Township-based bioenergy systems for distributed energy supply and efficient household waste re-utilisation: Techno-economic and environmental feasibility

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ABSTRACT

Sustainable waste management and climate change have been two of the major challenges worldwide. This study designed township-based bioenergy systems to treat solid waste in Glasgow based on anaerobic digestion and gasification technologies. The economic feasibility and environmental impacts (i.e. global warming potential, eutrophication potential, and acidification potential) were evaluated using Monte Carlo simulation-based cost-benefit analysis and life cycle assessment. It was found that township-based bioenergy systems could save over 300 kg of CO_2 per tonne of municipal solid waste treated when biogenic carbon is excluded. It was shown that the proposed systems have profitability chances ranging from 68-98 %, when the sale of by-products (digestate and biochar) is considered. This study also explored the effects of by-product selling and carbon tax on the economic feasibility of township-based bioenergy systems. The township-based bioenergy system can satisfy 20-23 % of electricity demands and 4-5 % of heat demands of each township served. The study can facilitate investors and policymakers to make informed decisions about planning distributed Waste-to-Energy (WtE) systems.

KEYWORDS: Gasification; Anaerobic Digestion; Life cycle assessment; Cost-benefit analysis; Municipal solid waste; Distributed bioenergy

Glossary

AD	Anaerobic digestion	HUR	Heat utilisation rate
AP	Acidification potential	IR	Interest rate
AW	Annual worth	ISO	International Standards Organization
BCR	Benefit-cost ratio	LCA	Life cycle assessment
BL	Biogas leakage	LCI	Life cycle inventory
CAPEX	Capital cost	LCIA	Life cycle impact assessment

CEPCI	Chemical Engineering Plant Cost Index	LWTR	Leather-Wood-Textiles-Rubber
CHP	Combined heat and power	MSW	Municipal solid waste
CT	Carbon tax	O&M	Operation and maintenance
EP	Eutrophication potential	OFMSW	Organic fraction of municipal solid waste
FI	Feedstock input	PW	Present worth
FIT	Feed-In Tariff	RHI	Renewable Heat Incentive
FU	Functional unit	TS	Total Solids
FW	Food waste	VS	Volatile Solids
GWP	Global warming potential	WTE	Waste-to-energy

1. INTRODUCTION

Over the past two decades the global population has increased by over 1.5 billion, leading to ever greater energy demand and waste volume [1]. A Municipal Solid Waste (MSW) generation rate of 2.2 billion tonnes per annum is expected by 2025 worldwide [2]. In 2015, 31 % of all MSW was still landfilled in the EU and about 25 % in the UK. This represents a significant percentage. Furthermore, the UK is the largest exporter of waste in Europe, mostly shipping their waste to other European countries, India, Turkey, and China [3]. In Glasgow, UK, the council disposes of around 30 million bin collections every year. To improve the waste treatment and collection process, the first Cleansing Waste Strategy and Action Plan was implemented by the local government in 2010 [4]. Some of the government goals include that no more than 5 % of all waste can be landfilled by 2025 and that 70 % of all waste will be recycled, composted or prepared for re-use by 2025 [4]. All this clearly indicates that suitable solutions to these issues need to be found.

Significant effort has been put to design sustainable waste treatment systems based on various waste-to-energy (WTE) technologies, such as anaerobic digestion (AD) and gasification [5]. Anaerobic digestion is an attractive way for recovering energy from organic waste, whilst potentially generating a valuable by-product in the form of digestate. Digestate can be utilised as fertilizer for agricultural land application to displace mineral fertilisers [6]. Gasification can recover energy from organic and non-organic waste, making it a more versatile technology. The biochar generated from the gasification process has various potential uses, e.g. soil amendment being one of the most common ones.

The economic and environmental feasibility of AD- and gasification-based WTE systems have been extensively explored by existing studies using cost-benefit analysis (CBA) and life cycle assessment (LCA). For example, Ahamed *et al*. [7] compared three different food waste (FW) management technologies (incineration, AD, and food waste-to-energy biodiesel) for Singapore and found incineration was the least favoured option for FW treatment on environmental and economic basis. Whiting and Azapagic [8] evaluated the life cycle environmental impacts of AD plants treating agricultural wastes for combined heat and power (CHP). They found that using energy crops, such as maize, as an alternative feedstock reduced the overall global warming potential (GWP) at the cost of increasing 8 of the 11 impact categories considered. Luz *et al*. [9] evaluated the techno-economic feasibility of municipal solid waste (MSW) gasification and found that the net present value (NPV) was positive for municipalities with more than 35,000 inhabitants based on an annual rate of interest of 7.5 %.

In recent years, much research has been conducted on designing decentralised WTE systems due to their advantages over centralised systems in terms of transportation reduction and pathogen transmission alleviation. You *et al*. [10] evaluated the economic feasibility and environmental impact of a decentralised palm biomass gasification system in Indonesia and found that the electrical efficiency and capital cost both had a significant impact on the economic feasibility of the proposed systems. Patterson *et al*. [11] compared centralised and distributed biogas infrastructures. CHP with 80 % heat utilisation was found to be the most environmentally friendly alternative.

This study explores the techno-economic and environmental feasibility of decentralised waste treatment utilising AD and gasification to tackle the issues of waste pile-up and a need for renewable energy in Glasgow. This is in line with the local government goal of increasing landfill diversion as defined in the Cleansing Waste Strategy and Action Plan [4]. Monte Carlo simulation-based CBA is used to evaluate the economic feasibility. Environmental feasibility is explored using LCA with various impact categories, such as global warming potential. The novelty of this work is twofold. Firstly, the focus of this work lies on decentralised waste treatment systems utilising a combination of AD and gasification. Secondly, the feasibility of such a system is studied in terms of different sub-areas (townships) in a European city (Glasgow). This allows for a comparison of different degrees of decentralisation, as well as a comparison of different WTE technologies. Thus, the focus does not lie on comparing AD and gasification to more commonly employed waste treatment alternatives such as landfilling and incineration; but rather on finding the most suitable waste treatment system, based on AD and gasification.

2. METHODOLOGY

2.1 Waste-to-energy (WTE) technologies

2.1.1 Gasification

Gasification is a thermochemical conversion technology capable of converting solid waste to syngas (also called synthesis gas) in an oxygen-deficient environment at a temperature range of generally around 550°C to 1000° C where the oxidation is too low for stoichiometric combustion [12,13]. The syngas generally comprises of CO, H_2, CH_4, CO_2 and potentially N_2 if air is used as a gasifying agent. Moharir et al. suggests a typical H_2 content of 33.7% for syngas produced from MSW [12].

This study considers gasification using a moving grate reactor design (e.g. ENERGOS technology) [14,15]. This allows for the gasification of feedstocks with high moisture contents, such as the organic fraction of municipal solid waste (OFMSW). Additionally, moving grate gasification requires little pre-treatment, is suitable for non-uniform morphology in the feedstock, and can have a conversion efficiency of over 90 %. An important operating parameter for gasification is the equivalence ratio, which is the ratio of the oxygen content in the air supply to the value required for complete stoichiometric combustion. Moving grate gasification utilises higher ratios, of up to 0.5, than most other types of gasification. The major drawback of a moving grate reactor design is an increased capital cost and higher ash contents in the syngas, compared to e.g. a fixed bed downdraft gasifier [14]. Furthermore, this reactor design has the potential to replace existing moving grate incineration plants in the UK, which are the dominant type of incineration plants. The conversion to gasification is possible without extensive hardware modifications [16].

When it comes to gasifying agents, air generally produces a gas with a high nitrogen content and low calorific values (4 to 7 MJ/m^3) [10,14]. This is much lower than the calorific value of e.g. natural gas which is approximately 38 MJ/m^3. However, due to recent advancements in gas turbine technologies, low heating value syngas can be used effectively in a gas turbine-based CHP unit [14].

Biochar produced during the gasification process has the potential to become an environmentally and economically valuable by-product [17]. Biochar can be used for soil amendment which has a positive effect on groundwater contamination and soil fertility [18]. The soil application of biochar presents a valid strategy for climate change mitigation as it acts as a carbon sink by drawing carbon from the atmosphere. Furthermore, it represents a stable form of carbon which is released slowly [19]. However, the potential of biochar from MSW is uncertain. The utilisation for soil applications might not be suitable for biochar from MSW, due to contaminants in the feedstock [20]. A biochar yield ranging from 10 % to 20 % was frequently reported [7,21,22], and thus the biochar yield is assumed to be 15 % in this study.

Depending on the pre-treatment required and the main energy generation device used gasification can have high overall efficiencies with the potential of higher efficiencies than incineration. For example, the net electrical efficiency of gasification plants using a gas turbine lies around 20–30 % [14,23].

2.1.2 Anaerobic digestion (AD)

Anaerobic digestion describes a biological treatment method for the treatment of organic wastes.Biogas is the main gaseous end-product of AD and consists of mainly methane (CH_4) and carbon dioxide (CO_2) [24].

AD is mostly classified by their solid content, operating temperature and reactor design. Low solid content processes are called wet digestion (~ 12 % TS) and high solid content processes are called dry digestion (~ 20 % TS) [25]. Operating temperatures of approximately 35–40 °C are classified as mesophilic, whereas conditions of 55–60 °C are classified as thermophilic. Thermophilic conditions generally increase gas production and decrease operating time for organic matter degradation. However, they are less stable and require a higher heat input than mesophilic ones [26]. An increased heat input does not matter as much for cases where excess heat is not utilised or for plants in high-temperature regions. In this study however, the excess heat will be used for district heating. Dry digestion at mesophilic conditions has a high organic matter removal rate combined with a low specific growth rate of microorganisms and a small accumulation of volatile acids [26]. Hence, mesophilic AD is considered in this work.

In this work, the design is assumed to be a two-stage process with a continuously stirred tank reactor. This is comparable to the system considered by Renda *et al*. [27]. The benefit of this reactor design is that it's already commonly employed in industrial scale plants, making it a mature technology. It is suitable for high moisture waste such as the OFMSW and has good biogas yields with relatively low operational costs [28].

Digestate production is highly dependent on the feedstock composition and reactor used. A dry digestate production of approximately 700 kgt^{-1} of input was stated in Monson *et al.* [6], whereas Tan *et al.* [5] quoted a production rate of 300 kgt^{-1} of input. Based on the reasonable range, a digestate production rate of 500 kgt^{-1} of input is assumed for this work. Digestate has the potential to displace mineral fertilisers. A previous UK study showed that the heavy metal concentrations in digestate from AD complied with PAS 110 from the British Standards Institution, which made the digestate suitable for farmland application [29].

The specific energy of CH_4 is 55.6 MJ/kg, which corresponds to an energy density of 21.9 MJ/m^3 for biogas with a CH_4 content of 60 % [30]. Biogas fired CHP units offer high conversion efficiencies and thus are an attractive way of generating electricity and thermal energy. Pöschl *et al*. reported an electrical efficiency of 33 %, a thermal efficiency of 50 % and a required electricity input of 4.5 % of the electricity produced by a biogas fired CHP unit [31]. The values given apply to small-scale units, which is consistent with Walla and Schneeberger that reported electrical efficiencies ranging from 32.8-37.4 % for CHP units sized 50-500 kW_{el} [32]. Even higher electrical conversion efficiencies of 40 % have been used for larger AD plants [33].

Typical waste categories that are suitable for AD include food/kitchen waste, garden waste and other organics which can be further categorised as OFMSW. The European

Commission defined OFMSW as "biodegradable park and garden waste, food and kitchen waste from household, restaurants, caterers and retail premises and comparable waste from food processing plants" [34,35].

Existing studies compared the biogas yield from digesting the OFMSW at various conditions. The biogas yield for dry digestion at mesophilic conditions generally lies around approximately 250-500 Nm^3/t VS. Hence, a biogas yield of 350 Nm^3/t VS is assumed with a methane content of 60 %. The biogas yield is highly contingent upon the process conditions and other factors, such as the local climate. However, the value chosen is based on relevant literature and tries to estimate a realistic value for the given conditions. Nonetheless, all uncertainty in this value cannot be eliminated, but the assumed value is rather conservative and AD has the potential for better performances, especially with potential technological improvements [31,36,37].

2.2 Waste Generation

Exact data on MSW generated per capita for Glasgow itself was not available, but in a recent report by SEPA (Scottish Environment Protection Agency), a household waste generation of 216,873 t/y was reported for 2016 in Glasgow. This corresponds to approximately 349 kg per annum per capita of household waste, considering a population of 621,020 in 2017 for Glasgow [38]. However, the definition of household waste used in their study is narrower than the one used for MSW in this report. For example, public institutions like hospitals, school and prisons as well as industrial waste were not included in their study [39–41]. In a study by Evangelisti *et al*. [34], a value of 440 kg of household waste per annum per capita was used. This value was reported in 2010 for London Borough of Greenwhich. In the most recent OECD (Organisation for Economic Cooperation and Development) environment statistics report UK values of 534, 521, 491, 477, and 494 kg of

MSW per annum per capita were documented for the years 2009, 2010, 2011, 2012, and 2013 respectively [42]. This agrees with recent values reported by Eurostat. For the UK, 483 kg of MSW per annum per capita was reported for 2016 by Eurostat [43]. Thus, in this study it is assumed that 480 kg of MSW per annum per capita is generated in Glasgow, which corresponds to approximately 300,000 tonnes of MSW per annum.

The MSW waste composition is based on the data obtained from Zero Waste Scotland. This national-level study analysed the composition of MSW in Scotland. The findings are based on waste sampling of eight Scottish councils; one of which is Glasgow [44,45]. A detailed breakdown is shown in Table 1.

The OFMSW will be treated by AD and represents the combination of the categories “food/kitchen waste”, “garden waste”, and “other organics” which makes up for 31.6% of all MSW on weight basis.

The categories “Paper”, “Cardboard”, “Plastic film & dense plastic”, and “Leather-Wood-Textiles-Rubber (LWTR)” are treated by the gasification plant. The combination of these four categories sums up to 46.3% of all MSW on weight basis. The proximate and ultimate compositions, and heating values of the waste from existing literature are summarised in Table 2.

It is critical to have accurate parameters as the input of the analysis, such as biogenic and fossil carbon content, and heating values. In this study, we rely on the use of average and indicative values from existing reports and literature to make the consideration of the local waste composition as representative as possible. Unfortunately, data specific to Glasgow with a higher level of accuracy is not available at this point.

2.3 Township and scenario design

Glasgow is made up of 23 wards which act as electoral districts. For this study, several wards are clustered together to make up a township (Figure 1), and each township is allocated with a decentralised system as proposed in this study. Population data for Glasgow's wards was obtained from the local city council [46].

The feasibility of a decentralised waste treatment system is studied in terms of three different scenarios. A summary of the different three different Scenarios is shown in Table 3. For Scenario 6A&G, either three or four wards are grouped together to make up a township. The number of inhabitants per township ranges from 84,232 to 114,194 which corresponds to an MSW generation of 39,476 to 53,518 t. It is assumed that each township has both a gasification and AD plant installed to treat local MSW. The size of each system is dependent on the total amount of waste produced in the township.

Scenario 6G uses the same township arrangement as Scenario 6A&G, but it only utilises a gasification plant. Having one bigger plant instead of two smaller ones is generally more economical due to economies of scale.

Economies of scale is also the main incentive for Scenario 3A&G. For this scenario, each two townships (i.e. 1 and 2, 3 and 4, and 5 and 6) are combined to create a larger township, resulting in 3 instead of 6 townships. This results in a township size of around 200,000 inhabitants. Each of those townships utilises both AD and gasification for waste management. The townships for Scenario 3A&G are shown in Figure 1 (b).

Lists of input parameters for the AD system in scenarios 6A&G and 3A&G, and for the gasification system in scenarios 6A&G, 6G, and 3A&G are given in Table 4 and Table 5 respectively.

2.4 Life cycle assessment (LCA)

LCA is a standardised tool for evaluating the possible environmental impacts of a product, process, or system. In LCA, the environmental aspects and impacts of the product, process, or system are considered throughout its entire life cycle. It assists in identifying hot spots e.g. excess CO_2 produced, in a system's life cycle and thus shows opportunities for improvement. The LCA of waste management practices helps government and investors to identify the most preferable practice in terms of environmental impacts.

An LCA consists of four sequential phases, i.e. goal and scope definition, inventory analysis, impact assessment, and interpretation. The LCA of this work is carried out using the software “GaBi” and ReCiPe 1.08 Midpoint impact categories [47].

The goal of this study is the comparison of different MSW treatment options, and evaluating the suitability of a township-based, distributed waste treatment system based on AD and gasification for Glasgow.

A functional unit (FU) is defined to allow a comparison of different processes and scenarios. Alternatives can only be compared fairly if they all fulfil the same essential purpose or function. In the analysis of this work, the FU is taken to be 1 tonne of MSW. Specifically, the FU can be divided into three categories: (i) 316 kg of OFMSW which can be treated by either AD or gasification; (ii) 463 kg of waste most suitable for gasification; (iii) 221 kg of waste which cannot be treated by either AD or gasification. The third fraction can be assumed to be treated in the same fashion for all scenarios. As a result, the treatment of this fraction is neglected in the comparison.

A system can become over complex to include every single impact or process and thus, it is important to define suitable system boundaries. The system boundary and a basic flow chart of the different processes are shown in Figure 2. We exclude the environmental impacts related to the by-products of gasification and AD in the system boundary. Both digestate and biochar have the potential to impose a major positive environmental impact (carbon saving). However, the quality of these by-products is highly variable based on the feedstock and technology used. This generally results in different utilisation options for which additional treatment becomes necessary. All these factors lead to the exclusion of biochar and digestate from the system boundary. These by-products generally result in an overall positive environmental impact [34], making the results obtained in this study conservative.

Furthermore, the electricity and heat generated are assumed to substitute electricity and heat generated by natural gas. Currently, electricity from natural gas is the greatest contributor to the UK's national grid. In the first quarter of 2018, 31.6 % of the electricity generated in the UK came from natural gas [48].

Data is collected and processed to model relevant emissions. Input parameters to the AD and gasification conversion pathways have been summarised in Tables 4 and 5 respectively. It should be noted that the efficiency was increased from 34 % to 40 % for the larger plants used in scenarios 6G and 3A&G due to the increase in efficiency for larger plants [33,49].

Emissions from an AD-based CHP system are shown in Table 6. Complete combustion of methane is assumed for the biogas utilisation in the CHP unit. The biogas consists of 60 % CH_4 and 40 % CO_2, where the CO_2 is directly emitted to the air in addition to the CO_2 generated in the combustion process [6]. Emissions from biogas leakage are based on a 3 % biogas loss [34]. The impact of the biogas loss on the overall environmental impact is also considered in the sensitivity analysis in Section 3.3. The waste heat emissions are calculated from the electric and thermal efficiency of the process.

Emissions resulting from the energy generation by a gasification-based CHP system are shown in Table 7. The syngas production rate is assumed to be 2600 m^3/t of feedstock input based on Yao *et al*. [21]. A syngas composition of 20 % CO, 15 % CO_2, 2 % CH_4, 52 % N_2 and 11 % H_2 is assumed [10,21].

Exact distances travelled during the waste collection process and the resulting emissions are difficult to model. However, the MSW collection process itself can be assumed to be the same for each scenario. The only difference lies in the transport distance from the collection point to the treatment facility. Based on this, transport emissions are modelled using in-built GaBi processes. For the transport from the waste collection point to the treatment facility, a Euro 5 truck with a gross weight of 20-26 t, a payload of 11 t and an average transport distance of 16 km (return) is used for scenarios 6A&G and 6G. For Scenario 3A&G, the transport distance is increased to 32 km (return). A utilisation factor of 0.5 is used for all scenarios to account for the empty return trip. The transportation of the fraction of MSW which cannot be treated with gasification or AD – namely glass, metals, electronic waste, etc. – is modelled using a Euro 3 truck with a gross weight of 7.5-12 t, a payload of 5 t and an average transport distance of 50 km (return). Again, a utilisation factor of 0.5 is used.

2.5 Cost-benefit analysis (CBA)

The economic feasibility is evaluated using a Monte Carlo simulation-based CBA [50]. Monte Carlo simulation is a suitable technique to assess risks and uncertainties in an investment [51]. The Monte Carlo simulation-based CBA was conducted using MATLAB. Data from previous studies and existing literature on various cost and benefit elements is used as a reference. Triangular distributions are assumed for variable elements to account for uncertainties.

The benefit to cost ratio (BCR) is used as the economic indicator and calculated as

$$BCR = \frac{AW(\mathrm{B})}{AW(\mathrm{CAPEX}) + AW(\mathrm{O\&}M)} \qquad (1)$$

where AW denote an annual worth; B denotes the benefits of the project, CAPEX denotes the capital cost of the project (without a salvage value at the end of the lifetime), and O&M denotes the operation and maintenance cost [52]. When BCR is greater than 1 meaning the benefits outweigh the costs, the system is considered economically feasible.

AW and present worth (PW) are related by

$$AW = PW\left[\frac{i(1+i)^N}{(1+i)^N - 1}\right] \qquad (2)$$

where i denotes the effective interest rate and N denotes the study period in years, which in this case is the AD/gasification systems' lifetime (20 years) [52]. An interest rate of 6-8 % has been suggested for solid waste management in developed countries [51]. A constant interest rate of 6 % is used for all scenarios in this work.

The capital cost is based on the capacity of the plant and consists of the construction cost and the land cost. For Scenario 6A&G and 3A&G, each AD plant has a capacity of approximately 12,000-17,000 t and 28,000-34,000 t, respectively. The capital cost is based on recent findings by Renda *et al*. and includes the CHP unit [27]. Additionally, the systems are assumed to operate for 8000 hours per annum.

The values are converted from Euro to US$ using an average exchange rate of 1.0656 for 2016. The Chemical Engineering Plant Cost Index (CEPCI) is used to convert the cost from the base year 2016 to a 2017 equivalent which corresponds to the most recent CEPCI value:

$$\mathrm{Cost}_i = \mathrm{Cost}_j(CEPCI_i/CEPCI_j) \qquad (3)$$

where i and j represent the reference year (2017) and base year (2016), respectively. The value for 2017 was used since the value for 2018 was not available. The CEPCI value for 2016 was 541.7 and the value for 2017 was 567.5 [50,53]. The capital cost is strongly dependent on scale, thus further scaling was done using the power-sizing technique given by

$$\mathrm{Cost}_k = \mathrm{Cost}_i (S_k/S_i)^f \tag{4}$$

where the designed facility capacity and base capacity are denoted by S_k and S_i, respectively [50]. The scaling factor is denoted by f and is taken as 0.7 for all cases [50]. The capacity of the base facility is taken as 300 kW based on Renda *et al*. and the average designed facility capacity is taken as 1000 kW and 2000 kW for scenarios 6A&G and 3A&G respectively [27]. Finally, the calculated values are converted to US$/kW and lower and upper limits of triangular distributions are assumed. The capital cost distribution used in the Monte Carlo simulation, as well as other cost elements are summarised in Table 8.

The capital cost of the gasification system is calculated in a similar fashion as the one for AD. However, the capital cost of the gasifier and the CHP unit are calculated separately and then added. The gasifier cost is based on Basu [54]: a gasifier with a capacity of 170 tonnes of feedstock input per day has a corresponding capital cost of 25,000 US$ per tonne per day. These values are updated using Eq. (3) with a CEPCI value of 394.3 for 2001 [53]. It is assumed that each gasification system operates for 330 days per year and scaling is done using Eq. (4), where the average capacities are 20,000, 35,000, 45,000 t/y for scenarios 6A&G, 6G, and 3A&G respectively. A US$/t feedstock cost is then calculated and the upper limit, mode, and lower limit of triangular distribution for the Monte Carlo simulation are set as shown in Table 8.

The capital cost of the CHP unit for gasification is based on [49]: a 5000 kW_e CHP unit has a capital cost of 2,910,000 € in 2015. This is converted to USD using an exchange rate of 1.1097 and further updated using Eq. (3) and a CEPCI value of 556.8 for 2015 [53].

Values are further updated using Eq. (4) with an average kW_e size of 4500, 4800 and 9000 for scenarios 6A&G, 6G, and 3A&G respectively. The US$/$kW_e$ costs used in the Monte Carlo simulation can be found in Table 8 as well.

The ratio between capital cost and annual O&M cost for AD plants was calculated for four UK plants with capacities of 20,000 – 60,000 t/y [51]. This resulted in ratios ranging from 5.9 % to 8.8 %. Reference [55] considered a value of 4 % for the ratio. Hence, a triangular distribution with a lower limit, mode, and upper limit of 3 %, 7 %, and 10 % of the capital cost, respectively, is assumed for the O&M cost.

The annual O&M costs for gasification plants were reported to be approximately 17 % of the capital cost [10,56]. This is comparable to You *et al*. where a triangular distribution of 9.6 %, 16.8 %, and 24 % was used for the ratio of O&M cost and capital cost [50]. A distribution of 12 %, 17 %, and 20 % is used for this analysis.

Any income from the sale of digestate is not considered for the baseline scenarios 6A&G, 6G, and 3A&G. Monson *et al*. reviewed the potential market for digestate from MSW and suggested that the sale of digestate should be excluded from economic considerations until the market was more mature [6]. In a recent Italian study income from digestate sales was quoted at 15 €/t which is used as a reference value for altered scenarios considering the sale of digestate [27].

The economic benefit from biochar sales is, as with digestate sales, hard to quantify for the current UK market. A global average biochar price of 2650 US$/t was reported for 2016 in [57]. In comparison, the average biochar price in Australia was found to be approximately 800 US$ in 2015 [58]. However, the actual price achievable in the UK is uncertain. Thus, any biochar sales are not considered for the base scenarios. However, biochar sales are considered in a sensitivity analysis to assess their impact on the economic feasibility of the different scenarios.

In 2010, the Feed-In Tariff (FIT) scheme was implemented by the British government to promote small-scale renewable and low-carbon electricity generation. One of the technologies eligible is AD. AD plants of capacities ranging from 500 to 5000 kW receives a tariff of 1.57 p/kWh [59]. A lower limit, mode and higher limit of 1.30, 1.60, 1.70 p/kWh, respectively are assumed for the triangular distribution. Currently, the FIT scheme does not cover electricity generated from gasification. However, it is assumed that electricity generated from gasification would receive the same tariffs as AD and thus the same distribution is applied [59].

The Non-Domestic Renewable Heat Incentive (RHI) scheme currently offers tariffs for the heat generated by the combustion of biogas or syngas. Plants of a size greater than 600 kW_{th} currently receive a tariff of 1.36 p/kWh_{th}. Based on this, a distribution of 1.10, 1.40, 1.50 p/kWh_{th} is assumed [60].

It is assumed that all the heat generated (after subtracting auxiliary needs) is fed into a district heating network. The capital cost of implementing a district heating network is based on a 2011 study by Trømborg *et al*. [61]. Scenarios 6A&G and 6G have a capital cost of approximately 45 €/MWh heat exported, whereas Scenario 3A&G has a capital cost of approximately 42 €/MWh heat exported. An average Euro to US$ conversion rate of 1.39 is used for the year 2011 to convert these values, before they are further updated to current time. Additionally, a constant annual O&M cost of 1 % of the capital cost is used based on [61].

It is worth noting that the transportation logistics can significantly contribute to the costing of overall waste management processes [64], however, they are not considered in the CBA of this work for three reasons. Firstly, waste collection and transport can be run by some existing businesses that are separate from the waste treatment systems and can be paid by the waste gate fee not considered as an income of the CBA in this work. This will probably make the CBA conservative. Secondly, without accurate logistics date, additional uncertainty may be incurred if the cost of logistics is incorporated in the CBA of this work. Thirdly, this work focuses on the comparison of different technology and system options. It can be assumed that the transportation logistics remain the same across the different scenarios and would not affect the comparative assessment. Actually, existing studies [50,62,63] showed that transportation would generally have limited effect on the results of strategy studies comparing different waste treatment.

3. RESULTS AND DISCUSSION

The total electricity generated throughout all six townships of Scenario 6A&G was found to be 204,562 MWh/year. In comparison, scenarios 6G and 3A&G yielded 198,672 MWh/year and 239,301 MWh/year respectively. Households in Glasgow consumed on average 3332 kWh of electricity per year [68]. Based on this, it was calculated that each waste treatment system in Scenario 6A&G covers the annual electricity demand of on average 10,232 households located in its township. In comparison, each system in Scenario 6G provides electricity for 9938 households. Each system in Scenario 3A&G covers 23,940 of its local households. These values correspond to 20 %, 19 %, and 23 % of local households, based on average township sizes, for scenarios 6A&G, 6G and 3A&G respectively. The total thermal energy generated throughout all six townships of Scenario 6A&G was 196,136 MWh/year. Values of 149,004 MWh/year and 197,487 MWh/year were obtained for scenarios 6G and 3A&G respectively. Based on the latest typical domestic consumption values an annual heat demand of 12 MWh/year is assumed [65]. Thus approximately 5 %, 4 %, and 5 % of all households' annual thermal energy demand can be covered by scenarios 6A&G, 6G and 3A&G respectively. Similarly, to the electricity generation, Scenario 6G is outclassed by Scenario 6A&G and 3A&G in terms of heat generation.

The seasonal demand for district heating is bound to fluctuate due to higher heating demands in winter than summer. However, the heating demand in the UK is still substantial all year round, with the minimal daily heat demand, which is occurring in the summer months, being approximately one third of the maximum daily demand [66]. Furthermore, the district heating system only covers a small fraction of the annual thermal energy demand. For these reasons, the baseline scenario assumes that all the heat generated can be sold.

The increased efficiency of larger plants contributes to the increase in electricity generation of Scenario 3A&G compared to Scenario 6A&G. Additionally, the obtained results indicate that AD is more suitable than gasification for the treatment of the OFMSW when looking at energy recovery. This can be seen from the decrease in both electricity and thermal energy generated in Scenario 6G where the OFMSW is treated by AD instead of gasification.

3.1 LCA

The LCA results for the four impact categories (GWP (including biogenic CO_2), GWP (excluding biogenic CO_2), AP, terrestrial EP) are shown in Figure 3. Figure 3 (a) and (b) show that throughout the different scenarios, transportation has a minimal effect on both GWP (including biogenic CO_2) and GWP (excluding biogenic CO_2). Including biogenic CO_2 results in Scenario 3A&G emitting 539 kg $CO_{2\text{-eq}}$, which is the lowest GWP out of the three scenarios. Scenario 6A&G has a similar impact as Scenario 3A&G, whereas Scenario 6G has a substantially greater adverse effect on global warming (911 kg $CO_{2\text{-eq.}}$) than the other two scenarios when biogenic CO_2 is included. However, Scenario 6G is the most preferred alternative, when biogenic CO_2 is not considered in the analysis.

For the case of biogenic carbon being included in the GWP, the waste treatment has a detrimental effect on the environment due to positive CO_2 emissions. However, gasification and AD have been shown to result in substantially lower CO_2 emissions than, for example incineration [67]. It has also been shown, that landfilling results in substantially higher CO_2 emissions than thermal treatment methods, such as gasification [68]. Thus, gasification and AD represent a better alternative than conventional treatment methods, such as incineration and landfilling.

The overall results for the impact categories terrestrial EP and AP (Figure 3 (c) and (d)) are very similar for all three scenarios. All the scenarios obtained negative totals in these two impact categories, indicating a beneficial environmental impact. Looking at terrestrial EP, Scenario 6A&G resulted in 0.718 Mole of $N_{-eq.}$ being displaced, whereas scenarios 6G and 3A&G displaced 0.736 and 0.686 Mole of $N_{-eq.}$ respectively. The impact category AP yielded a displacement of 0.191, 0.167, and 0.190 Mole of $H+_{-eq.}$ for scenarios 6A&G, 6G, and 3A&G respectively. Negative values mainly resulted from the displacement of electricity and heat generated by natural gas. Transportation has a much greater impact on these impact categories than on GWP. Transportation contributed to a similar extent to terrestrial EP as AD and gasification. For example, transportation contributed 0.229 Mole of $N_{-eq.}$ to Scenario 6A&G, while AD and gasification contributed 0.288 and 0.215 Mole of $N_{-eq.}$ respectively. In comparison, the contribution of transportation to the impact category GWP (including biogenic CO_2) was approximately 44 times smaller than that of AD and approximately 160 times smaller than that of gasification for Scenario 6A&G.

Throughout all scenarios and impact categories the factor ED was found to have a much greater impact than HD. For example, for Scenario 6A&G ED displaced 324 kg $CO_{2-eq.}$ when looking at GWP (excluding biogenic CO_2), whereas HD displaced only 106 kg $CO_{2-eq.}$ Values of -0.266 and -0.0874 Mole of $H+_{-eq.}$ were obtained for the impact category AP for Scenario 6A&G. Similar results were found for the other two impact categories considered, as shown in Figure 3.

3.2 Cost-benefit analysis (CBA)

3.2.1 Without Carbon Tax

The CBA results of the three different baseline scenarios are presented in comparison with their altered scenarios which consider the potential benefit resulting from the sale of by-products as shown in Figure 4.

The altered scenarios are denoted by a * symbol (e.g. Scenario 6A&G*). For the altered scenarios, the lower limit, mode, and upper limit of the biochar price distribution was set to be 100, 500, 700 US$/t, which is more conservative than a recent study by You *et al*. where a triangular distribution of 0, 500, 2650 US$/t was used [10]. The biochar yield is taken to be 15% of the feedstock input on weight basis. The digestate price was set to a triangular distribution of 0, 15, 25 US$/t. A digestate production rate of 500 kg per tonne of OFMSW input was used.

None of the scenarios reaches the threshold BCR of 1 without considering the sale of any by-products, making them non-profitable. The ratios for scenarios 6A&G, 6G, and 3A&G are in the range of 0.39-0.6, 0.4-0.75 and 0.55-0.8 respectively. When considering the sale of by-products all three scenarios have the potential to be economically viable and to generate profits in the long run. Hence, the potential economic benefit of digestate and biochar sales can be substantial and even determine if an alternative is feasible.

Scenario 6A&G* represents the riskiest alternative among the three altered scenarios and has a profitability chance of 68 %. The profitability chances of scenarios 6G* and 3A&G* are 95 % and 98 % respectively. Additionally, the maximum BCRs of 3.0 and 2.2 obtained for scenarios 6G* and 3A&G* respectively are greater than the one obtained for Scenario 6A&G* which is 1.6.

3.2.2 With Carbon Tax

The implementation of a carbon tax has been long been seen as an effective mean of reducing carbon emissions [69]. In 2009 the Scottish Government introduced the Climate Change (Scotland) Act which included a target to reduce the net Scottish emissions by at least 80 % by 2050 compared to the baseline year 1990 [70]. Allan *et al*. suggested a carbon tax of £50 per tonne of CO_2 as an effective mean to meet this target [71].

It is assumed that a carbon tax may result in revenues due to negative CO_2 emissions. For the purpose of this study it is further assumed that a carbon tax excludes biogenic CO_2. Figure 5 shows a comparison of the baseline scenarios to an altered scenario using a carbon tax of £50 per tonne of CO_2. These are further compared to a scenario using a carbon tax resulting in 95±1 % of the BCRs being greater than 1, which is considered as the break-even level. CT is used as an abbreviation for carbon tax followed by the carbon tax value in £ per tonne of CO_2.

Introducing a CT of £50, as proposed by Allan *et al*., resulted in profitability chances of 0, 21, and 57 % for scenarios 6A&G, 6G, and 3A&G respectively [71]. Increasing the carbon tax of Scenario 6A&G to £140 resulted in 94 % of all BCRs being greater than 1. Scenario 6G required a CT of £90 to guarantee a break-even based on 95 % of all BCRs being greater than 1. A carbon tax of £70 was required for Scenario 3A&G.

These results demonstrate that a CT may be a sufficient incentive for a switch to green WTE technologies. Even without considering the sale of by-products scenarios 6G and 3A&G have the chance to break-even when a carbon tax of £50 was used. Scenario 6A&G has lower efficiencies and displaces less CO_2. This results in a higher CT being required to make this scenario economically feasible without considering the sale of by-products.

3.3 Sensitivity analysis

A sensitivity analysis is carried out to study the impacts of variable parameters on the results of LCA and CBA. Laurent *et al.* reviewed the sensitivity analysis of various LCA studies and found that collection and transport generally had a minor effect on the overall results [72]. Hence, collection and transport were not considered in the sensitivity analysis of LCA results.

Based on Scenario 6A&G, the sensitivity analysis was conducted by varying the heat utilisation rate (denoted by HUR50% and HUR0%), interest rate (denoted by IR3% and IR10%), and biogas leakage (denoted by BL10%). It is to be noted that the baseline scenarios assume a heat utilisation rate of 100 % which is optimistic. The abbreviations used to denote the various sensitivity analyses are summarised in Table 9. It is to be noted that an added * symbol at the end of a sensitivity analysis denotes that the sale of by-products is considered for the sensitivity analysis concerned. Furthermore, it is to be noted that not every sensitivity analysis affects both LCA and BCR results (e.g. altering the interest rate does not alter the environmental impacts of the system). Table 9 also indicates which results are affected by which sensitivity analysis.

The impacts of altering the heat utilisation rate and biogas leakage on the GWP of Scenario 6A&G are shown in Figure 6. Although the impact of a reduced heat utilisation is only shown for the impact category GWP and Scenario 6A&G, it affects all other impact categories similarly, as well as the other two scenarios. Namely, a 50 % reduction in heat utilisation (HUR50%) halves the environmental benefit related to the displacement of heat generated from natural gas. A reduction to 0 % (HUR0%) leads to none of the heat generated from natural gas being displaced which eliminates the positive impact of this contributor entirely.

Increasing the biogas leakage (BL10%) was found to increase the GWP (including biogenic CO_2) from 562 kg $CO_{2\text{-eq.}}$ to 614 kg $CO_{2\text{-eq.}}$ This represents a substantial impact on the GWP, which suggests the importance of creating an efficient system with minimal leakage. It is worth noting that AP and EP were not affected by the biogas leakage. Not utilising any of the generated heat (HUR0%) has the highest GWP impact among all the cases. Specifically, the favourable impact of Scenario 6A&G on the impact category GWP (excluding biogenic CO_2) was reduced by about one third (-320 kg $CO_{2\text{-eq}}$. to -214 kg $CO_{2\text{-eq.}}$) by sensitivity analysis HUR0%. HUR50% and BL10% impacted the GWP to a similar magnitude. They both reduced the benefit on the impact category GWP (excluding biogenic CO_2) by about one sixth or more specifically by 53 kg $CO_{2\text{-eq.}}$ and 58 kg $CO_{2\text{-eq.}}$ for HUR50% and BL10% respectively.

Impacts of heat utilisation rate and interest rate changes on the BCR of Scenario 6A&G are shown in Figure 7. The effects on the BCR results are only shown for Scenario 6A&G. However, the different factors considered in the sensitivity analysis affect all three scenarios in a similar fashion. Figure 7 (a) and (b) indicate that the economic feasibility of even Scenario 6A&G* is unlikely when the heat utilisation rate is reduced. The probability of the BCR being greater than 1 is lower than 50 %, indicating a significant risk involved with such an investment.

Decreasing the interest rate to 3% for sensitivity analysis IR3% increases the economic viability of Scenario 6A&G substantially. A BCR of greater than 1 is obtained with a likelihood of 84 % when considering biochar and digestate sales. IR10% reduces the probability of the BCR being greater than 1 to below 50 %. This will, similar to HUR50% and HUR0%, make Scenario 6A&G a risky investment.

4. CONCLUSIONS

This study examined the economic feasibility and environmental impacts of township-based bioenergy systems based on AD and gasification using Monte Carlo simulation-based CBA and LCA. It was found that all scenarios resulted in an avoidance of over 300 kg $CO_{2\text{-eq.}}$ per tonne of waste treated (excluding biogenic CO_2). Digestate and biochar can have a significant impact on the economic feasibility of a distributed bioenergy system. The BCR distribution lies under 1 for the baseline Scenarios, however upon considering the sale of by-products, Scenarios 6G* and 3A&G* stand out with profitability chances of 95 % and 98 % respectively.

The results of the sensitivity analysis indicate the importance of utilising as much of the heat generated as possible. Replacing heat otherwise generated by natural gas results in avoiding significant emissions. Heat sales represent a major source of income which strongly effects the economic feasibility of the project.

One of the aspects which is not considered in this study, is the potential environmental benefit of digestate and biochar. Both by-products show great potential in further increasing the environmental benefits of employing a waste treatment system based on AD and gasification. Furthermore, this study considered separate CHP units fired on biogas and syngas, respectively. A CHP unit capable of running on a combination of biogas and syngas might be more efficient than two separate ones.

Another relevant factor which is not considered in this study is how changes in recycling and composting, changes in the population, and changes in the waste production per capita may alter the amount of waste available for the proposed system in the future. Accounting for such factors is difficult due to uncertainties in future developments and are beyond the scope of this study.

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REFERENCES

[1] World Energy Council. World Energy Resources: 2013 survey. 2013. doi:http://www.worldenergy.org/wp-content/uploads/2013/09/Complete_WER_2013_Survey.pdf.

[2] Moyaa D, Aldás C, López G, Kaparaju P, Moya D, Aldás C, et al. Municipal solid waste as a valuable renewable energy resource. J. Energy Procedia, vol. 134, 2017, p. 286–95. doi:10.1016/j.egypro.2017.09.618.

[3] Malinauskaite J, Jouhara H, Czajczyńska D, Stanchev P, Katsou E, Rostkowski P, et al. Municipal solid waste management and waste-to-energy in the context of a circular economy and energy recycling in Europe. Energy 2017;141:2013–44. doi:10.1016/j.energy.2017.11.128.

[4] Glasgow City Council. Tackling Glasgow's Waste – Cleansing Waste Strategy and Action Plan 2015 - 2020. 2015.

[5] Tan ST, Ho WS, Hashim H, Lee CT, Taib MR, Ho CS. Energy, economic and environmental (3E) analysis of waste-to-energy (WTE) strategies for municipal solid waste (MSW) management in Malaysia. Energy Convers Manag 2015. doi:10.1016/j.enconman.2015.02.010.

[6] Monson KD, Esteves SR, Guwy AJ, Dinsdale RM. Anaerobic Digestion of Biodegradable Municipal Wastes: A Review. 2007.

[7] Ahamed A, Yin K, Ng BJH, Ren F, Chang VWC, Wang JY. Life cycle assessment of the present and proposed food waste management technologies from environmental and economic impact perspectives. J Clean Prod 2016;131:607–14. doi:10.1016/j.jclepro.2016.04.127.

[8] Whiting A, Azapagic A. Life cycle environmental impacts of generating electricity and heat from biogas produced by anaerobic digestion. Energy 2014;70:181–93. doi:10.1016/j.energy.2014.03.103.

[9] Luz FC, Rocha MH, Lora EES, Venturini OJ, Andrade RV, Leme MMV, et al. Techno-economic analysis of municipal solid waste gasification for electricity generation in Brazil. Energy Convers Manag 2015;103:321–37. doi:10.1016/j.enconman.2015.06.074.

[10] You S, Tong H, Armin-Hoiland J, Tong YW, Wang CH. Techno-economic and greenhouse gas savings assessment of decentralized biomass gasification for electrifying the rural areas of Indonesia. Appl Energy 2017;208:495–510. doi:10.1016/j.apenergy.2017.10.001.

[11] Patterson T, Esteves S, Dinsdale R, Guwy A. Life cycle assessment of biogas infrastructure options on a regional scale. Bioresour Technol 2011. doi:10.1016/j.biortech.2011.04.063.

[12] Moharir R V., Gautam P, Kumar S. Waste Treatment Processes/Technologies for Energy Recovery. Waste Treat. Process. Energy Recover., 2019.

[13] Dou B, Zhang H, Song Y, Zhao L, Jiang B, He M. Hydrogen production from the thermochemical conversion of biomass : issues and challenges. Sustain Energy Fuels 2019:314–42. doi:10.1039/c8se00535d.

[14] Arena U. Process and technological aspects of municipal solid waste gasification. A review. Waste Manag 2012. doi:10.1016/j.wasman.2011.09.025.

[15] Ramachandran S, Yao Z, You S, Massier T, Stimming U, Wang CH. Life cycle assessment of a sewage sludge and woody biomass co-gasification system. Energy 2017;137:369–76. doi:10.1016/j.energy.2017.04.139.

[16] Yang Y Bin, Sharifi VN, Swithenbank J. Converting moving-grate incineration from combustion to gasification - Numerical simulation of the burning characteristics. Waste Manag 2007. doi:10.1016/j.wasman.2006.03.014.

[17] Hansen V, Müller-Stöver D, Ahrenfeldt J, Holm JK, Henriksen UB, Hauggaard-Nielsen H. Gasification biochar as a valuable by-product for carbon sequestration and soil amendment. Biomass and Bioenergy 2015. doi:10.1016/j.biombioe.2014.10.013.

[18] Zornoza R, Moreno-Barriga F, Acosta JA, Muñoz MA, Faz A. Stability, nutrient availability and hydrophobicity of biochars derived from manure, crop residues, and municipal solid waste for their use as soil amendments. Chemosphere 2016. doi:10.1016/j.chemosphere.2015.08.046.

[19] Oliveira FR, Patel AK, Jaisi DP, Adhikari S, Lu H, Khanal SK. Environmental application of biochar: Current status and perspectives. Bioresour Technol 2017. doi:10.1016/j.biortech.2017.08.122.

[20] Shackley S, Hammond J, Gaunt J, Ibarrola R. The feasibility and costs of biochar deployment in the UK. Carbon Manag 2011;2:335–56. doi:10.4155/cmt.11.22.

[21] Yao Z, You S, Ge T, Wang CH. Biomass gasification for syngas and biochar co-production: Energy application and economic evaluation. Appl Energy 2018. doi:10.1016/j.apenergy.2017.10.077.

[22] Panepinto D, Tedesco V, Brizio E, Genon G. Environmental Performances and Energy Efficiency for MSW Gasification Treatment. Waste and Biomass Valorization 2014. doi:10.1007/s12649-014-9322-7.

[23] Arafat HA, Jijakli K. Modeling and comparative assessment of municipal solid waste gasification for energy production. Waste Manag 2013. doi:10.1016/j.wasman.2013.04.008.

[24] Walker L, Charles W, Cord-Ruwisch R. Comparison of static, in-vessel composting of MSW with thermophilic anaerobic digestion and combinations of the two processes. Bioresour Technol 2009;100:3799–807. doi:10.1016/j.biortech.2009.02.015.

[25] Forster-Carneiro T, Pérez M, Romero LI. Thermophilic anaerobic digestion of source-sorted organic fraction of municipal solid waste. Bioresour Technol 2008;99:6763–70. doi:10.1016/j.biortech.2008.01.052.

[26] Moya D, Aldás C, López G, Kaparaju P. Municipal solid waste as a valuable renewable energy resource: A worldwide opportunity of energy recovery by using

Waste-To-Energy Technologies. Energy Procedia, 2017. doi:10.1016/j.egypro.2017.09.618.

[27] Renda R, Gigli E, Cappelli A, Simoni S, Guerriero E, Romagnoli F. Economic Feasibility Study of a Small-scale Biogas Plant Using a Two-stage Process and a Fixed Bio-film Reactor for a Cost-efficient Production. Energy Procedia, 2016. doi:10.1016/j.egypro.2016.09.042.

[28] Fu X, Achu NI, Kreuger E, Björnsson L. Comparison of reactor configurations for biogas production from energy crops. Asia-Pacific Power Energy Eng Conf APPEEC 2010:1–4. doi:10.1109/APPEEC.2010.5448770.

[29] Banks C, Chesshire M, Heaven S, Arnold R, Lewis L. Biocycle anaerobic digester: Performance and benefits. Proc Inst Civ Eng Waste Resour Manag 2011;164:141–50. doi:10.1680/warm.2011.164.3.141.

[30] Dorin H, Demmin PE, Gabel DL. Prentice Hall chemistry : the study of matter. 4th ed. Needham, Mass. : Prentice Hall Inc.; 1989.

[31] Pöschl M, Ward S, Owende P. Evaluation of energy efficiency of various biogas production and utilization pathways. Appl Energy 2010. doi:10.1016/j.apenergy.2010.05.011.

[32] Walla C, Schneeberger W. The optimal size for biogas plants. Biomass and Bioenergy 2008. doi:10.1016/j.biombioe.2007.11.009.

[33] Sanscartier D, MacLean HL, Saville B. Electricity production from anaerobic digestion of household organic waste in Ontario: Techno-economic and GHG emission analyses. Environ Sci Technol 2012. doi:10.1021/es2016268.

[34] Evangelisti S, Lettieri P, Borello D, Clift R. Life cycle assessment of energy from waste via anaerobic digestion: A UK case study. Waste Manag 2014;34:226–37. doi:10.1016/j.wasman.2013.09.013.

[35] European-Comission. Communication from the Commission to the Council and the European Parliament on Future Steps in Bio-Waste Management in the European Union 2010.

[36] Davidsson Å, Gruvberger C, Christensen TH, Hansen TL, Jansen J la C. Methane yield in source-sorted organic fraction of municipal solid waste. Waste Manag 2007. doi:10.1016/j.wasman.2006.02.013.

[37] Chen X, Yan W, Sheng K, Sanati M. Comparison of high-solids to liquid anaerobic co-digestion of food waste and green waste. Bioresour Technol 2014. doi:10.1016/j.biortech.2013.12.054.

[38] Glasgow's population continues to rise 2017. https://www.glasgow.gov.uk/index.aspx?articleid=22481 (accessed July 11, 2018).

[39] SEPA. Household waste data. Scottish Environ Prot Agency 2016;3:2017–9. https://www.sepa.org.uk/environment/waste/waste-data/waste-data-reporting/household-waste-data/ (accessed July 11, 2018).

[40] SEPA. Reporting definitions and terms. Scottish Environ Prot Agency 2016. https://www.sepa.org.uk/environment/waste/waste-data/waste-data-reporting/reporting-definitions-and-terms/ (accessed July 11, 2018).

[41] SEPA. Household waste – Summary data 2016.

[42] Organisation for Economic Cooperation and Development (2015): Environmental Statistics 2015. doi:10.5257/oecd/env/2015-09.

[43] Municipal waste statistics - Statistics Explained 2016. http://ec.europa.eu/eurostat/statistics-explained/index.php/Municipal_waste_statistics#Municipal_waste_generated_by_country (accessed July 11, 2018).

[44] The Composition of Municipal Waste in Scotland | WRAP UK. Wrap 2010. http://www.wrap.org.uk/content/composition-municipal-waste-scotland (accessed July 11, 2018).

[45] Zero Waste Scotland. The composition of municipal solid waste in Scotland Executive summary 2010.

[46] Local Ward Factsheets 2015. https://www.glasgow.gov.uk/index.aspx?articleid=18820 (accessed July 11, 2018).

[47] Thinkstep. thinkstep GaBi - life cylce assessment software 2019. http://www.gabi-software.com/uk-ireland/index/ (accessed March 18, 2019).

[48] ofgem. Electricity generation mix by quarter and fuel source (GB) 2018. https://www.ofgem.gov.uk/data-portal/electricity-generation-mix-quarter-and-fuel-source-gb (accessed August 9, 2018).

[49] Obernberger I, Hammerschmid A, Forstinger M. IEA Bioenergy Task 32 project report: Techno-economic evaluation of selected decentralised CHP applications based on biomass combustion with steam turbine and ORC processes Secretariat or of its individual Member countries. 2015.

[50] You S, Wang W, Dai Y, Tong YW, Wang CH. Comparison of the co-gasification of sewage sludge and food wastes and cost-benefit analysis of gasification- and incineration-based waste treatment schemes. Bioresour Technol 2016;218:595–605. doi:10.1016/j.biortech.2016.07.017.

[51] Chang N-B, Pires A. SUSTAINABLE SOLID WASTE MANAGEMENT - A Systems Engineering Approach. John Wiley & Sons, Inc., Hoboken, New Jersey; 2015.

[52] Sullivan WG, Wicks EM, Koelling CP. Engineering Economy. 16th editi. 2014.

[53] The Chemical Engineering Plant Cost Index 2018. http://www.chemengonline.com/pci-home (accessed July 13, 2018).

[54] Basu P. Biomass Gasification, Pyrolysis and Torrefaction - Practical Design and Theory. 2nd Editio. Academic Press; 2013.

[55] Anaerobic Digestion—What Are the Economics? - Waste Advantage Magazine n.d. https://wasteadvantagemag.com/anaerobic-digestion-economics/ (accessed August 2, 2018).

[56] Eliasson A, Karstensson J, Anders K, Karlsson HT, Hulteberg C, Svensson H. Feasibility study of gasification of biomass for synthetic natural gas (SNG) production 2015.

[57] Ahmed MB, Zhou JL, Ngo HH, Guo W. Insight into biochar properties and its cost

analysis. Biomass and Bioenergy 2016. doi:10.1016/j.biombioe.2015.11.002.

[58] Wrobel-Tobiszewska A, Boersma M, Sargison J, Adams P, Jarick S. An economic analysis of biochar production using residues from Eucalypt plantations. Biomass and Bioenergy 2015;81:177–82. doi:10.1016/j.biombioe.2015.06.015.

[59] ofgem. Feed-In Tariff (FIT) rates. Ofgem 2018. https://www.ofgem.gov.uk/environmental-programmes/fit/fit-tariff-rates (accessed July 13, 2018).

[60] ofgem. Non-Domestic RHI main guidance | Ofgem n.d. https://www.ofgem.gov.uk/publications-and-updates/non-domestic-rhi-main-guidance (accessed August 3, 2018).

[61] Trømborg E, Havskjold M, Lislebø O, Rørstad PK. Projecting demand and supply of forest biomass for heating in Norway. Energy Policy 2011;39:7049–58. doi:10.1016/j.enpol.2011.08.009.

[62] Bernstad A, la Cour Jansen J. A life cycle approach to the management of household food waste - A Swedish full-scale case study. Waste Manag 2011. doi:10.1016/j.wasman.2011.02.026.

[63] Sundqvist, J. O., Baky, A., Clarlsson Reich, M., Eriksson, O., & Granath J. How Should Household Waste be Treated–Evaluation of Different Treatment Strategies. IVL, Swedish Environ Inst 2002.

[64] Hupponen M, Grönman K, Horttanainen M. How should greenhouse gas emissions be taken into account in the decision making of municipal solid waste management procurements? A case study of the South Karelia region, Finland. Waste Manag 2015;42:196–207. doi:10.1016/J.WASMAN.2015.03.040.

[65] ofgem. Typical Domestic Consumption Values 2017. https://www.ofgem.gov.uk/gas/retail-market/monitoring-data-and-statistics/typical-domestic-consumption-values (accessed January 7, 2019).

[66] Wilson IAG, Rennie AJR, Ding Y, Eames PC, Hall PJ, Kelly NJ. Historical daily gas and electrical energy flows through Great Britain's transmission networks and the decarbonisation of domestic heat. Energy Policy 2013;61:301–5. doi:10.1016/j.enpol.2013.05.110.

[67] Arafat HA, Jijakli K, Ahsan A. Environmental performance and energy recovery potential of five processes for municipal solid waste treatment. J Clean Prod 2015. doi:10.1016/j.jclepro.2013.11.071.

[68] Cleary J. Life cycle assessments of municipal solid waste management systems: A comparative analysis of selected peer-reviewed literature. Environ Int 2009. doi:10.1016/j.envint.2009.07.009.

[69] Zhang K, Wang Q, Liang QM, Chen H. A bibliometric analysis of research on carbon tax from 1989 to 2014. Renew Sustain Energy Rev 2016;58:297–310. doi:10.1016/j.rser.2015.12.089.

[70] Scottish Parliament. Climate Change (Scotland) Act 2009 2009. https://www.legislation.gov.uk/asp/2009/12/contents (accessed January 30, 2019).

[71] Allan G, Lecca P, McGregor P, Swales K. The economic and environmental impact of

a carbon tax for Scotland: A computable general equilibrium analysis. Ecol Econ 2014;100:40–50. doi:10.1016/j.ecolecon.2014.01.012.

[72] Laurent A, Clavreul J, Bernstad A, Bakas I, Niero M, Gentil E, et al. Review of LCA studies of solid waste management systems - Part II: Methodological guidance for a better practice. Waste Manag 2014. doi:10.1016/j.wasman.2013.12.004.