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Comparison of the co-gasification of sewage sludge and food wastes and cost-benefit analysis of gasification- and incineration-based waste treatment schemes

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ABSTRACT

The compositions of food wastes and their co-gasification producer gas were compared with the existing data of sewage sludge. Results showed that food wastes are more favorable than sewage sludge for co-gasification based on residue generation and energy output. Two decentralized gasification-based schemes were proposed to dispose of the sewage sludge and food wastes in Singapore. Monte Carlo simulation-based cost-benefit analysis was conducted to compare the proposed schemes with the existing incineration-based scheme. It was found that the gasification-based schemes are financially superior to the incineration-based scheme based on the data of net present value (NPV), benefit-cost ratio (BCR), and internal rate of return (IRR). Sensitivity analysis was conducted to suggest effective measures to improve the economics of the schemes.

KEYWORDS

Cost-benefit analysis; Gasification; Food waste; Sewage sludge; Incineration; Monte Carlo simulation.

1. INTRODUCTION

Management of solid wastes has been one of the greatest challenges for megacities. Landfill remains as one of the predominant methods of waste disposal worldwide. However, even modern engineered landfill suffers from a variety of problems such as noxious gas emission, dust, and leachate production, rodent infestation, etc. (Hamer, 2003). Furthermore, the land space requirement of landfill makes it an unfavorable choice for countries that have limited land space, such as Singapore. Alternative waste treatment technologies such as incineration and gasification have gained increasing attention due to their great potential for energy- and resource-harvesting. Incineration could effectively reduce the volume of solid waste by 90 – 95%, but the corresponding waste burning process produces a cocktail of toxic by-products that are harmful to the environment and general public health (Tian et al., 2012). Compared to incineration, gasification is generally not only more efficient but also bears much less environmental concerns because the oxygen-deficient environment in a gasifier does not favor the formation of those environmental pollutants produced in an incinerator. Moreover, the gasification technology is well suitable for the decentralized application (Buragohain et al., 2010), which offers significant flexibility to waste treatment and could potentially reduce the contamination incurred during waste transportation.

In Singapore, two solid wastes, i.e. food waste and sewage sludge, among various types of wastes, are being paid special attention. Food waste (788,600 tons in 2014) is one of the major solid wastes generated in Singapore, but its recycling rate is only 13% and is among one of the lowest (NEA, 2016b). Sewage sludge is an unavoidable product from water reclamation plants (WRP) during the treatment of municipal and industrial wastewaters. There are four WRPs in Singapore and their capacity information is listed in Table 1. 148,500 tons of ash and sewage sludge were produced in 2014, with 21,700 recycled for a recycling rate of 15% (NEA, 2016b). In view of the annual ash production of around 50,000 tons (MEWR, 2016), the amount of sewage sludge produced per year is estimated to around 98,500 tons. The disposal of sewage sludge is challenging due to the fact that it comprises of a variety of harmful substances such as heavy metals, bacteria, viruses, poorly biodegradable organic compounds, dioxins etc. Currently in Singapore, the disposal of sewage sludge mainly relies on incineration. However, the high moisture content in sewage sludge makes it not an ideal fuel for incineration. Alternatively, the gasification technology serves as a potential candidate for tackling the disposal dilemma.

A great number of studies have been conducted to explore the gasification of food waste and sewage sludge. Sewage sludge has been co-gasified with biomass to effectively mitigate the adverse effects of various characteristics (e.g., high moisture content and toxic compounds) of sewage sludge on the process and enhance the gasification efficiency (Manara and Zabaniotou, 2012). However, the existing gasification experiments of food wastes and sewage sludge were conducted by different studies which generally have different operating conditions, equipment design, and experimental procedures. As a result, the comparison between the existing co-gasification experimental data of food wastes and sewage sludge is difficult, while such comparison provides information about the relative pros and cons of food wastes and sewage sludge for co-gasification which is important for the practical designing (e.g., electricity generator capacity planning based on the amount of food wastes and sewage sludge handled) and management (e.g., selection of food wastes or sewage sludge for gasification by decision makers) of gasification-based waste disposal. For example, the studies of Ong et al. (2015) and Yang et al. (2016) conducted the co-gasification experiments of sewage sludge and food waste, respectively, based on the same fixed-bed downdraft gasifier. However, the experimental conditions such as pretreatment of wastes in the two studies differed, making the experimental results less comparable.

Cost-benefit analysis (CBA) has been widely employed to evaluate the economic feasibility of various waste- and energy-related projects or programs (Koupaie et al., 2014; Ruffino et al., 2015). Through the systematic and analytical comparison of benefits and costs, CBA not only answers such questions as whether a proposed project or program is worthwhile, but also could serve as an effective tool for making reasonable decisions on the utilization and distribution of society's resources. However, there is still lacking relevant cost-benefit analysis about the deployment of decentralized gasification systems in megacities for waste disposal, especially for food wastes and sewage sludge, while such analysis would be critical to the decision-making process for policy-makers and investors.

In this work, we would conduct a series of co-gasification experiments of food wastes with woodchips. The experimental conditions are similar to the ones of sewage sludge reported by the study of Ong et al., (2015) to achieve better comparison. The respective pros and cons of food wastes and sewage sludge would be discussed in terms of the compositions of wastes and producer gas. Based on the experimental data, two decentralized gasification-based waste disposal schemes would be proposed to handle the sewage sludge and food wastes of Singapore. A CBA would be conducted to compare the proposed schemes with the existing incineration-based scheme considering both private and environmental costs, which sheds light on the practical application and arrangement of waste disposal systems.

2. MATERIALS AND METHODS

2.1 Feedstock Materials and Characterization

The co-gasification feedstock consists of a mixture of food wastes and woodchips. Food wastes were collected from the Techo Edge Canteen of the National University of Singapore (NUS). The collected food waste is divided into four categories mainly based on their nutrient composition, i.e. carbohydrate, protein, fats and bones, which account for 65 wt.%, 15 wt. %, 5 wt. %, and 15 wt.%, respectively. The carbohydrate category mainly contains rice, potato, noodle, pasta, vegetables, etc. The protein category mainly contains chicken, pork, fish, egg, etc. The fats category mainly contains pork fats and chicken skin.

The nutrient-based categorization improves the differentiation among different types of food wastes and enhances the reproducibility of experiments. In contrast, the study of Yang et al. (2016) does not clarify the criteria for categorizing food wastes, which may limit their experimental data to the types of food wastes considered in their study only. Since it is difficult to grind and make bones into small balls that fit into the gasifier, they are excluded in the subsequent experiments. The moisture content of food wastes was determined by the freeze-drying method (Baysal et al., 2015). Mesquite woodchips (Kingsford, The Clorox Company, USA) were used as the co-gasification agent, similar to the study of Ong et al., (2015). Proximate, ultimate, and inductively coupled plasma (ICP) analysis were conducted to characterize the compositions of the feedstocks. The details of the analysis could be found in the previous studies (Ong et al., 2015; Yang et al., 2016) and are not repeated in this work. The higher heating value (HHV) of feedstocks was calculated using the unified correlation for fuels developed by (Channiwala and Parikh, 2002)

$$\begin{aligned}\mathrm{HHV(MJ/kg)} = {} & 0.3491(\mathrm{C}) + 1.1783(\mathrm{H}) + 0.1005(\mathrm{S}) - 0.1034(\mathrm{O}) \\ & - 0.0151(\mathrm{N}) - 0.0211(\mathrm{ASH})\end{aligned} \tag{1}$$

where C, H, S, O, N and ASH are the mass percentage fractions of the respective components in the feedstocks as obtained from the proximate and ultimate analysis.

2.2 Co-gasification Experiments

2.2.1 Feedstock pretreatment

The pre-treatment of food wastes was conducted to control its size and moisture content for a smooth running of gasification experiments. For a fixed-bed downdraft gasifier, the moisture content of feedstock was suggested to be lower than 25 wt.% (Puig-Arnavat et al., 2010). Hence, similar to the sewage study of Ong et al., (2015), the wet food waste was rolled into balls and solar-dried to reduce the moisture content to below 25 wt.%, which ensures that the dried balls could mix well with the woodchips and subsequently be fed smoothly into the reactor. The original food waste balls were around 2.5 cm in diameter and shrunk to about 2 cm after drying. After the drying, the moisture content of food waste balls was reduced to around 10 wt.%. The woodchips have an initial moisture content of approximately 8 wt.% and no further pre-treatment was needed before gasification. The woodchips were sorted and handpicked to ensure their length and width between 1 to 4 cm, so that they could be fed smoothly into the reactor via the screw feeder.

2.2.2 Experimental design

The air flow rate of 7×10^{-3} m^3/s was used. The study of Ong et al., (2015) suggested an optimal sludge-woodchips ratio of 1:4 for the co-gasification experiments. For the food waste, the carbohydrate balls were firstly mixed with the protein balls in a ratio of 4:1. The content of protein balls would increase the HHV of feedstock and serve as a source of nitrogen. The nitrogen content in the protein food wastes suggests a potential environmental concern of the emission of nitrogen oxides or ammonia during the gasification process. Note that the ratio of the carbohydrate and protein food wastes in the overall food wastes collected from the canteen is nearly 4:1 (65% wt. vs. 15% wt.), therefore, a mixture ratio of 4:1 also allows the full utilization of carbohydrate and protein food wastes in the study. Then, the carbohydrate-protein complex was further mixed with woodchips in a ratio of 1:4 for the co-gasification experiments. The fat food wastes were not used because of its high oil content which may cause potential technical problems (e.g., blockage) for gasification (Abe et al., 2007). The co-gasification experiments were conducted in the fixed-bed downdraft gasifier (All Power Labs) with a capacity of 10kg/hr.

A schematic diagram of the gasifier is shown by Figure 1. The experimental procedure is the same as that in the study of Ong et al., (2015). During the experiments, a mixture of waste and wood chips were firstly poured into the hopper (1). The hopper was then gas-tight sealed and a cold run was performed to ensure there is no gas leakage. The feedstock entered a heat exchanger drying bucket (2) where it was pre-heated by hot producer gas. The feedstock was then fed into the pyro-coil (4) via a motorized screw feeder (3). When the pyro-coil and reactor (5) were completely filled, a level switch incorporated onto the pyro-coil lid switched off the screw feeder. Pyrolysis, combustion, and gasification reactions are taking place in the reactor (1). As the pressure in the reactor was below the atmospheric pressure, ambient air was sucked into a heat exchanger jacket where it was pre-heated by hot producer gases before entering the combustion zone of the reactor. The hot producer gas that left the reactor went through the drying bucket, heating up the feedstock. Subsequently, the producer gas left the cyclone (6) and drying bucket (2) in sequence and passed through the gas filter (10) before entering the flare (12). The experiments were initiated by switching on the vacuum pump and igniting an auxiliary fuel (kerosene) through an ignition port located on the side-wall of the reactor. After ignition, the reactor is left to reach temperatures of 800 – 1000 ˚C at the combustion and gasification zones, followed by the ignition of the flare (12). The feedstock feeding rate was around 10 kg/hr. The producer gas was sampled continuously using a non-dispersive infrared thermal conductivity detector (NDIR – TCD) via a Gasboard 3100P gas analyzer (8) (Wuhan Cubic Optoelectronics Co. Ltd.). The contents of CO, H_2, CO_2, CH_4, O_2, and C_nH_m in the producer gas and the corresponding lower heating values (LHV) were recorded. Prior to analysis, the producer gas was passed through a simple gas conditioning system (7) to remove water vapor, dust and tar. All the recorded data were sent to a Process Control Unit (PCU) and logged every second.

2.3 Cost-benefit Analysis

2.3.1 Scheme Proposal and Parameter Selection

Two gasification-based waste disposal schemes are proposed: (1) N (N=100, 500, and 1000) decentralized gasification stations are deployed with respect to population without differentiating the gasification of food wastes and sewage sludge; (2) each WRP has its own gasification station catering for the demand of its sewage sludge disposal while food wastes are gasified by other (N-4) gasification stations. The information about the sewage sludge production of each WRP is not available and is calculated by multiplying the annual sewage sludge production (98,500 tons) by its capacity fraction with respect to the total capacity of all four WRPs as shown in Table 1. A third scheme is based on incineration which is the primary waste treatment and disposal method in Singapore with four incineration plants. The diagrams of the three schemes are shown in e-supplement Figure 1. Cost-benefit analysis is conducted to compare the proposed gasification-based waste disposal schemes with the existing incineration-based one. In the cost-benefit analysis, the private cost involves the initial investment such as the construction of facilities and land cost, and operating and maintenance (O&M) cost (e.g., staff salary, training program, etc.). The transport cost is not considered in this work due to the lacking of precise information. Actually, existing studies (Bernstad and la Cour Jansen, 2011; Sundqvist et al., 2002) have suggested that transportation would generally have limited effect on the results of strategy studies comparing different waste treatment options. External costs are considered based on the monetary valuation of damages caused by the pollutants emitted during a process.

Uncertainties always exist in the cost-benefit analysis due to the variability and availability of considered factors (Graham, 1981). The data of some factors are from existing studies. The potential uncertainties of the data were generally not quantified by the original studies, but they still serve as important references for estimating the potential range of the factors. To further account for the uncertainties, triangular distributions would be assumed for the potentially variable parameters and modeled by Monte Carlo simulation with a total of 104 iterations. Triangular distributions are widely assumed in decision-making related researches and have been employed in existing cost-benefit analysis of emission related projects (Barrett et al., 2012; Withers et al., 2014).

The cost of a gasification system generating electricity was suggested to be 1500 US$/kW in 2007 (Abe et al., 2007). Recently, Suramaythangkoor and Gheewala (2010) summarized the investment of gasification system to be 45-56 MBaht/MW (or 1592 US$/kW for an exchange rate of 0.028 US$/Baht). In this work, we applied a triangular distribution with a lower limit, mode, and upper limit of 1000, 1500 and 2000 US$/kW, respectively, for the construction cost of each gasification station in 2007. The cost was further updated to the current year (2016) using Chemical Engineering Plant Cost Index (CEPCI) as

$$\mathrm{Cost}_i = \mathrm{Cost}_j(\mathrm{CEPCI}_i/\mathrm{CEPCI}_j) \tag{2}$$

where i and j denote the current year (2016) and base year (2007), respectively. However, considering that the annual value of CEPCI for 2016 was not available, the annual value for 2015, 556.8, was used to represent the current year . The annual value of CEPCI for 2007 was 525.4 . In view of the scale dependence of facility cost, further scaling is done by Eq. (3) (Holmgren et al., 2015)

$$\mathrm{Cost}_k = \mathrm{Cost}_i(\mathrm{S}_k/\mathrm{S}_i)^f \tag{3}$$

where S_k and S_i denote the designed facility capacity and base facility capacity, respectively. The base facility capacity was set to be 1 MW according to the study of (Suramaythangkoor and Gheewala, 2010). f is the scaling factor typically ranging from 0.6 to 0.8 and $f = 0.7$ was applied in this work. For the land cost, a triangular distribution with a lower limit, mode, and upper limit of 500. 1500, 2500 US$/m^2, respectively, is assumed based on the price range data of Singapore's public housing (HDB, 2016). The pilot-scale gasification system in our experiments has a consumption rate of 0.12 ton/day for daily operation duration of 12 hours and occupies an area of around 4 m^2. It is assumed that the occupied area of each gasification station will be linearly proportional to 4 m^2 in terms of the consumption rate. The ratio between the monthly O&M cost and the capital cost (construction cost + land cost) was about 0.01 for a system applying the integrated gasification combined cycle (IGCC) technology (Christou et al., 2008). In this work, the ratio between the monthly O&M cost and capital cost is assumed to be a triangular distribution with a lower limit, mode, and upper limit of 0.008, 0.014, and 0.02, respectively. The efficiency of the IGCC technology was identified to range from 40% to 55% by the study of Christou et al. (2008). In the study of Münster and Lund (2010), an efficiency of 47% was recognized for the case of electricity production from syngas. However, these efficiencies correspond to large plants of capacities over hundreds of megawatt. For decentralized gasification stations with much smaller capacities, the efficiency would be significantly smaller. Unfortunately, there is no definite relationship between the efficiency and scale of gasification systems, and a triangular distribution with a lower limit, mode, and upper limit of 20%, 30%, and 40%, respectively, is assumed in the cost-benefit analysis. The cost of incineration plants ranges from 3500 to 8200 US$/kW calculated based on the investment and capacity information of the four incineration plants in Singapore (NEA, 2016a). A triangular distribution with a lower limit, mode, and upper limit of 3000, 6000, and 9000 US$/kW is assumed for the capital cost of incineration plants. Similar to the gasification-based schemes, cost updates in terms of year and facility capacity were conducted based on Eq. (1) and Eq. (2), respectively. The base year in Eq. (1) was set to be 2007, which corresponds to the period when the newest incineration plant in Singapore was developed. The base facility capacity was set to be 40 MW corresponding to the average capacity of the four incineration plants in Singapore. The ratio between the monthly O&M cost and capital cost is set to be triangularly distributed with a lower limit, mode and upper limit of 0.01, 0.03, and 0.05 based on the study by Kannan et al. (2007) where a ratio of 0.025 was used. An electricity efficiency of 19.5% was used previously for the incineration technology (Münster and Lund, 2010). Correspondingly, a triangular distribution with a lower limit, mode, and upper limit of 15%, 20%, and 25%, respectively, is used.

Two pollutants, dioxins, and CO_2, are considered for the external cost. Dioxins are one type of the major by-products of waste incineration and pose a great threat to human health. For example, it has been recognized by World Health Organization (WHO) as one of the potential causes for human reproductive and developmental problems, damage to the immune system, and even cancer (Ahlborg et al., 1994). CO_2 is widely considered in the environmental cost of cost-benefit analysis in view of its global warming potentials. Based on the study by Rabl et al. (2008), a triangular distribution with the lower limit, mode, and upper limit of 1.13×10^{7}, 1.47×10^{8}, and 2.82×10^{8} US$/kg, respectively, is assumed for Dioxins cost; a triangular distribution with the lower limit, mode, and upper limit of 6.4×10^{-4}, 1.18×10^{-2} and 2.3×10^{-2} US$/kg, respectively, is assumed for the CO_2 cost. The CO_2 emissions from the gasification of food waste and sewage sludge are obtained by our experiments and reported below (see section 3.2). The Dioxins emission from the gasification is zero, as the oxygen-deficient environment in gasifier suppresses the formation of Dioxins. The CO_2 and dioxins emissions from the incineration of solid waste are set to be 861800 and 5.15×10^{-7} g per tons of waste (Rabl et al., 2008), respectively.

Both wood and horticulture materials have been found to be suitable as co-gasification agents. A great amount of wood (367900 tons) and horticultural (362000 tons) wastes is produced per year in Singapore (NEA, 2016b). These wastes could be used as co-gasification agents to benefit both waste disposal management and the profitability of the gasification-based schemes. The total amount of wood and horticultural wastes could satisfy the co-gasification demand of food waste and sewage sludge, considering that around 70 wt.% of moisture content in the food waste and sewage sludge is removed before co-gasification. Hence, in the gasification-based schemes, wood and horticultural wastes are used as co-gasification agent to save cost and increase the waste income by refuse disposal fee. Another cost would be incurred by the disposal of waste treatment residues, i.e. ash for incineration and biochar for gasification. The residues are disposed of by landfilling and are subject to refuse disposal fee. The refuse disposal fee is now 56.5 US$/ton (NEA, 2016a). Considering the potential fluctuation, a triangular distribution with a lower limit, mode, and upper limit of 50, 60, and 70 US$/ton, respectively, is assumed for the refuse disposal fee. For incineration, the mass of ash is calculated based on the ash content data from the approximate analysis (see Table 3). In our gasification experiments, the weight of residues was measured to be around 10% of consumed feedstocks which is applied in the cost-benefit analysis for gasification. It should be noted that the cost of energy required to drying the co-gasification feedstocks is neglected, because we presume solar drying is employed for the pretreatment process and relevant operating costs are considered in the overall O&M cost.

The direct profits from the waste treatment schemes include selling electricity (energy income) and refuse disposal fees (waste income). The tariff of electricity for low tension supplied varied from 0.14 to 0.19 US$/kWh from January 2013 to January 2016 (SP, 2016). A triangular distribution with a lower limit, mode, and upper limit of 0.1, 0.2, and 0.3 US$/kWh, respectively, is assumed for the tariff of electricity. The waste income is estimated by the product of net waste handled by incineration or gasification (i.e. excluding the mass of residue to be landfilled) and the refuse disposal fee.

Three indicators (net present value (NPV), benefit-cost ratio (BCR), and internal rate of return (IRR)) are calculated in the cost-benefit analysis. NPV is calculated as

$$\mathrm{NPV} = \sum_{t}^{LT} \frac{C_{it}}{(1+r)^t} - C_0 \qquad (4)$$

where C_t is the net cash inflow during a year *t*; C_0 is the total initial investment including the construction and land costs; *LT*=20 years denotes the life time of facilities; r is the discount rate. A near-zero discount rate means that the cost of borrowing from the future is low, and future benefits and costs are worth about the same as today (Quah and Toh, 2011). The potential discount rate has been suggested to be in a range of 5% to 10% (Ertürk, 2012), while another study (Manioğlu and Yılmaz, 2006) used 15% for economic analysis. To examine the potential impact of the discount rate on the cost benefit analysis, the discount rate is assumed to have a triangular distribution with a lower limit, mode, and upper limit of 1%, 8%, and 15%, respectively. BCR is calculated as

$$\mathrm{BCR} = \mathrm{NPV} / \left(\sum_{t}^{LT} \frac{C_{et}}{(1+r)^t} + C_0 \right) \qquad (5)$$

where C_{et} is the expenditure cost (O&M and emission costs) during a year t. IRR corresponds to a discount rate that leads to a zero NPV. IRR could not be calculated analytically as shown by Eq. (4) and is calculated using an algorithm provided by Matlab (Matlab R2014a). A summary of considered factors is listed in Table 2.

3. RESULTS AND DISCUSSION

3.1 Comparison between Food Wastes and Sewage Sludge

The moisture of raw materials, proximate and ultimate compositions, and HHVs of feedstocks are listed in Table 3. The received basis moisture of food wastes is about 20% lower than that of sewage sludge, both of which are significantly higher than the suggested limit of 25 wt.% for gasification (Puig-Arnavat et al., 2010). The moisture content of food wastes and sewage sludge dropped to around 10 wt.% after drying. The fixed carbon contents for carbohydrate and protein food waste are around 15% and comparable to that for sewage sludge and woodchips. The ash contents of food wastes are much lower than that of sewage sludge and comparable to woodchips, with the carbohydrate food waste of the lowest ash content of 2.7 wt.%. High ash contents in sewage sludge may pose problems such as slagging and clinker formation in the reactor, making the gasification process unstable (Ong et al., 2015). This suggests that food waste is more favorable than sewage sludge for co-gasifying with woodchips, in terms of the amount of ash residue. Based on the mass fractions of carbon, hydrogen, oxygen, and nitrogen, the equivalence ratios for the co-gasification of food wastes and sewage sludge were calculated to be 0.31 and 0.32, respectively. The HHV of sewage sludge is smaller than both the carbohydrate and protein food wastes. This could be explained by the unified correlation, Eq. (1) together with the fact that the sewage sludge has lower carbon content while a higher ash content than the food wastes. The HHV of protein food waste is the highest, suggesting that it is a favorable feedstock for gasification in terms of energy content. However, the nitrogen content is significantly richer in the protein food wastes than in the carbohydrate food wastes, consistent with their respective nutrient compositions. As mentioned in section 2.2.2, the high nitrogen content in the protein food wastes suggests a potential environmental concern of the emission of nitrogen oxides or ammonia. On the whole, in view of the potential energy output and gas pollutant emission, a mixture of carbohydrate and protein food wastes is a good choice for gasification. The metallic element contents in the feedstocks are listed in Table 4. It is shown that the carbohydrate food waste has the highest Ca content, which is almost double of sewage sludge and triple of woodchips. The calcium content in the wastes may be transformed to quicklime (CaO) during the calcination of biochar, which could be used as catalysts in various processes. Compared to the food wastes, the sewage sludge has a significant more amount of Cu and Fe, which may have come from pipeline corrosion during the transport process of wastewater. Recently, the gasification bottom ash (or biochar) of various types of wastes have been converted to fertilizers or soil conditioners for agricultural application (Yang et al. 2016). However, the significant Cu and Fe contents in the sewage sludge would limit its application as fertilizers. Additional measures (e.g., leaching) need to be taken to remove the metallic contents before practical agricultural applications.

3.2 Gas Composition and LHV

The producer gas compositions and LHVs for the cases of food waste and sewage sludge co-gasification are given by Table 5 where the data for the case of pure woodchips gasification from the study of Ong et al., (2015) is also added. The co-gasification of food wastes produced a higher volume fraction of CO than sewage sludge, but a lower volume fraction than pure woodchips. On the other hand, the volume fraction of H_2 for the case of food waste co-gasification is the lowest. The volume fraction of syngas ($CO+H_2$) generated during the co-gasification of food wastes is higher than that of sewage sludge (32.9 vol.% vs. 32.4 vol.%). Both the food waste and sewage sludge co-gasification generated a smaller amount of CO and H_2, and thus a lower volume fraction of syngas than the gasification of pure woodchips. The volume fraction of CO_2 in the producer gas from the co-gasification of food wastes is much higher than that of sewage sludge and the gasification of pure woodchips. The food waste co-gasification produced a higher volume fraction of CH_4 than sewage sludge and pure woodchips. The production of CH_4 would increase the energy content of producer gas. Despite the amount of CH_4 is only one sixth of that of CO_2 in the producer gas, CH_4 serves as a potential source of greenhouse gas (GHG) in terms of itself or its combustion product CO_2. The LHV of the producer gas from the food waste co-gasification is similar to that from pure woodchip gasification, both of which are slightly higher than that from the sewage sludge co-gasification. The syngas ($CO+H_2$) yield rate for the food waste co-gasification is smaller than that for the pure woodchip gasification (2.303×10^{-3} m^3/s vs. 2.408×10^{-3} m^3/s based on the flow rate of 7×10^{-3} m^3/s), meaning a smaller LHV accounted for by the syngas for food wastes than pure woodchips. However, the higher CH_4 yield rate (0.175×10^{-3} m^3/s vs. 0.119×10^{-3} m^3/s based on the flow rate of 7×10^{-3} m^3/s) for the food waste co-gasification than the woodchip gasification make up the lower LHV accounted for by CO and H_2, because CH_4 has a much higher LHV in the unit of MJ/Nm^3 than syngas (Ghenai, 2010). The higher LHV suggests higher energy output estimation for the case of food wastes than that of sewage sludge, consistent with the HHV results (the feedstock HHVs for the cases of food wastes and sewage sludge are 18.16 and

17.50 MJ/kg, respectively based on Table 3). On the whole, the mixture of carbohydrate and protein food wastes potentially serves as a better co-gasification agent, compared to sewage sludge in terms of energy output. The energy output of the mixture of food waste and pure woodchips is similar to that of the gasification of pure woodchips in view of the similar LHV and HHV data, feedstock feeding rate (10 kg/hr), and gas yield rate (7×10^{-3} m^3/s).

To achieve fair comparison, it is important to have a common basis such as the similar operational conditions used in this work. However, it should be noted that another common basis could be optimum operational conditions which are not necessarily similar to each other for different wastes. Hence, comparing the effectiveness of the co-gasification of food waste and sewage sludge could be based on their respective optimum operational conditions, and much more research is needed to find the optimum conditions in the future. Once the co-gasification data under the optimum operational conditions are available, they could also be incorporated into such a cost-benefit analysis as proposed by this work to evaluate relevant economics.

3.3 Cost-benefit Analysis

3.3.1 Cost and benefit components

The mass fraction, moisture content, HHV, and CO_2 data from experiments is used for the cost-benefit analysis. There is a distribution for each component corresponding to the Monte Carlo simulation. The mean and standard deviation of each distribution are calculated and listed in Table 6. It is shown that the construction and O&M costs of scheme 1 and 2 are increased by about 50% and 20% as the number of gasification stations increase from 100 to 500 and from 500 to 1000, respectively. Other cost and benefit components (land cost, energy income, waste income, and CO_2 emission cost) are merely affected by the number of stations in the gasification-based scheme 1 and 2. This is because these components are generally dependent on the total amount of wastes disposed of which is not affected by the number of gasification stations. Similarly, there is a limited difference of these components between scheme 1 and 2. For the same number of stations, the construction and O&M costs in scheme 2 are 2% - 5% lower than those in scheme 1, suggesting that handling food waste and sewage sludge separately is more economic. The difference between the gasification-based scheme 1 and 2, and incineration-based scheme 3, is significant. Specifically, the capital cost of scheme 3 is about 130%, 85%, and 70% of that of scheme 1 with 100, 500, and 1000 stations, respectively. Note that around 2 times larger amount of wastes (wood and horticultural wastes were added as co-gasification agents in scheme 1 and 2) are disposed of by scheme 1 and 2 increasing the overall capacity of gasification stations. The O&M cost of scheme 3 is about 150% - 290% of that of scheme 1 and 2. The energy income, however, of scheme 3 is about one order of magnitude less than that of scheme 1 and 2, due to (1) the lower efficiency of the incineration-based scheme compared to the gasification-based schemes and (2) the added mass of feedstocks by co-gasification agents (i.e. wood and horticulture wastes). The much higher O&M cost and lower energy incomes make the incineration-based scheme 3 less profitable compared to the gasification-based scheme 1 and 2. The CO_2 emission costs of scheme 1 and 2 are about double of scheme 3 and more than half of the costs are corresponding to the added co-gasification agents in scheme 1 and 2. The co-gasification agents are not needed to be considered in the incineration scheme, because of the specific focus on disposing of food waste and sewage sludge. However, if they are also included in the incineration, the CO_2 emission from scheme 3 would be around one order of magnitude higher than that of scheme 1 and 2 due to the higher CO_2 emission per unit feedstocks for incineration (Table 2 and Table 6). From a point of view of CO_2 emission per unit mass of feedstocks, the gasification-based scheme could be more environmentally friendly.

For the gasification-based schemes, the O&M costs are the highest among the cost components and around double of the capital costs. The energy income is about 370% of the waste income and overtakes the O&M costs for the cases of 100 and 500 stations. The environmental externality, i.e. CO_2 emission cost, is generally two orders of magnitude lower than the other cost and benefit components and thus is negligible. This may justify the negligence of other potential pollutants from co-gasification whose volume fraction is significantly less than CO_2. For the incineration-based schemes, the O&M cost is the highest among all the components and is about 6 times the summation of energy and waste incomes. The environmental externalities are negligible compared to the other cost and benefit components for the incineration-based scheme as well, with the Dioxins emission cost two orders of magnitude lower than the CO_2 one.

3.3.2 Net present value (NPV)

The distributions of NPVs of different schemes are shown in Figure 2 ((a), (b), and (c) for scheme 1, 2, and 3, respectively) with the means and standard deviations shown as insets. Only the cases of 100 stations are shown for scheme 1 and 2 because the shape of the distributions for the cases of 500 and 1000 stations are similar to that of 100 stations but with the means shift to the left. Consistent with Table 6, the NPV distribution of scheme 1 is similar to that of scheme 2 in shape but has a smaller mean. The positive values of NPV mean that the gasification-based schemes could potentially be economically efficient and viable for disposing of food waste and sewage sludge. Statistically, the fraction of positive

NPV is more than 80%, which means that there is more than 80% of chance for the gasification-based schemes to be profitable. On the other hand, the values of the NPV distribution for the incineration-based scheme 3 are all negative with a mean of -4.48 billion over a life-time of 20 years, suggesting that the incineration-based scheme 3 is not financially viable. In the calculation of NPV for scheme 3, the construction cost has been included as well. In view of the fact that Singapore already has four incineration plants, we recalculate the NPV of scheme 3 by disregarding the construction cost. However, in this case the NPV distribution only shifts to the right to a limited extent and the values of NPV distribution are still all negative, because the limited magnitude of initial construction cost compared to the overwhelming O&M cost over a course of 20 years as shown by Table 6 (1.06×10^9 vs. 3.90×10^9 US$). Hence, the gasification-based schemes are more viable than the incineration-based one, no matter existing or new incineration plants are considered.

3.3.3 Benefit-cost ratio (BCR)

The distributions of BCRs of different schemes are shown in Figure 3 ((a), (b), and c) for scheme 1, 2, and 3, respectively) with the means and standard deviations shown as insets. Only the cases of 100 stations are shown for scheme 1 and 2. Similar to the case of NPV, the distribution of scheme 1 is similar to that of scheme 2 in shape but has a smaller mean. The mean BCRs of 0.35 and 0.37 suggest that the mean net profit would be around 35% and 37% of the overall expenditure for scheme 1 and 2, respectively. The BCR distribution of scheme 3 has a mean of -0.87, meaning that the income from incineration could only cover about 13% of the overall expenditure, re-emphasizing the need to reduce the construction and O&M costs and increase the efficiency of the incineration-based scheme.

3.3.4 Internal rate of return (IRR)

The distributions of IRRs of different schemes are shown in Figure 4 with the means and standard deviations shown as insets. Only the cases of 100 stations are shown for scheme 1 and 2. The IRRs of scheme 1 and 2 are similar to each other and the fraction of positive IRRs is more than 95%, which suggests the potential for the gasification-based schemes to be profitable. In contrast, the values of IRR for the incineration-based scheme 3 are all negative with a mean of -1.63, meaning the benefits from scheme 3 could not repay the investment cost during the designated life-cycle. The IRR results show that the gasification-based schemes are better candidates than the incineration-based scheme for the disposal of food wastes and sewage sludge, consistent with the results based on the data of NPV and BCR shown above.

Note that a service time of 20 years is adopted in the above analysis. Longer service time would increase the NPV and BCR of schemes and favor the deployment. For example, increasing the service time from 20 to 25 years increases the mean NPV, BCR, and IRR from 7.95×10^8 US$, 0.35, and 0.19 to 9.63×10^8 US$, 0.39, and 0.194, respectively, for scheme 1 (100 stations). Based on Table 6, the average CO_2 emission cost per unit food wastes is calculated to be 0.0043 US$/kg, which is larger than that per unit sewage sludge, 0.0038 US$/kg. However, the average energy income per unit food wastes is 0.30 US$/kg, which is larger than that per unit sewage sludge, 0.25 US$/kg. As a result, food wastes would be generally more financially viable than sewage sludge for the co-gasification-based disposal management. The dispersion of NPV and BCR distributions is closely associated with the dispersion of triangular distributions employed. In this work, triangular distributions are assumed to consider the variation of potentially variable parameters, and the parameter selection is generally based on the existing data. The calculated distributions of NPV, BCR, and IRR should be indicative of their potential ranges for practical deployment of relevant waste disposal schemes. This method could be directly applied to provide more accurate predictions, whenever more accurate distributions of parameters are available. The analysis is specifically for the food wastes and sewage sludge in Singapore. However, the method could be easily extended to the cases of other megacities and types of wastes in the future, if relevant input parameters are accumulated. Gasification residues have the potential to be turned into various high-value commercial products such as soil conditioners and fertilizers, which may add extra cost and benefit components to the whole analysis. Due to limited data, this part of cost and benefit is not included in the current analysis and should be explored in the future. Finally, the feasibility comparison among the different waste disposal schemes is conducted from an economical point of view in this work. The conclusions may be changed when the comparison is subjected to other points of view (e.g., urban planning) or analysis methods (e.g., life cycle analysis).

3.4 Sensitivity Analysis

The design-of-experiments (DOE) method (Montgomery, 2008) is used to analyze the sensitivity of cost-benefit parameters (represented by NPV) to various input factors. Six input factors (i.e. a 2^6 factorial design) were analyzed, including (A) construction cost, (B) the ratio between O&M cost and capital cost, (C) electricity efficiency, (D) electricity tariff, (E) refuse disposal fee, and (F) discount rate. In the factorial design, the low and high levels of the input factors are ±20% of the nominal values that correspond to the modes of the factor’s triangular distributions listed in Table 2. The factors related to emission costs are not analyzed because the emission costs have limited contribution to the overall NPVs as shown above. The modes of the triangular distributions of emission costs are used in the sensitivity analysis. For scheme 1 and 2, the sensitivity analysis is based on their cases of 100 stations. The main effects and interactions of the factors are calculated by

$$Eff = \frac{1}{2^5}\sum_{j=1}^{64} \pm \mathrm{NPV}_{i,j} \tag{6}$$

where $\pm$ corresponds to the (+/-) signs of each main effect and interaction for each NPV. Normal probability plots of the main effects and interactions are used to display the significance of the factors and their interactions on the NPVs of the three schemes (Figure 5). In a normal probability plot, the factors or interactions that have insignificant effects on the response behave like a small, normally distributed random errors and follow the straight dash lines, whereas the factors or interactions that deviate away from the line would have significant effects on the response (Montgomery, 2008). Generally, the further away from the straight line, the more significant effect a factor or an interaction has on the response.

Figure 5 shows that eight interactions EF, BEF, CEF, DEF, BCEF, BDEF, CDEF, BCDEF have the most significant effect in all the three schemes, suggesting the effects of (B) the ratio between O&M cost and capital cost, (C) electricity efficiency, (D) electricity tariff, and (E) refuse disposal fee depend on the level of (F) discount rate. This is because the discount rate serves as an overall adjustment factor in the calculation of NPVs as shown in Eq. (4). In scheme 1 and 2, the significant interactions have a positive relationship with the NPV, compared to the inverse relationship in scheme 3. In terms of the absolute value, the effects of the significant interactions are the highest in scheme 3 (-8.78×10^9 US$), followed by scheme 2 (1.58×109 US$) and scheme 1 (1.51×109 US$), respectively. The NPV is also moderately sensitive to the main effects (A) construction cost, (B) the ratio between O&M cost and capital cost, (C) electricity efficiency, and (D) electricity tariff for both scheme 1 and 2, whereas it is only moderately sensitive to the main effects (A) construction cost and (B) the ratio between O&M cost and capital cost in scheme 3. This is consistent with the data in Table 6 which shows that the construction cost, O&M cost, and energy income constitute the major part of the overall cost and benefit for scheme 1 and 2, while only the construction and O&M costs constitute a major part of the overall cost and benefit. On the whole, in view of the fact that the discount rate and electricity tariff is generally less controllable, it would be more favorable to reduce the construction and O&M costs, and increase the electricity efficiency in order to improve the economics of the gasification-based schemes. The methods of lowering the construction and O&M costs should be paid special attention for the incineration-based scheme.

4. CONCLUSIONS

Food wastes are more favorable than sewage sludge for co-gasification in terms of residue generation and energy output. Two decentralized gasification-based waste disposal schemes were proposed towards the management of the sewage sludge and food wastes in Singapore. Using the Monte Carlo simulation-based cost-benefit analysis, it was found that the gasification-based schemes are financially superior to the incineration-based scheme. Sensitivity analysis shows that reducing the construction and O&M costs (for both gasification- and incineration-based schemes), and increasing the electricity efficiency (for gasification-based schemes) would be effective to improve the economics of the schemes.

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REFERENCES

1. Abe, H., Katayama, A., Sah, B.P., Toriu, T., Samy, S., Pheach, P., Adams, M.A., Grierson, P.F. 2007. Potential for rural electrification based on biomass gasification in Cambodia. *Biomass Bioenerg*, 31(9), 656-664.
2. Ahlborg, U.G., Becking, G., Birnbaum, L., Brouwer, A.a., Derks, H., Feeley, M., Golor, G., Hanberg, A., Larsen, J., Liem, A. 1994. Toxic equivalency factors for dioxin-like PCBs: Report on WHO-ECEH and IPCS consultation, December 1993. *Chemosphere*, 28(6), 1049-1067.
3. Barrett, S.R., Yim, S.H., Gilmore, C.K., Murray, L.T., Kuhn, S.R., Tai, A.P., Yantosca, R.M., Byun, D.W., Ngan, F., Li, X. 2012. Public health, climate, and economic impacts of desulfurizing jet fuel. *Environ. Sci. Technol.*, 46(8), 4275-4282.
4. Baysal, T., Ozbalta, N., Gokbulut, S., Capar, B., Tastan, O., Gurlek, G. 2015. Investigation of effects of various drying methods on the quality characteristics of apple slices and energy efficiency. *J Therm. Sci. Tech.-JPN*, 35(1), 135-144.
5. Bernstad, A., la Cour Jansen, J. 2011. A life cycle approach to the management of household food waste–a Swedish full-scale case study. *Waste Manag.*, 31(8), 1879-1896.
6. Buragohain, B., Mahanta, P., Moholkar, V.S. 2010. Biomass gasification for decentralized power generation: The Indian perspective. *Renew. Sust. Energ. Rev.*, 14(1), 73-92.
7. Channiwala, S., Parikh, P. 2002. A unified correlation for estimating HHV of solid, liquid and gaseous fuels. *Fuel*, 81(8), 1051-1063.
8. Chemical Engineering, 2016. Plant cost index. Available at:<http://www.chemengonline.com/pci-home> (accessed June 2016).
9. Christou, C., Hadjipaschalis, I., Poullikkas, A. 2008. Assessment of integrated gasification combined cycle technology competitiveness. *Renew. Sust. Energ. Rev.*, 12(9), 2459-2471.
10. Ertürk, M. 2012. The evaluation of feed-in tariff regulation of Turkey for onshore wind energy based on the economic analysis. *Energ. Policy*, 45, 359-367.

11. Ghenai, C. 2010. Combustion of syngas fuel in gas turbine can combustor. *Adv. Mech. Eng.*, 2, 342357.

12. Graham, D.A. 1981. Cost-benefit analysis under uncertainty. *Am. Econ. Rev.*, 71(4), 715-725.

13. Hamer, G. 2003. Solid waste treatment and disposal: effects on public health and environmental safety. *Biotechnol. Adv.*, 22(1), 71-79.

14. HDB, 2016. Key statistics, HDB annual report, 2014/2015. Available at:<http://www10.hdb.gov.sg/ebook/ar2015/key-statistics.html> (accessed March 2016).

15. Holmgren, K.M., Berntsson, T.S., Andersson, E., Rydberg, T. 2015. Perspectives on investment cost estimates for gasification-based biofuel production systems. *Chem. Eng. Trans.*, 45, 427-432.

16. Kannan, R., Leong, K., Osman, R., Ho, H. 2007. Life cycle energy, emissions and cost inventory of power generation technologies in Singapore. *Renew. Sust. Energ. Rev.*, 11(4), 702-715.

17. Koupaie, E.H., Leiva, M.B., Eskicioglu, C., Dutil, C. 2014. Mesophilic batch anaerobic co-digestion of fruit-juice industrial waste and municipal waste sludge: Process and cost-benefit analysis. *Bioresour. Technol.*, 152, 66-73.

18. Manara, P., Zabaniotou, A. 2012. Towards sewage sludge based biofuels via thermochemical conversion–a review. *Renew. Sust. Energ. Rev.*, 16(5), 2566-2582.

19. Manioğlu, G., Yılmaz, Z. 2006. Economic evaluation of the building envelope and operation period of heating system in terms of thermal comfort. *Energ Buildings*, 38(3), 266-272.

20. MEWR, 2016. Land. Available at:<https://www.mewr.gov.sg/docs/default-source/default-document-library/grab-our-research/052-063-land.pdf> (accessed March 2016).

21. Montgomery, D.C. 2008. *Design and analysis of experiments*. John Wiley & Sons.

Münster, M., Lund, H. 2010. Comparing waste-to-energy technologies by applying energy system analysis. *Waste Manag.*, 30(7), 1251-1263.

22. NEA, 2016a. Refuse disposal facility. Available at:<http://www.nea.gov.sg/energy-waste/waste-management/refuse-disposal-facility> (accessed March 2016).

23. NEA, 2016b. Waste statistics and overall recycling. Available at:<http://www.nea.gov.sg/energy-waste/waste-management/waste-statistics-and-overall-recycling> (accessed March 2016).

24. Ong, Z., Cheng, Y., Maneerung, T., Yao, Z., Tong, Y.W., Wang, C.H., Dai, Y. 2015. Co‐gasification of woody biomass and sewage sludge in a fixed‐bed downdraft gasifier. *AIChE J.*, 61(8), 2508-2521.

25. PUB, 2015. Overview of Singapore's deep tunnel sewage system. Available at:<http://www.pub.gov.sg/dtss/Documents/SWA_30Apr2015.pdf> (accessed March 2016).

26. Puig-Arnavat, M., Bruno, J.C., Coronas, A. 2010. Review and analysis of biomass gasification models. *Renew. Sust. Energ. Rev.*, 14(9), 2841-2851.

27. Quah, E., Toh, R. 2011. *Cost-benefit analysis: cases and materials*. Routledge.

28. Rabl, A., Spadaro, J.V., Zoughaib, A. 2008. Environmental impacts and costs of solid waste: a comparison of landfill and incineration. *Waste Manag. Res.*, 26(2), 147-162.

29. Ruffino, B., Campo, G., Genon, G., Lorenzi, E., Novarino, D., Scibilia, G., Zanetti, M. 2015. Improvement of anaerobic digestion of sewage sludge in a wastewater treatment plant by means of mechanical and thermal pre-treatments: Performance, energy and economical assessment. *Bioresour. Technol.*, 175, 298-308.

30. SP, 2016. Historical electricity tariff. Available at:<http://www.singaporepower.com.sg/irj/go/km/docs/wpccontent/Sites/SP%20Services/Site%20Content/Tariffs/documents/Historical%20Electricity%20Tariff.xls> (accessed March 2016).

31. Sundqvist, J., Baky, A., Clarlsson Reich, M., Eriksson, O., Granath, J. 2002. How should household waste be treated–evaluation of different treatment strategies. IVL, Swedish Environmental Institute.

32. Suramaythangkoor, T., Gheewala, S.H. 2010. Potential alternatives of heat and power technology application using rice straw in Thailand. *Appl. Energy*, 87(1), 128-133.

33. Tian, H., Gao, J., Lu, L., Zhao, D., Cheng, K., Qiu, P. 2012. Temporal trends and spatial variation characteristics of hazardous air pollutant emission inventory from municipal solid waste incineration in China. *Environ. Sci. Technol.*, 46(18), 10364-10371.

34. Withers, M.R., Malina, R., Gilmore, C.K., Gibbs, J.M., Trigg, C., Wolfe, P.J., Trivedi, P., Barrett, S.R. 2014. Economic and environmental assessment of liquefied natural gas as a supplemental aircraft fuel. *Prog. Aerosp. Sci.*, 66, 17-36.

35. Yang, Z., Koh, S.K., Ng, W.C., Lim, R.C., Tan, H.T., Tong, Y.W., Dai, Y., Chong, C.,

36. Wang, C.-H. 2016. Potential application of gasification to recycle food waste and rehabilitate acidic soil from secondary forests on degraded land in Southeast Asia. *J. Environ. Manag.*, 172, 40-48.

Captions for Figures

Figure 1. A schematic diagram of downdraft gasifier. 1 Hopper, 2 Heat exchanger drying bucket, 3 Motorized screw feeder, 4 Pyro-coil heat exchanger, 5 Rector, 6 Cyclone, 7 Gas conditioning system, 8 Gas analyzer, 9 Gas analysis system, 10 Filter, 11 Air blower, 12 Flare.

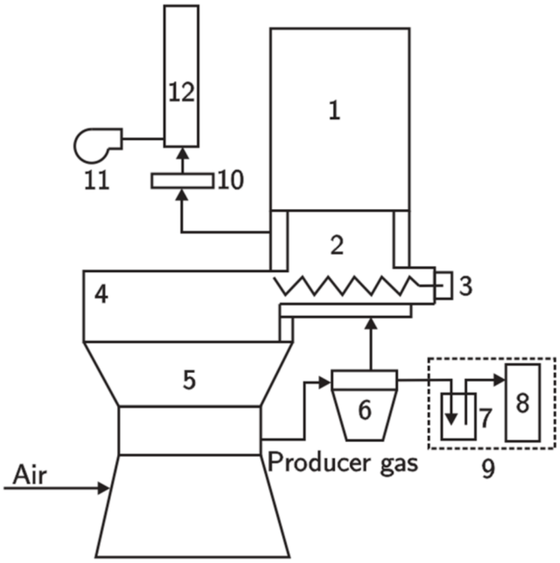
Figure 2. The distributions of NPV for (a) scheme 1, (b) scheme 2 of 100 stations, and (c) scheme 3, respectively. The means and standard deviations of the distributions are shown by the insets.

Figure 3. The distributions of BCR for (a) scheme 1, (b) scheme 2 of 100 stations, and (c) scheme 3, respectively. The means and standard deviations of the distributions are shown by the insets.

Figure 4. The distributions of IRR for scheme 1 (a), scheme 2 (b) of 100 stations, and scheme 3 (c), respectively. The means and standard deviations of the distributions are shown by the insets.

Figure 5. Normal probability plots of the effects for the 2^6 factorial design for scheme 1 (a), scheme 2 (b) of 100 stations, and scheme 3 (c), respectively. (A) Construction cost, (B) Ratio between O&M cost and capital cost, (C) Electricity efficiency, (D) Electricity tariff, (E) Refuse disposal fee, (F) Discount rate.

Figures



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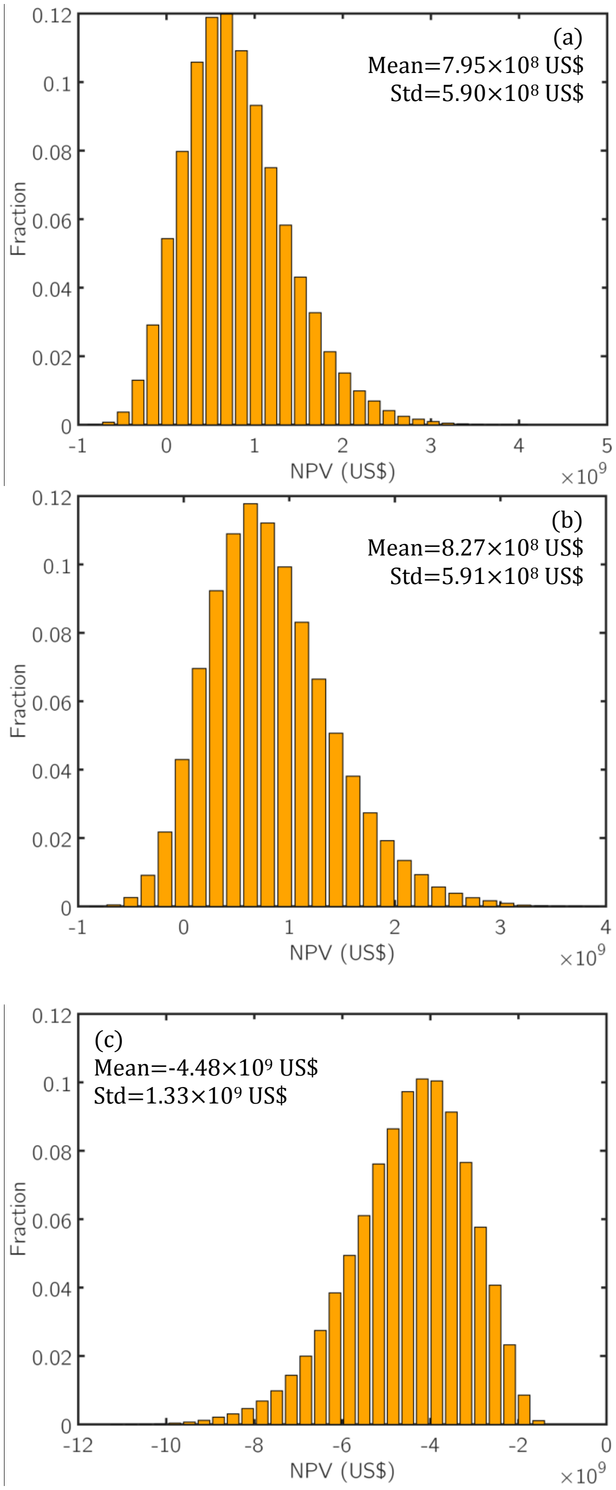

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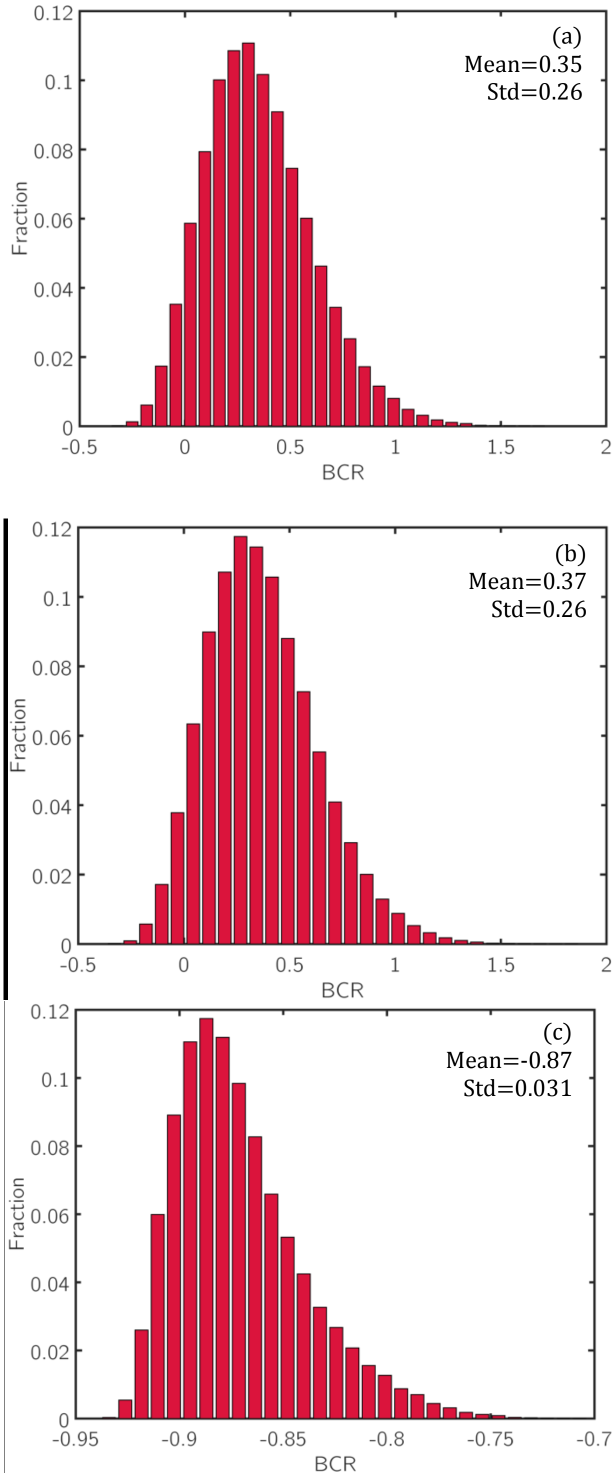

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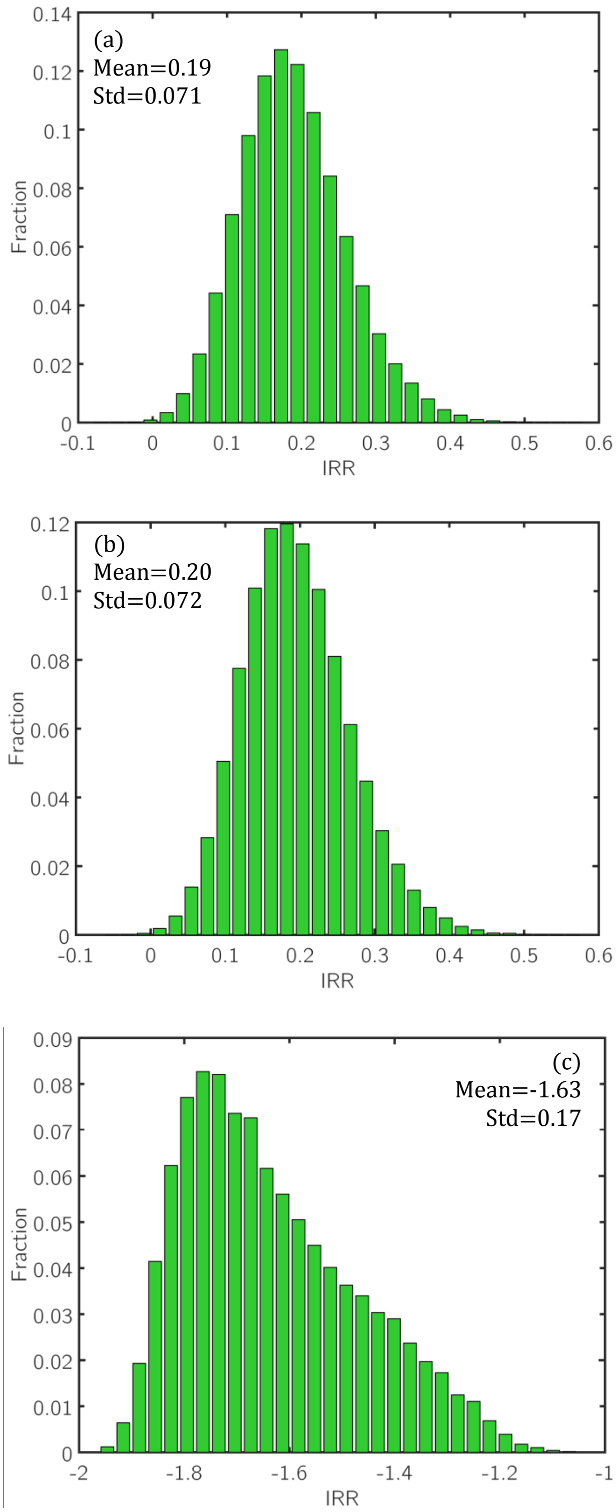

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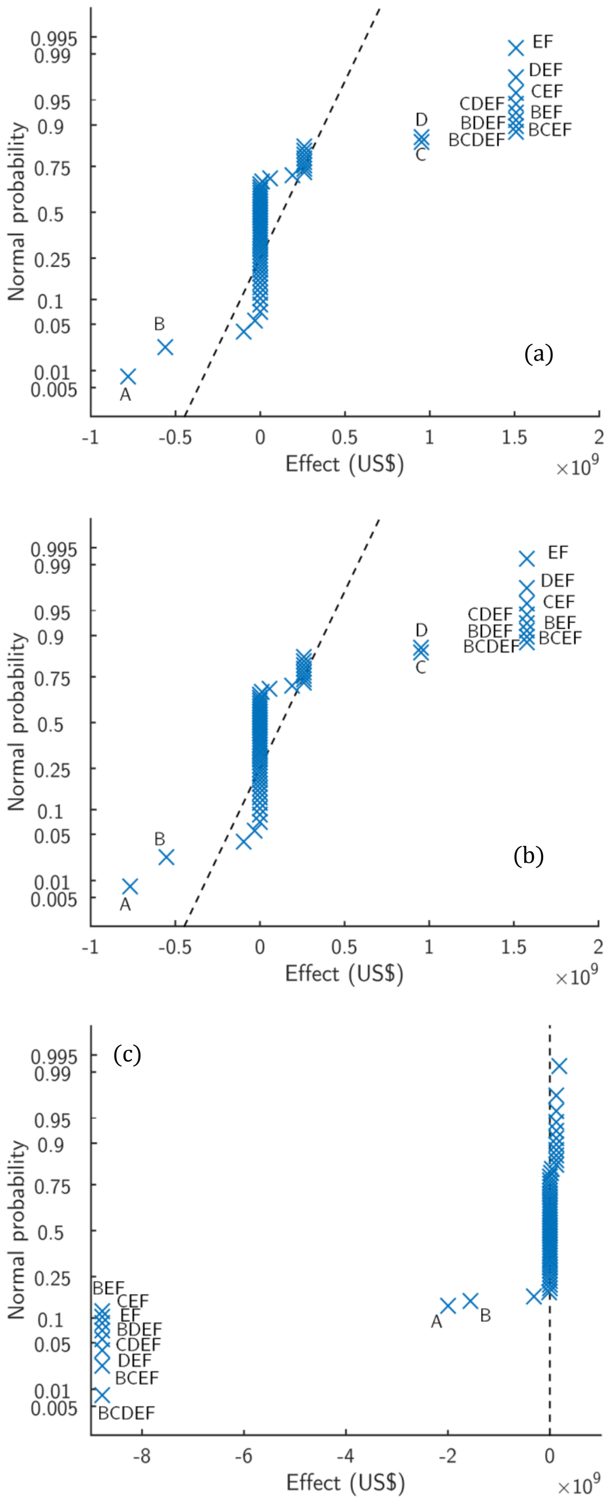

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Tables

Table 1. Information of existing WRPs in Singapore (PUB, 2015).

WRP	Capacity (million gallons per day)	Sewage sludge production (tons/day)
Jurong WRP	45	30.9*
Kranji WRP	34	23.4
Ulu Pandan WRP	79	54.3
Changi WRP	176	120.9

* The sewage sludge production is assumed to be proportional to the capacity.

Table 2. List of factors considered during the cost-benefit analysis

	Distribution	Parameters	Scheme 1§	Scheme 2§	Scheme 3¶
Construction cost (US$/kW)	Triangular	Lower	1000	1000	3000
		Mode	1500	1500	6000
		Upper	2000	2000	9000
Land cost (US$/m^2)	Triangular	Lower	500	500	
		Mode	1500	1500	-#
		Upper	2500	2500	
O & M cost/capital cost*	Triangular	Lower	0.008	0.008	0.01
		Mode	0.014	0.014	0.03
		Upper	0.02	0.02	0.05
Electricity efficiency	Triangular	Lower	20%	20%	15%
		Mode	30%	30%	20%
		Upper	40%	40%	25%
CO_2 emission cost (US$/kg)	Triangular	Lower	6.4×10^{-4}	6.4×10^{-4}	6.4×10^{-4}
		Mode	1.18×10^{-2}	1.18×10^{-2}	1.18×10^{-2}
		Upper	2.3×10^{-2}	2.3×10^{-2}	2.3×10^{-2}
Dioxins emission cost (US$/kg)	Triangular	Lower	1.13×10^{7}	1.13×10^{7}	1.13×10^{7}
		Mode	1.47×10^{8}	1.47×10^{8}	1.47×10^{8}
		Upper	2.82×10^{8}	2.82×10^{8}	2.82×10^{8}
CO_2 emission (g/ton)	-	-	-&	-&	861800
Dioxin emission (g/ton)	-	-	0	0	5.15×10^{-7}
Electricity tariff (US$/kWh)	Triangular	Lower	0.1	0.1	0.1
		Mode	0.2	0.2	0.2
		Upper	0.3	0.3	0.3
Refuse disposal fee	Triangular	Lower	50	50	50

(US$/ton)		Mode	60	60	60
		Upper	70	70	70
Facility life-time (years)	-	-	20	20	20
Discount rate	Triangular	Lower	1%	1%	1%
		Mode	8%	8%	8%
		Upper	15%	15%	15%

§ For the gasification-based schemes, the reference data are 1500 and 1592 US$/kW, 2200-3400 US$/m^2, 0.01, 40%-55%, 7.21×10^{-4} – 2.59×10^{-2} US$/kg, 0.14 to 0.19 US$/kWh, 56.5 US$/ton, and 5%-15% for the constructions cost, land cost, ratio between O & M cost and capital cost, electricity efficiency, CO_2 emission cost, electricity tariff, refuse disposal fee, and discount rate, respectively.

¶ For the incineration-based schemes, the reference data are 3500 and 8200 US$/kW, 0.025, 19.5%, 7.21×10^{-4} – 2.59×10^{-2} US$/kg, 1.13×10^{7} – 2.82×10^{8} US$/kg, 0.14 to 0.19 US$/kWh, 56.5 US$/ton, and 5%-15% for the constructions cost, ratio between O & M cost and capital cost, electricity efficiency, CO_2 emission cost, electricity tariff, refuse disposal fee, and discount rate, respectively.

* Capital cost consists of construction cost and land cost in this work.

& The CO_2 emission will be estimated using the volume fraction data of CO_2 in the producer gas, flow rate, and the consumption rate of feedstocks from the co-gasification experiments of this work.

For the incineration-based scheme, the construction cost considered included the land cost.

Table 3. The proximate and ultimate compositions and HHVs of feedstocks

		Carbohydrate food waste	Protein food waste	Sewage sludge*	Woodchips
Freeze-drying (received basis wt.%)	Moisture	66.8	53.8	80[&]	8.2
Proximate analysis (wt.%)	Moisture	10.8	12.2	7.6	8.2[#]
	Volatiles	70.7	67.6	50.8	69.2
	Fixed carbon	15.8	13.9	15.1	16.2
	Ash	2.7	6.3	26.5	6.4
Ultimate analysis (dry basis wt.%)	Carbon	41.8	48.2	35.0	44.2
	Hydrogen	6.2	7.1	4.8	6.0
	Oxygen	46.9	29.0	27.8	41.6
	Nitrogen	2.0	8.9	5.2	0.9
	Sulfur	<0.50	<0.50	1.7	1.0
HHV (MJ/kg)		17.01	21.99	14.7	18.2

* The average data of Ong et al. (2015) is used.

No drying pretreatment for woodchips

& Using the same freeze-drying method, the moisture content of the sewage sludge that was used in the study of Ong et al. (2015) was also measured in this work

Table 4. Metallic element contents (ppm) in carbohydrate and protein food wastes, sewage sludge, and woodchips.

	Cd	Co	Cr	Cu	Fe	Mn	Ca	Pb	Hg
Carbohydrate food waste	<0.1*	<0.1	<0.1	<0.1	<0.1	<0.1	12.5	<0.1	<0.1
Protein food waste	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	9.9	<0.1	<0.1
Sewage sludge&	-	<0.1	<0.1	1.9	8.5	<0.1	6.0	<0.1	-
Woodchips&	-	-	-	<0.1	0.2	<0.1	3.7	<0.1	<0.1

* denotes non-detectable, as the ICP analysis could not detect the content less than 0.1 ppm.

& The average data of Ong et al. (2015) is used.

Table 5. Comparison of producer gas composition and LHV among different feedstocks.

Feedstock		Food waste + Woodchips	Sewage sludge + Woodchips*	Pure woodchips*
Gas composition (vol.%)	CO	16.4	15.6	17.1
	CO_2	14.5	12.7	11.9
	CH_4	2.5	2.1	1.7
	H_2	16.5	16.8	17.3
	O_2	0.82	1.0	1.3
	Total	50.6	48.2	49.1
LHV (MJ/Nm^3)	-	4.8	4.5	4.7

* The data of Ong et al. (2015) is used.

Table 6. Summary of cost and benefit components.

Components (US$)	Scheme 1			Scheme 2			Scheme 3
Number of stations	100	500	1000	100	500	1000	-
Construction cost	7.31×10^{8} (9.97×10^{7})&	1.18×10^{9} (1.61×10^{8})	1.46×10^{9} (1.99×10^{8})	7.18×10^{8} (9.76×10^{7})	1.13×10^{9} (1.55×10^{8})	1.38×10^{9} (1.88×10^{8})	1.09×10^{9} (2.23×10^{8})
Land cost	1.09×10^{8} (2.97×10^{7})	1.09×10^{8} (2.97×10^{7})	1.09×10^{8} (2.98×10^{7})	1.09×10^{8} (2.97×10^{7})	1.09×10^{8} (2.97×10^{7})	1.09×10^{8} (2.97×10^{7})	-*
O&M cost	1.44×10^{9} (4.49×10^{8})	2.21×10^{9} (6.91×10^{8})	2.69×10^{9} (8.46×10^{8})	1.41×10^{9} (4.43×10^{8})	2.13×10^{9} (6.69×10^{8})	2.54×10^{9} (7.98×10^{8})	4.02×10^{9} (1.67×10^{9})
Energy income[#]	2.44×10^{9} (6.52×10^{8})	2.44×10^{9} (6.49×10^{8})	2.44×10^{9} (6.55×10^{8})	2.44×10^{9} (6.51×10^{8})	2.44×10^{9} (6.52×10^{8})	2.44×10^{9} (6.52×10^{8})	3.16×10^{8} (7.89×10^{7})
Waste income§	6.64×10^{8} (1.48×10^{8})	6.64×10^{8} (1.47×10^{8})	6.65×10^{8} (1.48×10^{8})	6.64×10^{8} (1.48×10^{8})	6.64×10^{8} (1.48×10^{8})	6.65×10^{8} (1.48×10^{8})	3.18×10^{8} (7.06×10^{7})
Carbon dioxide emission cost	3.42×10^{7} (1.55×10^{7})	3.43×10^{7} (1.55×10^{7})	3.44×10^{7} (1.56×10^{7})	3.43×10^{7} (1.56×10^{7})	3.43×10^{7} (1.55×10^{7})	3.43×10^{7} (1.56×10^{7})	1.65×10^{7} (7.49×10^{6})
Dioxins emission cost	0	0	0	0	0	0	1.22×10^{5} (5.46×10^{4})

* Land cost is incorporated in the construction cost.

& Data in the brackets are standard deviations.

\# Energy income refers to the one from electricity selling.

§ Waste income refers to the one from refuse disposal fee.