Economic and Environmental Assessment of Organic Waste to Biomethane Conversion

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Abstract

Biomethane and biofertilizer production by anaerobic co-digestion of organic waste serves a promising method for reducing the environmental footprint of organic waste management. This study evaluated the techno-economic feasibility and environmental impacts of organic waste-to-biomethane development in Glasgow, UK using net present value (NPV) analysis and life cycle assessment. Four different biogas upgrading technologies (pressurized water scrubbing, chemical scrubbing, membrane separation, and pressure swing adsorption) were compared. The membrane separation technology-based biomethane production meets 0.8% of the gas demand for Glasgow households with a conversion efficiency of 83%. The organic waste-to-biomethane development saved up to 264 kg $CO_{2\text{-eq}}$ annually per tonne of waste treated, with an NPV ranged between £-9.0 million and £-12.0 million based on the upgrading technology. High costs for waste collection and transportation are primarily responsible for the negative NPV. Carbon taxes between £31.30 and £58.02 per tonne of CO_2 are needed for economically viable biomethane production.

Keywords: Waste Management; Biomethane; Life Cycle Assessment; Cost Benefit Analysis; Global Warming Potential

1 Introduction

The recent uprise in organic waste generation has necessitated governments worldwide to designate organic waste (e.g., food waste (FW) and cow slurry (CS)) as a high-priority waste stream because of its significant and socio-economic and environmental implications (Banks et al., 2011). One of the most widely used organic waste treatment strategies is anaerobic digestion (AD). The process has the potential to generate value-added products such as biogas and digestate, which could be used for combined heat-power generation and soil amendment as a biofertilizer, respectively. The renewable energy derived from biogas and the biofertilizer is characterized by considerably lower carbon footprints than fossil fuel-powered energy and chemical fertilizers (Zglobisz et al., 2010). Ultimately, this contributes to a circular economy concept with the interconnection of organic waste generation and transportation, combined heat and power generation, bio-derived natural gas (i.e. biomethane) production, and eco-friendly fertilizer production (Cucchiella et al., 2019a; Robert, 2020).

The AD technology has witnessed a constantly rising deployment in agricultural farms, municipal waste treatment plants, and agricultural farms (Sun et al., 2015). There are two classes of AD plants, (a) dry and (b) wet. Despite the dry AD offering several benefits such as shorter retention time, reduced water usage, and flexibility of feedstock, they are not favourable from economic and energetic perspectives and suffer from the technical challenge of limited mass transfer. In contrast, wet AD systems have lower capital expenditure and a higher amount of annual biogas yield per tonne of waste supplied, making them favourable for the organic waste treatment (Angelonidi & Smith, 2015).

Biogas contains approximately 60-70 vol.% CH_4 (Sun et al., 2015), which can be used for biomethane production. Biomethane has several promising applications that can serve to reduce greenhouse gas emissions such as transportation fuel, domestic cooking applications, and supplying to the centralized gas grids. However, the remaining fraction of the biogas contains about 30-40 vol.% CO_2 which is undesirable for the application of biomethane. For example, as per the current government norms, the purity of CH_4 to be injected into the gas grid needs to be greater than 95 vol.% (Yousef et al., 2019), suggesting that an additional biogas upgrading stage is needed for the use of AD-derived biomethane as a displacement of natural gas in the grid.

There are four commonly used biogas upgrading technologies: (1) pressurized water scrubbing (PWS), (2) chemical scrubbing, (3) membrane separation, and (4) pressure swing adsorption (PSA), which could achieve a CH_4 purity greater than 95 vol.%. The critical

performance parameters of the four technologies are compared and listed in Appendix A. PWS is one of the most popular biogas cleaning techniques that is used to separate CH_4 using the principle of different solubility of gases in water. These systems are featured by their low input energy requirements and provide a CH_4 purity in the range of 94-98% (see Appendix A). The energy consumption in PWS is mainly used for compressing raw gas and processing water by circulation pumps. The second technique, chemical scrubbing (or chemical absorption) differs from physical absorption in the chemical reaction between absorbed substances and solvent. The chemical solvents (such as amines) are preferred over physical solvents when a minute amount of CO_2 is present. Compared to PWS, the chemical scrubbing plants have high energy consumptions (see Appendix A) that is required to regenerate the chemical solvents. Both the PWS and chemical scrubbing plants have comparable efficiencies and capital expenditure for a given plant capacity.

The membrane separation technology has several benefits over the former two techniques. It takes advantage of the permeation of smaller CO_2 molecules through a semipermeable surface under high pressure to separate CO_2 from CH_4 (Sun et al., 2015). When compared to chemical scrubbing, the membrane separation-based systems have lower capital expenditure per unit plant capacity and nearly three-fold lower energy requirements (see Appendix A). The fourth technique, PSA is not based on the solubility of the molecules but instead, on the selective adsorption to the surface of the adsorbent. The PSA technology can be used to separate CH_4 from CO_2 since the CH_4 molecule is larger than CO_2 molecules. However, the system has high CH_4 loss percentages (see Appendix A), which become dominant at high CH_4 purity requirements. Despite having a comparable capital expenditure per unit plant capacity, the PSA is often not preferred due to its lower efficiency than the competing technologies. The differences in the technical principles and performance of the upgrading methods suggest the variations of their contributions to the economics and environmental impacts of a practical waste-to-biomethane system. Hence, upon the design of a biomethane production plant, it is important to systematically compare the economic feasibility and environmental impacts of the candidates based on the different upgrading methods.

A recent survey by the Waste and Resources Action Programme (WRAP) highlighted around 9.5 million tonnes of FW was generated in the UK during 2018, valued at £19 billion, and is associated with 25 million tonnes of greenhouse emissions (Dray, 2021). Another report indicated that nearly 31.5 million tonnes of undiluted CS is produced in UK annually (Waterton et al., 2017). Hence there is an urgent need to develop strategies integrating organic waste (FW

and CS) management systems to the natural gas grids. Although there have been studies evaluating the economics and environmental impacts of organic waste-to-energy production in the UK (Ascher et al., 2019; Ascher et al., 2020), the corresponding information regarding organic waste-to-biomethane development is still limited. Moreover, to the best knowledge of the authors, an extensive comparison of the economics and environmental impacts of biomethane production through the different upgrading technologies is still missing in the literature. To address these, this work evaluates the techno-economic feasibility and environmental impacts of a community-scale organic waste (FW and CS)-to-biomethane conversion system in Glasgow, UK based on the consideration of the four different biogas upgrading technologies. The environmental impacts of the proposed system are evaluated through life cycle assessment (LCA), while the economic aspects are assessed by net present value (NPV) analysis. The results reveal important insights on various economic and environmental factors that will guide the policymakers on planning biomethane generation.

2 Methodology

2.1 Scheme design

The genesis of this work is installing two large anaerobic digestion plants, each with a biogas upgrading system in the city of Glasgow. The city's 23 wards are home to a population of 621,020 people. A waste management strategy is to be proposed for the plethora of (1) FW disposed of by these wards and (2) CS generated by the nearby farms of South Lanarkshire. At first, the two sources of wastes will undergo anaerobic co-digestion to produce biogas, which mainly consists of CH_4 and CO_2. Subsequently, the biogas will be cleaned and upgraded, making it suitable for injection into the gas grid. An inherent by-product of the plant, digestate, is sold to the nearby farms as a fertilizer. The ADs operate under wet, mesophilic conditions with a 1:1.8 ratio of FW to CS to produce an expected methane content of 60% (Banks et al., 2011). The aim is to identify the most promising biogas upgrading process that can be integrated with AD through LCA and NPV approaches.

2.2 Anaerobic co-digestion of food waste and cow slurry

Co-digestion of two feedstocks (i.e. FW and CS) is considered in this work as it has several benefits, including an improved breakdown of organic material, ultimately leading to a high rate of waste loading. According to Eurostat (Ec.europa.eu, 2020) and WRAP statistics (Dick, 2020) for the UK, a UK citizen generates 463 kg of municipal solid waste annually, out of which 18% was FW, which results in approximately 83 kg annual FW per person. This suggests

that the entire population of the city of Glasgow generates approximately 51,544 tonnes of FW annually. However, under a real-world scenario, it is impractical to assume that 100% of the FW will be collected and transported to the waste management system. Therefore, a waste collection of efficiency of 60% is adopted based on the EU target for municipal waste recycling by 2030 (EEA, 2020), resulting in 30,926 tonnes of annual FW available for the AD process.

The second source of feedstock is the CS, to be collected from the nearby South Lanarkshire farms. Livestock farms generate slurry, a mix of faeces and urine, mixed with bedding material and cleaning water. The annual slurry production in Scotland, UK varies within a wide range of 250,000 and 500,000 tonnes, ensuring a sufficient supply of CS to be fed to the two co-digesters in Glasgow. Based on the FW to CS ratio of 1:1.8 (Banks et al., 2011), it is estimated that approximately 55,667 tonnes of CS are required for the AD process. Therefore, the total annual feedstock loading on the AD plant is 86,593 tonnes.

2.3 Waste collection quantification and modelling

Waste collection and transportation can have a strong impact on the environmental and economic aspects of the proposed scheme. It has been shown that the collection and transportation of feedstocks account for a large percentage of the overall cost of waste handling systems (Ascher et al., 2020). Another work (Edwards et al., 2016) devised a mathematical model (the Municipal Solid Waste - Collect or MSW-Collect model) to accurately predict the energy and time requirements of a collection regime that incorporates the unique characteristics of different locations. As confirmed by (Kristanto & Koven, 2019), the environmental impact is relatively small concerning the rest of the waste treatment and biogas upgrading scheme. The MSW-Collect model was used to derive inputs for the LCA and NPV analysis in their study (Ascher et al., 2020). In the present work, the diesel requirements were adjusted based on the proportions of FW and CS to be collected. It is assumed that 15 FW lorries and 8 CS tankers are required. Tankers carry 3 times the mass of waste that food trucks do. The tankers are also assumed to have the same fuel efficiency as the FW trucks. Consequently, the diesel requirement for this study was 6.92 L/tonne of feedstock. A detailed description on the waste collection modelling can be found in (Ascher et al., 2020).

2.4 Life cycle assessment

LCA is a standard method for evaluating the environmental impact of a process or a system throughout its whole lifecycle. The framework provided by international standard ISO14040 to conduct LCA is adopted in this work. The LCA was implanted in the GaBi software, a

designated software that contains datasets enabling environmental impact analysis of certain elements of the waste to biomethane project. The impact category of global warming potential (GWP) was evaluated by following the CML 2001 methodology.

The carbon that is accumulated during plant growth is referred to as biogenic carbon. There is not yet a consensus on whether biogenic carbon should be included in the LCA. According to the Intergovernmental Panel on Climate Change (IPCC), biogenic carbon should not be included in the national emissions estimates. Therefore, for this study, biogenic carbon is not included in the LCA (Møller et al., 2009).

The objective of the LCA is to evaluate the GWP of biomethane production based on FW-CS co-digestion and subsequent biogas upgrading. The system is designed with two main purposes: to treat two waste streams (food and slurry); and to produce biomethane, which can be added to the national grid as a source of green energy. The functional unit (FU) selected for the LCA was the treatment of 1 tonne of the waste comprised of 360 kg FW and 640 kg CS. The assessment is based on avoided greenhouse gas (GHG) emissions, which can be compared with other methods of disposal of the two feedstocks and other methods of natural gas and mineral fertilizer production displaced by the products of the system. These emissions are measured in terms of the GWP over 100 years (GWP100) in conformance with the IPCC. These are measured in terms of carbon dioxide equivalent per tonne of total waste (kg $CO_{2\text{-eq}}$ per tonne TW).

A representative flow chart of the process including the system boundary is shown in Figure 1. The scope of this project is limited to the measurement of GHG emissions from the point of waste collection to the injection of the biomethane into the grid. The transportation and application of the biomethane and digestate from the system are not included within the system boundary.

2.4.1 Avoided feedstock emissions

To provide a complete picture of emissions and enable a fair comparison, it is necessary to measure the GHG emissions caused by alternative methods of disposal of both FW and CS. In Scotland, FW is treated in three primary ways – (a) discarded in a landfill, (b) incinerated, or (c) it is anaerobically digested. The present work assumes that the proposed scheme would replace 50% incineration and 50% landfill. Avoided FW emissions data was collected from (Moult et al., 2018), who evaluated the net GHG emissions of several disposal options for FW in the UK, including landfill (with and without methane capture), incineration (with electricity collection), and AD. From this data, the emissions for 1 tonne of FW disposed by incineration

and landfill with 70% capture and flaring was taken and adjusted for the FW portion of the FU (360 kg). Consequently, the AD system will result in an avoided GWP of 132 kg $CO_{2\text{-eq}}$ for every 360 kg of FW.

Storage of slurry is a source of various emissions, e.g., NH_3, H_2S, and GHG, including N_2O, CH_4, and CO_2. A prior work by (Baral et al., 2018) investigated the emissions created under different methods of storing manure, and reported the cumulative GHG emissions during different seasonal periods of a year. The GWP data used in this work was calculated by finding the average GHG emissions over the summer and autumn periods as 143 kg and 4 kg of $CO_{2\text{-eq}}/m^3$, respectively. Another essential quantity i.e., the amount of total solids (TS) content is obtained as 9.31% from (Banks et al., 2011). Consequently, the density (ρ) was calculated as 1,352 kg/m^3 using Eq. (1) (Wang et al., 2019). This results in an avoided GWP for 640 kg of slurry (i.e., the FU) as 34.55 kg $CO_{2\text{-eq}}$. Finally, the total avoided GWP from the two feedstocks (FW and CS) is 167.21 kg $CO_{2\text{-eq}}$ per tonne feedstock.

$$\rho = 162.3\text{TS} - 159.3 \tag{1}$$

2.4.2 Collection and transportation emissions

Emissions caused by waste collection and transportation (C&T) were calculated based on the diesel requirement per tonne of waste collected determined using the C&T model discussed in Section 2.3. The environmental impact of the diesel that the waste collection vehicles use can be modeled in the GaBi software for both the production and combustion of diesel. Two of the inbuilt GaBi processes: "EU-28: Diesel mix at refinery" and the "US: Truck - Dump Truck / 52,000 lb payload - 8b" were utilized for this purpose (Ascher et al., 2020). The first calculates the diesel production impact on the environment and the second calculates the impact of diesel combustion by the collection trucks. It is worthwhile noting that the process is US-specific. However, no such explicit process was available which is UK-specific and the US-specific process is assumed to represent the closest choice to the real-life scenario (Ascher et al., 2020).

2.4.3 Anaerobic digestion plant modelling

The energy required to heat and power the anaerobic digesters was added to the GaBi model using the inbuilt process "EU-28: Electricity grid mix". For this study, the amount of electricity required to digest 1 tonne of feedstock (640 kg of CS and 360 kg of FW) is assumed to be 0.13 kWh as reported by (Cucchiella et al., 2019b) for a biomethane plant case study in Italy. In addition to the electrical power for the AD plant, there is also a power requirement for the upgrading plant. The electricity requirement for the upgrading plant was modeled using EU-

28, while the water requirement was modeled using "DE: Drinking water mix" (for water scrubbing plant). Additionally, biogas leakage reported in the literature falls within the range of 0-10% and methane leakage of 0.40-3.28% (Kvist & Aryal, 2019). In this work, a fixed value of 3% for biogas leakage was assumed, which is a conservative estimate.

2.4.4 Emissions from digestate

The volume of digestate is assumed to be the same as the volume of the feedstock introduced to the digester, but the mass reduces to approximately 85% (Turley et al., 2016). The digestate can be sold to nearby farms as a substitute for mineral fertilizers which are often more expensive and harmful to the environment. The emissions that are avoided from not producing these mineral or synthetic fertilizers are taken from the amount of Nitrogen (N), Phosphorous (P), and Potassium (K) that can be found in the feedstock with the assumption that these nutrients are not altered through the AD process. The amount of total solids (TS) from the CS and FW is obtained from (Banks et al., 2011) and the nutrients found in each from the fertilizer manual by Department for Environment, Food and Rural Affairs (DEFRA) (DEFRA, 2010) and the work by (Tampio et al., 2015). Using these values, one tonne of feedstock contains 2.82 kg of N, 0.37 kg of P and 1.07 kg of K resulting in total avoided emissions of 26.66 kg $CO_{2\text{-eq}}$ per tonne of feedstock (see Table 1). The emissions for each of the nutrients were taken from the study by (Evangelisti et al., 2014), on waste-to-energy for a UK AD plant.

2.4.5 Upgrading plant modelling

Once the biogas is acquired from the AD process, it is sent to the upgrading plant for the next stage. The transportation of the biogas to the upgrading facility is assumed to be insignificant as it is on the same site as the digester and any leakages are assumed to be negligible. According to (Kapoor et al., 2019), the four most common upgrading technologies, i.e. PWS, chemical scrubbing, PSA, and membrane separation, comprise 92% of the systems operating in the market and are compared in this work. Essential parameters that influence the efficiency of the upgrading systems are detailed in Table 2. It is also assumed that the biomethane can be injected on site for which there are no transport costs associated with the end product. As the GaBi software does not provide the information related to the upgrading processes, the inputs and output data shown in Table 2 were entered manually.

2.4.6 Natural gas emissions

Avoided emissions are calculated by subtracting the emissions from the production of natural gas from the emissions from the production of biomethane. The GWP of the natural gas process is 0.506 kg $CO_{2\text{-eq}}$/kWh according to the United States Department of Energy (Skone et al.,

2016). This value represents the life cycle emissions for an average natural gas power plant. When converted for the FU, the value is between 150 and 164 kg $CO_{2\text{-eq}}$ per tonne of feedstock depending on the upgrading method used. This value was used as an input in GaBi. For this study, it is assumed that, in terms of the energy content, one unit of natural gas is equivalent to one unit of biomethane.

2.5 Net present value analysis

The economic feasibility of a project is analyzed by evaluating its net present value (NPV). The associated cash flows are taken into account over a certain period and cast in an equivalent comparable form of present date cash flows. A positive value of NPV dictates that a project is economically profitable, and therefore suitable for implementation. The economic components considered herein for the NPV analysis are (1) capital expenditure (CAPEX) associated with constructing the anaerobic digestion plant and the upgrading plant, (2) operations and maintenance (O&M) cost, (3) collection and transportation (C&T) cost, (4) income from renewable heat incentive (RHI), (5) the income generated by selling the digestate (DS), (6) revenues generated by receiving compensation for intaking waste i.e. gate fees (GF), (7) income from grid injection (GI) of biomethane, and (8) revenues from selling biomethane certificates (BMC) to energy producers. The NPV depending on the present values of all the financial components is as follows,

$$NPV=[PV_{CAPEX}+PV_{O\&M}+PV_{C\&T}]-[PV_{RHI}+PV_{DS}+PV_{GF}+PV_{GI}+PV_{BMC}] \quad (2)$$

where PV denotes the present cash flow value of each economic element, given by,

$$PV=\frac{CF_t}{(1+i)^t} \quad (3)$$

Here CF is the annual cash flow, i stands for the discount rate, and t denotes the period of the case study. For this case values for i=5% and t=20 years were assumed, consistent with the literature (Ascher et al., 2019; Ascher et al., 2020; Chang & Pires, 2015).

2.5.1 Capital (CAPEX), operational, and maintenance (O&M) costs of the plant

The initial investment for the AD plant was calculated based on the amount of feedstock the plant could handle. The work by (Angelonidi & Smith, 2015) reported CAPEX data of several AD plants installed in Europe, out of which two were located in the UK. This data assisted by

an additional database from Anaerobic Digestion and Bioresources Association (ADBA) (Robert, 2020) provides an educated estimate of £7,718,974 for each AD plant.

In addition to the AD plant, the procurement costs for upgrading systems need to be accounted for. Table 3 lists capital costs for the plants of the four upgrading technologies as a function of the plant capacity. The resulting CAPEX for various upgrading technologies falls within the range of £950,000 to just over £1,300,000. It is worthwhile mentioning that the CAPEX associated with the biogas upgrading infrastructure is within the range of 12-17% of the CAPEX of the AD plant.

Following (Vo et al., 2018) the maintenance costs for the AD and upgrading plants are assumed to be 3% and 4% of the CAPEX, respectively. Each plant is assumed to have 5 employees: 2 operators for the AD plant, 2 operators for the upgrading plant, and 1 manager per site. In total, there are 10 employees with the managers earning £70,000/year and the operators earning £30,000/year (Vo et al., 2018).

2.5.2 Expenditure for collection and transportation

The main costs for collection and transportation (C&T) are the capital cost of the trucks, fuel, and wages. In 2018, the Swansea Council ordered 38 new refuse vehicles at a total price of £6 million (Dalton, 2018). This implies a cost of just under £160,000 which serves as the estimate used in this study. Another assumption made is the lifespan of the vehicles. The lifespan of the vehicles is conservatively estimated at 10 years so in the 11th year the vehicles are replaced, meaning there is a large cash outflow at that point (Ascher et al., 2020). The price of vehicles is adjusted for 2% inflation. The maintenance cost of trucks was estimated to be £2,725 per year (conversion from Euros to Pounds Sterling) based on (Groot et al., 2014). Each truck requires one driver and two operators to load the waste. Each staff is paid a wage of £8/hr so the wages for one truck are costed at £24/hr. The wage bill is £1.68 million per annum. The cost of diesel in Scotland is 116.8p/L, and the annual fuel cost is estimated to be £675,000.

2.5.3 Income from digestate

The income from digestate is calculated based on a study by (Renda et al., 2016). The value of €15/tonne is used and adjusted for inflation. In Pounds Sterling, the price is £15.12 which contributes £1,072,692 of annual income.

2.5.4 Income from gate fees

It is common that a waste management plant will require some form of compensation for taking the discarded waste. Usually, this comes in the form of gate fees which can contribute a significant proportion of the overall profitability of the project. The WRAP publishes an annual comparison of waste treatment options in the UK, including the gate fees that are used. The average gate fee for AD in the UK in 2019 was £27 per tonne of waste (Wrap.org.uk, 2020). However, the gate fees for FW are usually higher than the average gate fee. Gate fees are volatile year-on-year, but a price of £45/tonne of FW, which is lower than some estimates found in the literature (Ascher et al., 2020), is used in this project which assumes that increased supply will be mandated putting upward pressure on the price. For the CS section of the feedstock, there are currently no accompanying gate fees. The total income from gate fees is £1,352,133 per annum.

2.5.5 Income from RHI and selling biomethane

The wholesale price of gas, the price paid to producers of gas by buyers of gas, is traded in the market and varies markedly over time (12p - 86p/therm) (1 therm = 1.055×10^8 joules). An average price of 51p/therm (1.75p/kWh) is used to estimate the income for the project, which is consistent with the data obtained from ADBA (Robert, 2020).

The UK Government also offers a subsidy in the form of a renewable heat incentive (RHI) for producers injecting biomethane into the grid. The 3-tier biomethane scheme is as follows: the first 40,000MWh yields 4.92p/kWh; the second 40,000MWh receives 2.9p/kWh; and any remaining production receives 2.24p/kWh (Turley et al., 2016). The biomethane produced by the four upgrading systems ranges from 24,200MWh to 26,800MWh and would receive the highest tariff. To estimate the cash flows, it is assumed that the current RHI subsidy scheme is extended for the lifespan of the project.

2.5.6 Income from selling biomethane certificates

The final component of cash flow is the income from biomethane certificates. These are market-driven entities affected by the demand for renewable gas. Biomethane producers can sell these certificates to energy providers to demonstrate their renewable credentials or fulfill sustainability quotas. According to ADBA (Robert, 2020), the price of the certificates has risen from £2 to £9. The estimated income is dependent on system biomethane output. Given expectations of increased demand for renewable energy, the higher value of £9 per certificate is used in this study.

3 Results and Discussion

3.1 Energy efficiency and total energy production

The goal of upgrading biogas produced by co-digestion of FW-CS mixture is to separate CH_4 from CO_2 and other chemical compounds. This enhances the calorific value of the gas, ultimately reducing the NO_x emissions and making it suitable for injection into the gas grid. The selection of upgrading facility for the city of Glasgow is dictated by government-regulated injection quality to the gas grid (Bozorg et al., 2020). For all the four classes of upgrading plants the CH_4 purity, i.e. the content of CH_4 in biomethane is high enough to be injected into the gas grid (Barbera et al., 2019). This is due to low losses (or CH_4 slip) during the biogas upgradation.

Table 4 shows the energy generation, energy efficiency, and the number of equivalent households that can be powered by the AD plant with specific upgrading technologies. The efficiency is calculated as (Sun et al., 2015),

$$\eta = \frac{E_{\text{upgraded\_gas}}}{E_{\text{raw\_gas}} + E_{\text{upgrading}}} \tag{4}$$

where $E_{\text{upgraded\_gas}}$ is the energy content in the upgraded biomethane and $(E_{\text{raw\_gas}} + E_{\text{upgrading}})$ is the total input energy required by the entire plant. Based on the total number of households in the city of Glasgow 285,693 (GCC, 2017) and average annual UK household gas consumption of 12,000 kWh (OFGEM, 2020), the number of households that can be supplied with biomethane was calculated and shown in Table 4 for each case.

Although the chemical scrubbing technique generates the highest amount of energy, thus powering the largest number of households (2,260) in the city of Glasgow, the membrane separation technique has the highest efficiency (83%). Hence, the AD plant with the membrane separation upgrading technology has been considered for further implementation, which would generate sufficient energy to power 2252 households (or 0.8% of total) in Glasgow. In this context, it is important to note that the efficiency of combined heat and power systems is in the range of 50-80% (Ascher et al., 2020). This makes biomethane production an efficient alternative to the combined heat and power (CHP) systems. Even though the PSA offers the poorest efficiency amongst all (76%), it falls within the upper band of efficiency for CHP systems. In contrast, the existing natural gas systems have efficiencies up to 60%.

3.2 Environmental impact

Figure 2 shows contributions of eight GWP components for the four different upgrading technologies: (1) emissions resulting from CS, (2) emissions from FW, (3) emissions avoided by using digestate as biofertilizer, (4) natural gas emissions avoided by utilizing biomethane as a substitute, (5) emissions resulted from collection and transportation of FW and CS, (6) emissions due to grid electricity consumption, (7) emissions from biogas leakage in the AD plant, and (8) off-gas emissions resulted from the biogas upgrading process.

The GHG emissions released during C&T totaled 7 kg $CO_{2\text{-eq}}$ per tonne of feedstock, while that from powering the plants varied with the different technologies in the range of 6 to 15 kg $CO_{2\text{-eq}}$ per tonne of feedstock. These emissions could be avoided because (1) FW is not disposed of in a landfill or incinerated, (2) CS was not left untreated, and (3) the biogas generated from the plant partly reduces the demand for natural gas production. Avoided gases include CH_4, N_2O, NH_3, and CO_2. Avoided emissions overwhelm the emissions from the systems for all four biogas upgrading technologies, and net negative carbon emissions are achieved.

In Figure 2, the emissions are shown above the baseline (zero-level), while the avoided emissions are below the baseline. To conduct a like-for-like comparison across different upgrading technologies, the anaerobic co-digestion of the feedstock process preceding the biogas upgrading step is kept constant. Hence, the emission components from CS, FW, digestate, C&T, and biogas leakage are the same for all the upgrading technologies. The largest contributor to the carbon savings was the avoided emission from the displacement of natural gas (4.89 kg $CO_{2\text{-eq}}/m^3$ natural gas). Although chemical scrubbing significantly replaces natural gas and results in the lowest emissions from off-gas, it has the highest power requirement amongst all (0.87 kWh/m^3), which increases the overall emission. This scheme has a total carbon saving of 264 kg $CO_{2\text{-eq}}$ per tonne of feedstock. The membrane separation plant yields a similar carbon saving compared to the chemical scrubbing, but offers a higher efficiency (4% greater) of biomethane production (see Table 4). In addition, the digestate production contributes to carbon mitigation (saving 27 kg $CO_{2\text{-eq}}$ per tonne of feedstock), while the biogas leakage results in positive emission (22 kg $CO_{2\text{-eq}}$ per tonne of feedstock). It is evident from the net carbon emission savings that the PSA upgrading technology is unfavourable due to its high off-gas emissions (which is mostly CO_2), which can be further reduced by off-gas treatment (Sun et al., 2015).

3.3 Economic assessment

3.3.1 Contribution of economic components

The NPVs of anaerobic co-digestion of FW and CS with the four different upgrading technologies are shown in Figure 3. Overall, the project's NPV irrespective of the upgrading technology falls within the range of around £ -9.0 to £ -12.0 million (see Figure 3). The negative value of NPV indicates that the developments are not economically viable under the current circumstance. Hence, a deeper investigation of the NPV components are essential.

Salient sources of expenses for the proposed developments are (1) C&T costs, (2) capital costs, and (3) O&M costs. It is evident from Figure 3 that C&T of waste has the highest negative NPV (£-38.0 million), of which £ -21.7 million is contributed by wages. The NPV of diesel costs for operating the truck is the second most significant component (£-10.8 million), while the CAPEX of trucks contributes to £ -5.5 million. The CAPEX of the AD plant with biogas upgrading is another significant component towards the negative NPV of the project. The NPV of the AD plant is £-15.44 million, while the NPV of biogas upgrading plant varies since the CAPEX associated with the various upgrading technologies are different (see Table 3). The total NPVs for the four different upgrading techniques are as follows: (a) £-1.21 million for PWS, (b) £-1.32 million for chemical scrubbing, (c) £-1.2 million for PSA, and (d) £-0.95 million for membrane separation. O&M costs are another significant contributor which varies between a small range of £-12.0 million to £-12.4 million depending on the upgrading technology.

Shown as the positive components of NPV in Figure 3, the proposed developments have five income sources: (1) gaining RHI selling digestate, (2) injecting biomethane into the grid at a wholesale rate, (3) selling biomethane certificates, (4) gate fees, and (5) selling digestate. Although the RHI subsidy scheme provided by the government is likely to end in 2021, it is assumed that it will be replaced by an identical or similar scheme for the next 20 years. The Energy White Paper "Powering our Net Zero Future" published by the Secretary of State for Business, Energy and Industrial Strategy (BEIS) on 14th December 2020 is silent on the detail. However, they promise a Green Gas Support Scheme in the autumn of 2021 that will endorse the continued deployment of AD biomethane plants with the prospect of trebling the amount of green gas in the grid between 2018 and 2030 (Robert, 2020).

For the present sizes of plants investigated, the energy content of biomethane injected into the grid would not exceed 40,000 MWh, providing a tier-1 RHI subsidy of 4.92 p/kWh.

Chemical scrubbing achieves the highest biomethane generation, and therefore, has the highest income for £15.99 per tonne of feedstock. In contrast, PSA only achieves £14.40 due to the lower CH_4 purity achieved (Appendix A) and a relatively low CH_4 recovery rate (90.29%). The importance of the RHI scheme is clear for all the technologies considering its 28-30% contribution towards the total annual cash inflow.

The income from biomethane injection to the grid is dictated by the wholesale price of gas and the amount of biomethane injected. The total annual income from this ranges between £0.43 to £0.47 millions. Therefore, the revenues generated from RHI, grid injection, and biomethane production totals to a range of £21.5 to £23.7 millions. The third source of the income i.e., the biomethane certificate at £9 per certificate will contribute a total NPV of £2.9.to £3.2 million, making it 5% of the total income of the system.

The cash inflows of selling the digestate and gate fees are not directly dependent on the upgrading technologies. It is shown that selling the digestate contributes £15.12 per tonne of feedstock and generates an income of £1.1 million annually at the full capacity of the plant. Gate fees contribute a positive NPV of £17.7 million. However, it is worthwhile noting that there is potential uncertainty in its magnitude. Currently, there are gate fees for FW, but no gate fees for CS, which can be mitigated through strict regulations on the storage of CS. Moreover, the FW gate fee widely differs across the UK and lacks a proper database. Further research is required to address this discrepancy, which is beyond the scope of the present work. With no gate fees from CS, the total gate fee per tonne of feedstock is £16.20.

Since the NPV of the project, regardless of the biogas upgrading technology, is negative, the project is not economically profitable (see Figure 3). However, it is possible to make policies that will penalize fossil fuel-based technologies and will ascribe sufficiently a high carbon tax to break even the NPV. The influence of carbon tax is investigated in later sections. It is also noted that future changes in gate fees, population size, recycling behavior, and waste production per capita may significantly affect the available feedstock and the required size and related costs of the AD plants, but are not considered in this work. Hence, future works are recommended to focus on the analysis and quantification of the potential uncertainties.

3.3.2 Influence of input parameters

The results of this study (negative NPV regardless of upgrading technology) point to the need for financial support for the proposed systems producing biomethane from the co-digestion of

FW and CS. Subsidies accounted for £18.8 million to £20.7 million of the NPV, which is consistent with prior works (Cucchiella et al., 2019a; Cucchiella et al., 2019b). Another work (Benato & Macor, 2019) investigated the history of the biogas industry, where they noted the impact of generosity of subsidies in Germany and Italy. Given the catastrophic consequence of climate change, biomethane production can play a vital role in reducing GHG emissions. In this context, support in the UK should be forthcoming as indicated in the recently published BEIS white paper (BEIS, 2020).

Although it is beyond the scope of this work to propose an appropriate policy mix, the sensitivity analysis of the NPV based on the variations of techno-economic parameters is essential. This will provide the policymakers with some clarity on the conditions required to make an AD investment with the membrane separation technology (i.e. the most economically viable option according to this study) financially attractive. Figure 4 and Table 5 show NPVs for a wide range of parametric sensitivity scenarios. Essential parameters investigated are (1) biogas yield, (2) RHI subsidy, (3) digestate selling price, (4) gate fees, (5) number of waste collection vehicles, (6) wages of waste collection and transportation personnel, (7) biomethane certificate price, and (8) wholesale price of biomethane. To investigate the relative importance of different parameters, a sensitivity ratio SR is defined as,

$$\mathrm{SR}=\frac{\Delta_{\mathrm{output}}}{\Delta_{\mathrm{input}}} \tag{5}$$

where Δ_{output} is the percentage of change in output quantity i.e., the NPV and Δ_{input} is the percentage of changes in input parameters. A higher value of SR corresponds to a more significant change in the NPV due to input parameter alteration.

As shown in Figure 4 and Table 5, the NPV is most sensitive to biogas yield ($\mathrm{SR} = 2.63$), RHI subsidies ($\mathrm{SR} = 1.94$), gate fees ($\mathrm{SR} = 1.98$), and selling price of digestate ($\mathrm{SR} = 1.57$). The price of biomethane certificates and wholesale gas prices have a less severe impact on the NPV ($\mathrm{SR} < 0.7$). The sensitivity analysis also indicates the financial vulnerability of the project subject to market changes. Gate fees, digestate prices, and the wholesale price of gas for grid injection are relatively uncertain and time-varying as compared to the RHI and biomethane certificate schemes.

3.3.3 Influence of carbon tax alteration

An essential factor to generate economic benefit while adhering to the climate change guidelines is through assigning carbon taxes. Carbon pricing creates the signal for economic actors to redeploy capital away from fossil fuels to low carbon alternatives. Evidence from Sweden, which has the highest carbon tax in the world at US$ per tonne of CO_2 (WB, 2020), indicates that after implementation of the tax, initially at US$ 30 per tonne, CO_2 emissions from transport declined almost 11% relative to a synthetic control group of OECD countries with lower carbon taxes. It was estimated that carbon prices of at least US$ 50–100 per tonne of CO_2 by 2030 are required to cost-effectively reduce GHG emissions to fulfill the temperature goals of the Paris Agreement (WB, 2020).

By keeping all other assumptions constant, the financial model was used to calculate the impact of a carbon tax on NPV for the systems (see Table 6). A biomethane producer would receive revenue reflecting their contribution to society by avoiding emissions. The analysis shows that a carbon tax between £31.30 and £58.02 per tonne of CO_2 would be required to make the four schemes return positive NPVs.

Although a truly efficient carbon tax would require all market participants to apply the same tax, it is possible to apply it unilaterally to help achieve the desired outcome at least locally. In the absence of global cooperation, other policies can spur faster development of biomethane production and other low carbon technologies, through an increase in tariffs or minimization of expenditure, such as grants for research and development and reduced planning and regulatory burden.

4 Conclusions

The proposed biomethane-biofertilizer production scheme contributes to the concept of circular economy and organic waste recycling to tackle global warming. Membrane separation-assisted AD meets 0.8% of the natural gas demand of Glasgow households with a conversion efficiency of 83%. This mitigates 264 kg $CO_{2\text{-eq}}$ per tonne of organic waste. However, high CAPEX for the AD plant with the membrane separation-based technology (NPV of £-9.0 million) and high costs of waste C&T (NPV of £-38.1 million) renders the scheme economically unfavourable under current circumstances, unless a carbon tax £31.30 per tonne of CO_2 saved is provided to support the scheme.

Acknowledgments

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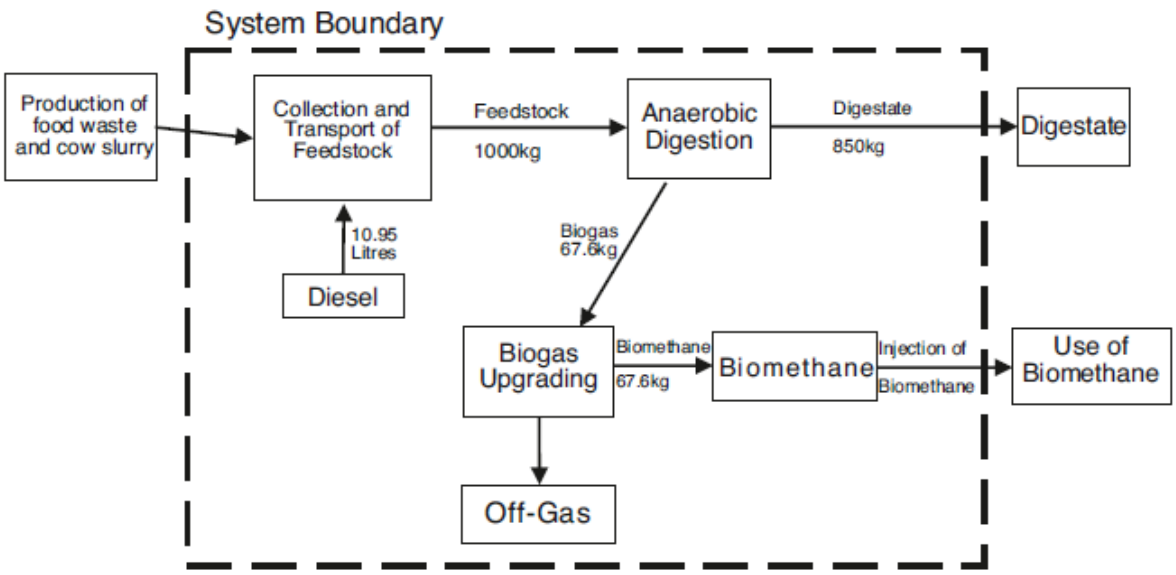

Figure 1: Flowchart of the waste to biomethane conversion system with LCA system boundary and salient processes.

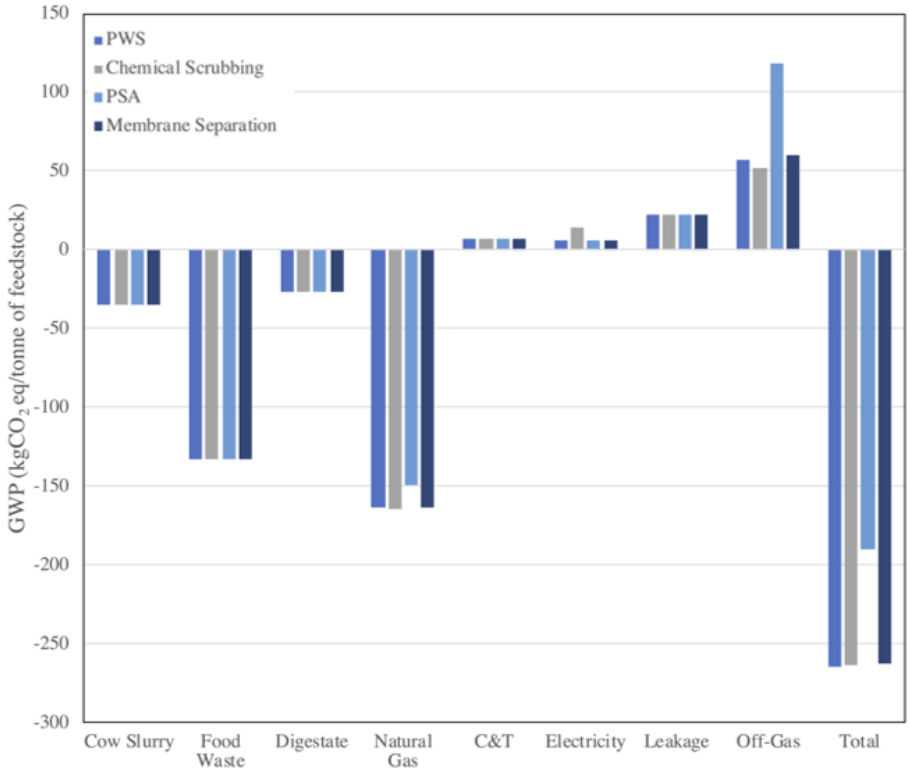

Figure 2: GWP breakdown for the four different biogas upgrading techniques, i.e. PWS, chemical scrubbing, PSA, and membrane separation.

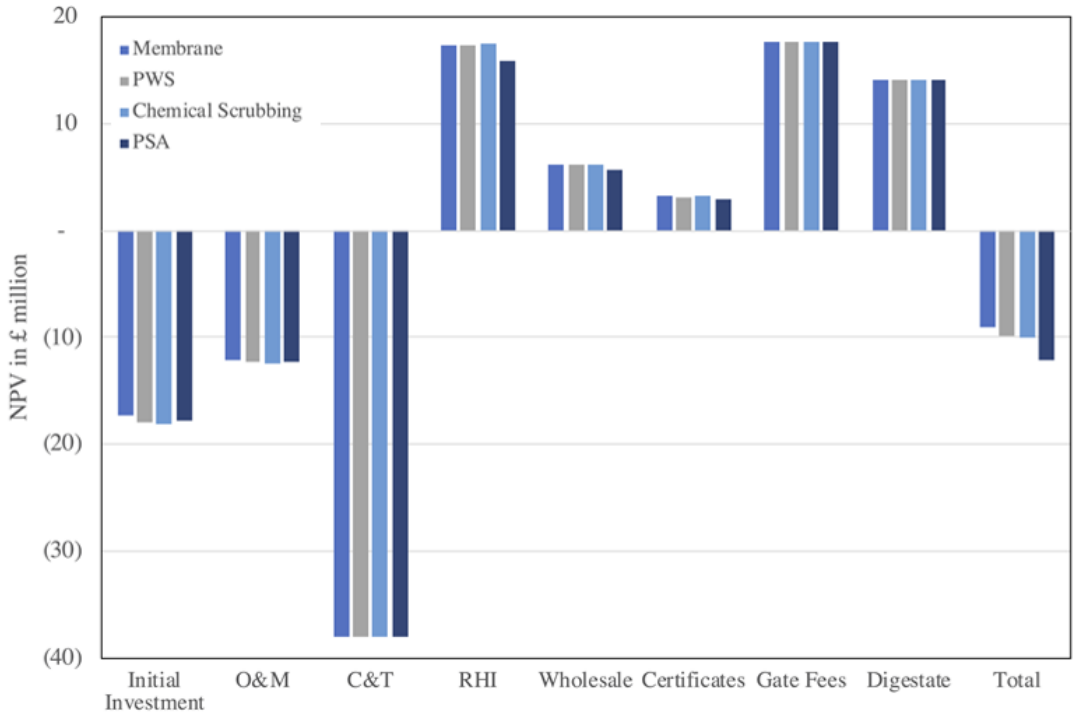

Figure 3: Components of net present value (NPV) of cash flows in £ million for the four different biogas upgrading technologies, i.e. PWS, chemical scrubbing, PSA, and membrane separation. Negative values on the y-axis are shown in brackets. The total NPVs for the four technlogies in £ are -9,886,001 (PWS), -9,9969,129 (chemical scrubbing), -12,034,405 (PSA), and -8,955,465 (membrane seperation).

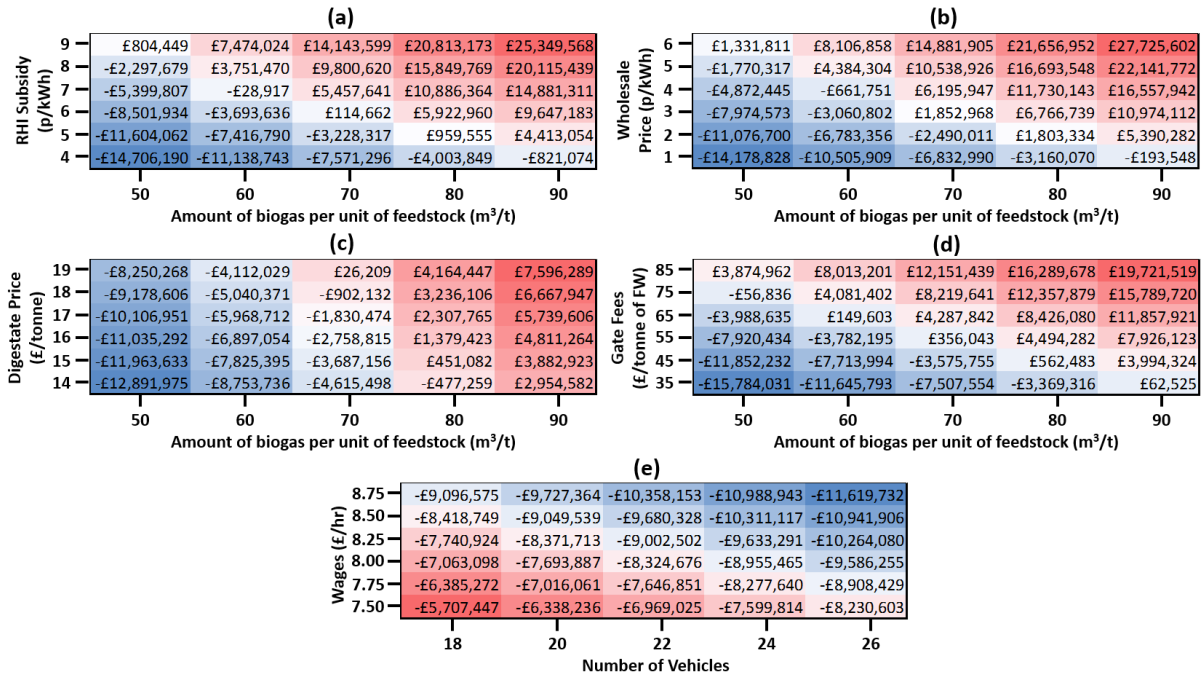

Figure 4: Heatmaps showing the sensitivity of NPVs in case of an AD plant with membrane separation for various dual-parametric alteration scenarios, i.e. (a) RHI subsidy *vs.* biogas yield, (b) wholesale price *vs.* biogas yield, (c) digestate price *vs.* biogas yield, (d) gate fees for FW collection *vs.* biogas yield, and (e) hourly wages *vs.* the number of vehicles. Blue and red colors are mapped for unfavourable (higher negative values) and favourable (values tending towards zeros or positive) NPVs, respectively.

Table 1: Avoided emissions from digestate production (Banks et al., 2011; DEFRA, 2010; Evangelisti et al., 2014; Tampio et al., 2015).

Nutrients	Food waste (g/kgTS)[a]	Cow slurry (g/kgTS)[b]	Content per tonne of feedstock (kg)	Emissions per kg (kg $CO_{2\text{-eq}}$)	Emissions per tonne of feedstock (kg $CO_{2\text{-eq}}$)[c]
Nitrogen	30.7	3.3	2.82	8.85	24.96
Phosphorus	3.8	0.8	0.37	1.80	0.67
Potassium	11.4	1.6	1.07	0.96	1.03
Total				11.61	26.66

a: TS in FW: 23.74% (Evangelisti et al., 2014)
b: TS in CS: 9.31% (Evangelisti et al., 2014)
c: Feedstock = FW:CS 1:1.8 (Banks et al., 2011)

Table 2: Input and output data for modeling the biogas upgrading processes in GaBi software.

Technology	Input		Output	
	Quantity	Value	Quantity	Value
PWS	Biogas[a]	55.29 m^3	Biomethane[b]	33.3 m^3
	Electricity[b]	0.27 kWh/m^3	Off-gas[c]	57.3 kg $CO_{2\text{-eq}}$
Chemical scrubbing	Biogas[a]	55.29 m^3	Biomethane[b]	33.6 m^3
	Electricity[b]	0.87 kWh/m^3	Off-gas[c]	51.5 kg $CO_{2\text{-eq}}$
Membrane separation	Biogas[a]	55.29 m^3	Biomethane[d]	33.5 m^3
	Electricity[d]	0.26 kWh/m^3	Off-gas[c]	60.5 kg $CO_{2\text{-eq}}$
PSA	Biogas[a]	55.29 m^3	Biomethane[b]	30.6 m^3
	Electricity[b]	0.26 kWh/m^3	Off-gas[c]	118.7 kg $CO_{2\text{-eq}}$

a: (Banks et al., 2011)
b: (Barbera et al., 2019)
c: calculated
d: (Bozorg et al., 2020)

Table 3: Captial expenditure as a function of plant capacity in m^3/h for four different biogas upgrading technologies.

Upgrading Technique	Capacity (m^3/h)	Total cost (£)	Reference
PWS	100	1,099,933	(Chen et al., 2015)
	250	1,497,432	(Chen et al., 2015)
	500	1,905,823	(Chen et al., 2015)
	2000	3,470,628	(Sun et al., 2015)
	200	1,397,545	(Sun et al., 2015)
	600	1,699,699	(Sun et al., 2015)
Chemical scrubbing	100	1,034,912	(Chen et al., 2015)
	250	1,361,725	(Chen et al., 2015)
	500	1,906,416	(Chen et al., 2015)
	100	1,004,390	(Sun et al., 2015)
	500	2,517,573	(Sun et al., 2015)
	1800	4,394,310	(Sun et al., 2015)
PSA	100	1,132,956	(Chen et al., 2015)
	250	1,470,664	(Chen et al., 2015)
	500	2,015,355	(Chen et al., 2015)
	130	740,779	(Sun et al., 2015)
	1000	3,051,604	(Sun et al., 2015)
	500	2,212,413	(Angelonidi & Smith, 2015)
Membrane separation	100	811,587	(Chen et al., 2015)
	250	1,307,256	(Chen et al., 2015)
	500	1,960,884	(Chen et al., 2015)
	1000	2,015,355	(Sun et al., 2015)
	130	253,826	(Sun et al., 2015)
	300	1,373,222	(Sun et al., 2015)

Table 4: Biomethane generation for domestic use with the four different upgrading technologies.

Upgrading Technique	Input Energy (kWh)	Energy Produced (kWh)	Efficiency (%)	Households Supplied
Membrane Seperation	32,601,935	27,025,900	83	2,252
PWS	32,625,755	26,878,038	82	2,240
Chemical Scrubbing	34,315,195	27,119,473	79	2,260
PSA	32,487,344	24,660,925	76	2,055

Table 5: Sensitivity of NPV to input parameter changes for waste to biomethane conversion system with membrane separation upgrading technique.

Parameter	Change of Input (%)	NPV (£)	Change of Output (%)	SR
Base Case	-	-8,955,465	-	-
Biogas Yield	-40	-18,390,649	-105	2.63
	-20	-13,673,057	-53	2.63
	20	-4,237,874	53	2.63
	40	-479,718	105	2.63
RHI Subsidies	-40	-15,915,151	-78	1.94
	-20	-12,435,308	-39	1.94
	20	-5,475,623	39	1.94
	40	-1,995,780	78	1.94
Gate Fees	-40	-16,032,703	-79	1.98
	-20	-12,494,084	-40	1.98
	20	-5,416,847	40	1.98
	40	-1,878,228	79	1.98
Digestate	-40	-14,570,074	-63	1.57
	-20	-11,762,770	-31	1.57
	20	-6,148,161	31	1.57
	40	-3,340,857	63	1.57
Certificates	-40	-10,233,054	-14	0.36
	-20	-9,594,260	-7	0.36
	20	-8,316,671	7	0.36
	40	-7,677,877	14	0.36
Wholesale Price	-40	-11,430,963	-28	0.69
	-20	-10,193,214	-14	0.69
	20	-7,717,716	14	0.69
	40	-6,479,967	28	0.69

Table 6: Impact of the carbon tax on NPV for different upgrading techniques.

Carbon tax (£ per tonne)	Upgrading Technique			
	Membrane Separation	PWS	Chemical Scrubbing	PSA
10	-6,094,405	-7,001,311	-7,094,384	-9,960,360
20	-3,233,344	-4,116,622	-4,219,640	-7,886,316
30	-372,283	-1,231,932	-1,344,895	-5,812,271
40	2,488,777	1,652,757	1,529,849	-3,738,227
50	5,349,838	4,537,447	4,404,594	-1,664,183
60	8,210,899	7,422,136	7,279,338	409,862

References

Angelonidi, E., Smith, S.R. 2015. A comparison of wet and dry anaerobic digestion processes for the treatment of municipal solid waste and food waste. *Water and environment journal*, **29**(4), 549-557. https://doi.org/10.1111/wej.12130.

Ascher, S., Watson, I., Wang, X., You, S. 2019. Township-based bioenergy systems for distributed energy supply and efficient household waste re-utilisation: Techno-economic and environmental feasibility. *Energy*, **181**, 455-467. https://doi.org/10.1016/j.energy.2019.05.191.

Ascher, S., Li, W., You, S. 2020. Life cycle assessment and net present worth analysis of a community-based food waste treatment system. *Bioresource technology*, **305**, 123076. https://doi.org/10.1016/j.biortech.2020.123076.

Banks, C.J., Salter, A.M., Heaven, S., Riley, K. 2011. Energetic and environmental benefits of co-digestion of food waste and cattle slurry: A preliminary assessment. *Resources, Conservation and Recycling*, **56**(1), 71-79. https://doi.org/10.1016/j.resconrec.2011.09.006.

Baral, K.R., Jégo, G., Amon, B., Bol, R., Chantigny, M.H., Olesen, J.E., Petersen, S.O. 2018. Greenhouse gas emissions during storage of manure and digestates: Key role of methane for prediction and mitigation. *Agricultural systems*, **166**, 26-35. https://doi.org/10.1016/j.agsy.2018.07.009.

Barbera, E., Menegon, S., Banzato, D., D'Alpaos, C., Bertucco, A. 2019. From biogas to biomethane: A process simulation-based techno-economic comparison of different upgrading technologies in the Italian context. *Renewable Energy*, **135**, 663-673. https://doi.org/10.1016/j.renene.2018.12.052.

BEIS. 2020. Energy white paper: Powering our net zero future. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/945899/201216_BEIS_EWP_Command_Paper_Accessible.pdf.

Benato, A., Macor, A. 2019. Italian biogas plants: Trend, subsidies, cost, biogas composition and engine emissions. *Energies*, **12**(6), 979. https://doi.org/10.3390/en12060979.

Bozorg, M., Ramírez-Santos, Á.A., Addis, B., Piccialli, V., Castel, C., Favre, E. 2020. Optimal process design of biogas upgrading membrane systems: Polymeric vs high performance inorganic membrane materials. *Chemical Engineering Science*, **225**, 115769. https://doi.org/10.1016/j.ces.2020.115769.

Chang, N.-B., Pires, A. 2015. *Sustainable solid waste management: A systems engineering approach*. John Wiley & Sons.

Chen, X.Y., Vinh-Thang, H., Ramirez, A.A., Rodrigue, D., Kaliaguine, S. 2015. Membrane gas separation technologies for biogas upgrading. *Rsc Advances*, **5**(31), 24399-24448. https://doi.org/10.1039/C5RA00666J.

Cucchiella, F., D'Adamo, I., Gastaldi, M. 2019a. An economic analysis of biogas-biomethane chain from animal residues in Italy. *Journal of Cleaner Production*, **230**, 888-897. https://doi.org/10.1016/j.jclepro.2019.05.116.

Cucchiella, F., D'Adamo, I., Gastaldi, M. 2019b. Sustainable Italian cities: The added value of biomethane from organic waste. *Applied Sciences*, **9**(11), 2221. https://doi.org/10.3390/app9112221.

Dalton, M. 2018. £6 million refuse vehicle order for Dennis Eagle. https://www.thetruckexpert.co.uk/six-million-refuse-bin-investment/.

DEFRA. 2010. Fertiliser Manual (RB209). Department for Environment Food and Rural Affairs. https://www.rothamsted.ac.uk/sites/default/files/rb209-fertiliser-manual-110412.pdf.

Dick, H. 2020. The composition of municipal solid waste in Scotland. https://www.wrap.org.uk/sites/files/wrap/Scotland_MSW_report_final.pdf.

Dray, S. 2021. Food waste in the UK. UK parliament, House of Lords, Library. https://lordslibrary.parliament.uk/food-waste-in-the-uk/.

Ec.europa.eu. 2020. Municipal Waste Statistics - Statistics Explained. https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Municipal_waste_statistics#Municipal_waste_generated_by_country.

Edwards, J., Othman, M., Burn, S., Crossin, E. 2016. Energy and time modelling of kerbside waste collection: Changes incurred when adding source separated food waste. *Waste Management*, **56**, 454-465. https://doi.org/10.1016/j.wasman.2016.06.033.

EEA. 2020. Waste Recycling. https://www.eea.europa.eu/data-and-maps/indicators/waste-recycling-1/assessment-1.

Evangelisti, S., Lettieri, P., Borello, D., Clift, R. 2014. Life cycle assessment of energy from waste via anaerobic digestion: a UK case study. *Waste management*, **34**(1), 226-237. https://doi.org/10.1016/j.wasman.2013.09.013.

GCC. 2017. Glasgow City Ward Factsheets. https://www.glasgow.gov.uk/CHttpHandler.ashx?id=3543&p=0.

Groot, J., Bing, X., Bos-Brouwers, H., Bloemhof-Ruwaard, J. 2014. A comprehensive waste collection cost model applied to post-consumer plastic packaging waste. *Resources, Conservation and Recycling*, **85**, 79-87. https://doi.org/10.1016/j.resconrec.2013.10.019.

Kapoor, R., Ghosh, P., Kumar, M., Vijay, V.K. 2019. Evaluation of biogas upgrading technologies and future perspectives: a review. *Environmental Science and Pollution Research*, **26**(12), 11631-11661. https://doi.org/10.1007/s11356-019-04767-1.

Kristanto, G.A., Koven, W. 2019. Estimating greenhouse gas emissions from municipal solid waste management in Depok, Indonesia. *City and Environment Interactions*, **4**, 100027. https://doi.org/10.1016/j.cacint.2020.100027.

Kvist, T., Aryal, N. 2019. Methane loss from commercially operating biogas upgrading plants. *Waste Management*, **87**, 295-300. https://doi.org/10.1016/j.wasman.2019.02.023.

Møller, J., Boldrin, A., Christensen, T.H. 2009. Anaerobic digestion and digestate use: accounting of greenhouse gases and global warming contribution. *Waste management & research*, **27**(8), 813-824. https://doi.org/10.1177/0734242X09344876.

Moult, J., Allan, S., Hewitt, C., Berners-Lee, M. 2018. Greenhouse gas emissions of food waste disposal options for UK retailers. *Food Policy*, **77**, 50-58. https://doi.org/10.1016/j.foodpol.2018.04.003.

OFGEM. 2020. Typical Domestic Consumption Values. https://www.ofgem.gov.uk/gas/retail-market/monitoring-data-and-statistics/typical-domesticconsumption-values.

Renda, R., Gigli, E., Cappelli, A., Simoni, S., Guerriero, E., Romagnoli, F. 2016. 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*, **95**, 385-392. https://doi.org/10.1016/j.egypro.2016.09.042.

Robert, Z. 2020. Biomethane: the pathway to 2030. Anaerobic Digestion and Bioresources Association. https://adbioresources.org/report-biomethane-the-pathway-to-2030/.

Skone, T.J., Marriott, J., Littlefield, J. 2016. Life cycle greenhouse gas emissions: natural gas and power production. NETL. https://www.eia.gov/conference/2015/pdf/presentations/skone.pdf.

Sun, Q., Li, H., Yan, J., Liu, L., Yu, Z., Yu, X. 2015. Selection of appropriate biogas upgrading technology-a review of biogas cleaning, upgrading and utilisation. *Renewable and Sustainable Energy Reviews*, **51**, 521-532. https://doi.org/10.1016/j.rser.2015.06.029.

Tampio, E., Ervasti, S., Rintala, J. 2015. Characteristics and agronomic usability of digestates from laboratory digesters treating food waste and autoclaved food waste. *Journal of Cleaner Production*, **94**, 86-92. https://doi.org/10.1016/j.jclepro.2015.01.086.

Turley, D., Hopwood, L., Burns, C., Maio, D.D. 2016. Assessment of digestate drying as an eligible heat use in the Renewable Heat Incentive. NNFCC The Bioeconomy Consultants. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/577039/Annex_D_-_Report_on_digestate_drying.pdf.

Vo, T.T., Wall, D.M., Ring, D., Rajendran, K., Murphy, J.D. 2018. Techno-economic analysis of biogas upgrading via amine scrubber, carbon capture and ex-situ methanation. *Applied energy*, **212**, 1191-1202. https://doi.org/10.1016/j.apenergy.2017.12.099.

Wang, H., Aguirre-Villegas, H.A., Larson, R.A., Alkan-Ozkaynak, A. 2019. Physical properties of dairy manure pre-and post-anaerobic digestion. *Applied Sciences*, **9**(13), 2703. https://doi.org/10.3390/app9132703.

Waterton, C., Norton, L., Cardwell, E., Chadwick, D., Gibbons, J., Macintosh, K., Sakrabani, R., Shrestha, S. 2017. Holistic decision-support for cattle slurry storage and treatment - techniques for maximum nutrient use efficiencies (SLURRY-MAX). http://wp.lancs.ac.uk/slurry-max/files/2018/10/Slurry-Max-Report.pdf.

WB. 2020. World bank, State and trends of carbon pricing. https://openknowledge.worldbank.org/bitstream/handle/10986/33809/9781464815867.pdf?sequence=4&isAllowed=y.

Wrap.org.uk. 2020. Comparing the costs of alternative waste treatment options. https://www.wrap.org.uk/sites/files/wrap/WRAP%5C%20gate%5C%20fees%5C%20report%5C%202019.pdf.

Yousef, A.M., El-Maghlany, W.M., Eldrainy, Y.A., Attia, A. 2019. Upgrading biogas to biomethane and liquid CO2: A novel cryogenic process. *Fuel*, **251**, 611-628. https://doi.org/10.1016/j.fuel.2019.03.127.

Zglobisz, N., Castillo-Castillo, A., Grimes, S., Jones, P. 2010. Influence of UK energy policy on the deployment of anaerobic digestion. *Energy Policy*, **38**(10), 5988-5999. https://doi.org/10.1016/j.enpol.2010.05.054.