million, respectively. The system's overall CAPEX and O&M cost were £ -10.55 million and £ -8.472 million over the system's life cycle and hence also represented significant costs. Profits due to the sale of electricity, heat, and digestate were relatively small, when compared to the cost elements, with NPW values of £ 5.485 million, £ 2.085 million, £ 3.962 million, respectively. The system's main source of profit came in the form of gate fees with an NPW of £ 29.65 million. Hence it was seen that gate fee values can greatly influence the economic feasibility of the system. Since gate fees are the systems main source of income, substantial research is necessary to accurately estimate the importance of gate fees. For example, by considering the change of gate fee prices with time, the accuracy of the study could be improved. However, very little data is currently available on this.

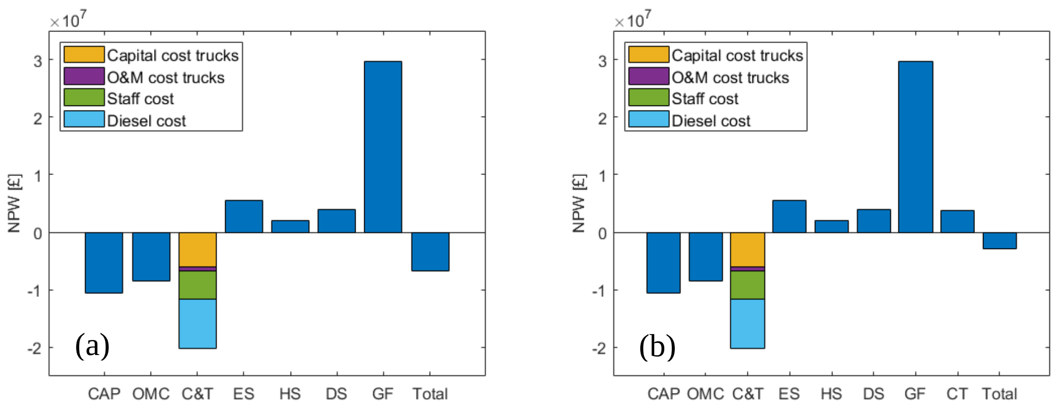

Fig. 3 NPW analysis results (a) excluding incomes due to CT, and (b) including incomes due to CT

For Fig. 3 (b) an additional source of income in the form of a CT was added. The NPW of incomes due to CT was £ 3.704 million. However, the system's total NPW did not reach the break-even level, even with this additional source of income. The new NPW value with the addition of a CT was M£ -2.941.

It is worth noting that future changes in gate fee prices, recycling behaviour, population size, and waste production per capita, etc. are not considered in this work. Such factors may significantly alter the available feedstock and hence the required size and related costs of the AD plants. Hence, recommended future works include the quantification of the potential uncertainties and impacts of these factors.

3.4 Parameter investigation

3.4.1 Environmental

The effects of some key parameters on the environmental impacts were explored. A reduced heat utilisation rate simply cut half of the environmental impact of displacing heat that is otherwise generated by natural gas. The effect of different biogas leakage rates was studied by e.g. Evangelisti *et al.* and Ascher *et al.* and was identified to have a significant impact on LCA results (Ascher et al., 2019; Evangelisti et al., 2014). However, modern AD systems should have fugitive emissions close to 0% and the baseline scenario using a rate of 3% was already conservative. Hence, the effect of altering the biogas leakage rate was not further investigated.

The results obtained in this work were compared to some of the results obtained by Evangelisti *et al.* (Evangelisti et al., 2014) who conducted a LCA of an AD plant treating 35,574 t of the organic fraction of MSW per year for London Borough of Greenwich, UK. The GWP was found to be -64.65 kg $CO_{2\text{-eq.}}$ which is comparable to the results found in this study. The CHP emissions were identified to be the key parameter when considering AP. This agrees with this study, however their system resulted in avoided $SO_{2\text{-eq.}}$ emissions of approximately 0.01687 kg $SO_{2\text{-eq.}}$. PMF was not considered in their study and hence cannot be compared. The key difference of the two studies lies in the distributed nature of this work, whereas a single plant was considered in Evangelisti *et al.* Additionally, among other factors, waste collection was not considered for their study since a comparative assessment was conducted and this process stayed identical for each alternative.

3.4.2 Economic

The impacts of some of the scheme's key parameters on the economics were also explored. One of these parameters that may vary significantly was the biogas yield, as previously stated in section 2.1. The previously assumed biogas yield of 105 m^3 $tonne^{-1}$ ww was conservative and the effect of increased biogas yields is studied. Increasing the biogas yield to 156 and 190 m^3 $tonne^{-1}$ ww resulted in NPWs of £ -2.968 million and £ -0.5166 million, respectively. Hence, even for increased biogas yields the system did not reach break-even. However, by further adding a CT break-even was reached. Upon considering a CT, positive NPWs of £ 0.7359 million and £ 3.187 million were reached for the biogas yields of 156 and 190 m^3 $tonne^{-1}$ ww, respectively. Hence, Fig 4 (a) represents the system's best-case scenario, using a biogas yield of 190 m^3 $tonne^{-1}$ ww and a CT of 50 £ $tonne^{-1}$. The biogas yield for which the system reaches break-even was 197 m^3 $tonne^{-1}$ ww when income due to CT was neglected and 145 m^3 $tonne^{-1}$ ww when a CT was considered.

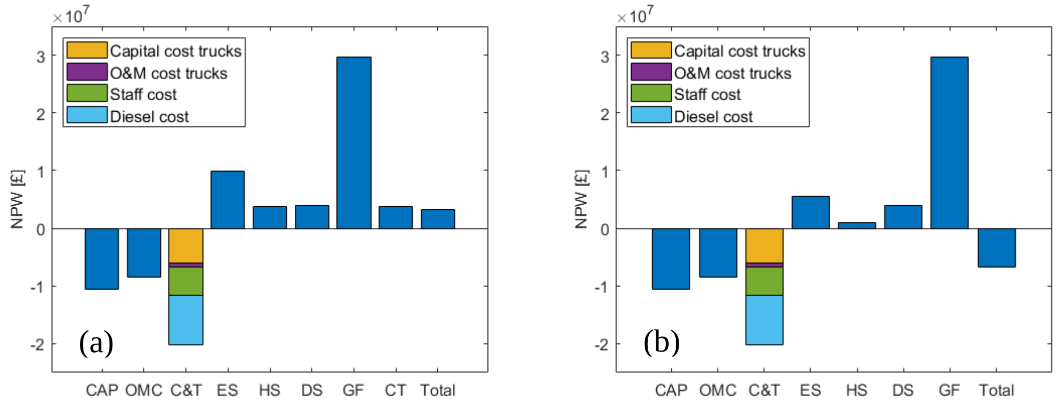

Fig. 4 Altered NPW analysis results (a) for a biogas yield of 190 m^3 $tonne^{-1}$ ww, and (b) for a heat utilisation rate of 50%

The effect of only selling half of the available, generated thermal energy was considered, i.e. a heat utilisation rate of 50%. This effectively cut the economic benefit from the heat sale in half. When incomes in the form of a CT were considered, the economic benefit was found to be £ -2.941 million. Excluding the CT income further reduced the system's NPW to £ -6.645 million. This second case is shown in Fig. 4 (b).

The economic feasibility of the scheme was highly dependent on gate fees that serve as the main source of income. Gate fees are highly variable throughout the UK with gate fees in Scotland and Wales being both higher and more stable than England (Dick and Scholes, 2018). Hence, the location of the system and the demand for FW treatment can be important factors governing gate fees. For areas with a higher demand for FW treatment and thus potentially higher gate fees, the scheme's economic feasibility may be significantly improved.

Incomes due to the sale of electricity and heat generated also played a key role in the economics of the system. The biogas yield is directly related to the amount of electricity and heat available for sale and high biogas yields substantially increase the likelihood of break-even. The government’s decision to close the FiT scheme without introducing any replacement subsidies, which are levied on energy sales, significantly increased the risks involved in investing in any AD based projects. The reintroduction of government subsidies may be necessary to incentivise the installation of small-scale renewable energy, which may otherwise struggle to become economically feasible.

4. CONCLUSIONS

The proposed distributed food waste treatment scheme showed potential to help with climate change mitigation. A total of 92.27 kg $CO_{2\text{-eq.}}$ was avoided per tonne of FW treated. The scheme was able to cover approximately 1.1% of households' electricity demand and 0.4% of households' heat demand in Glasgow.

However, there were great challenges in making the system economically viable due to high CAPEX and waste collection and transport costs. Waste collection and transport represented the largest cost element with an NPW of £ -20.02 million.

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