Life Cycle Assessment of Small-Scale Waste Gasification System in Vestmannaeyjar

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60 ECTS thesis submitted in partial fulfillment of a
Magister Scientiarum degree in Environment and Natural Resources

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Abstract

Waste generation and management are major issues confronting urban, small, and isolated communities in Iceland. Combined heat and power (CHP) gasification system provides the opportunity to convert waste wood to syngas, which can then be combusted to generate electricity and heat, thereby eliminating the need for landfilling or incineration. The study evaluates, for the first time, the life cycle GHG emissions and other environmental impacts of syngas production and energy generation from waste wood in comparison with incineration and landfill systems in the Vestmannaeyjar region in Iceland, using process life cycle assessment (LCA) methodology. Using a functional unit of 1 kg of waste wood treatment, the CML 2001 and CED methods were applied to evaluate the environmental impacts. The CHP power pallet gasifier owned by the University of Iceland, the proposed stepped heath incinerator to be constructed in Vestmannaeyjar, and the methane recovery landfill in Álfsnes were the case study´s systems employed for the life cycle modeling. The gasification process appeared to be the most environmentally beneficial concerning global warming and eight other environmental impact categories. The final conversion processes, along with electricity generation, and diesel and oil production were the most environmental hotspots. The incineration process generated the optimum net energy of 11,093 GJ per annum as compared to gasification (3,902 GJ) and the landfill (241 GJ). The sensitivity analysis highlights the vital role of systems´ parameters to the environmental impacts. Future studies can assess different scenarios of integrated biomass technology pathways with the aims of achieving both environmental sustainability and higher cleaner energy generation.

Útdráttur

Uppsöfnun sorps og sorpumhirða er flókið viðfangsefni fyrir minni sveitarfélög og dreifða byggð. Með gösunarstöð gefst möguleiki á að breyta sorpi, þá sérstaklega viðarúrgangi, í afgas sem nota má í að knýja sprengjuhreyfil til að framleiða hita og rafmagn í heimabyggð. Slík notkun gösunarstöðvar getur komið í veg fyrir óþarfa landfyllingu og að úrgangi sé einvörðungu brennt. Rannsóknin felur í sér vistferilsgreiningu (e. life cycle assessment, LCA) með áherslu á kolefnisfótspor og önnur umhverfisáhrif afgasframleiðslu og orkuframleiðslu úr sorpi. Greiningin felur einnig í sér samanburð á umhverfisáhrifum vegna landfyllingar og brennslu sorps í Vestmannaeyjum. Grunneiningin, eða aðgerðareiningin (e. functional unit), sem rannsóknin skilgreindi er 1 kg af unnum viðarúrgangi, og aðferðir sem valdar voru til útreikninga umhverfisáhrifa voru CML 2001 og CED (Cumulative energy demand). Eftirfarandi þrjú sorpumhirðuferli voru greind í rannsókninni; 1) Gösunareining í eigu Verk- og Náttúruvísindastofnunar Háskóla Íslands, sem framleitt getur bæði rafmagn og heitt vatn, 2) sorpbrennslustöð sem verið er að reisa í Vestmannaeyjum og 3) metangasframleiðsla úr landfyllingu í Álfsnesi. Gösunarferlið reyndist umhverfisvænasti kosturinn varðandi gróðurhúsaáhrif ásamt 8 öðrum umhverfisárhifaflokkum (e. environmental impact categories). Lokaskref gösunarferlisins og rafmagnsframleiðsla í gösunarstöðinni, ásamt dísel- og olíuframleiðslu vegna notkunar á eldsneyti á vistferli stöðvarinnar höfðu mestu umhverfisáhrifin. Hvað varðar orkuframleiðslugetu mismunandi sorphirðuferla, gat sorpbrennsluferlið framleitt 11.093 GJ árlega samanborið við gösunarferlið (3.902 GJ) og landfyllinguna (241 GJ). Næmnigreining á inntaksbreytum vistferilsgreiningarinnar sýnir fram á hvaða þættir hafa mest áhrif á mismunandi umhverfisáhrifaflokka. Framtíðarrannsóknir gætu tekið til greina fleiri sviðsmyndir á mismunandi efnamassa sorps til að ná fram meiri sjálfbærni í umhverfismálum ásamt hreinni orkuframleiðslu.

Dedication

To Almighty God for His sustenance, love, and protection

To Olivia, Ohemaa, Eno, Forson and all family members for your eternal love.

Preface

This thesis is submitted in candidature for an MSc. degree from the University of Iceland. The work has been conducted between May 2020 and May 2021 at the Faculty of Civil and Environmental Engineering, supervised by Dr. Jukka Taneli Heinonen, Dr. Ólafur Ögmundarson, Dr. Páll Jens Reynisson, M.Sc. Marta Rós Karlsdóttir, and M.Sc. Nargessadat Emami.

The work is conducted in collaboration with the Faculty of Industrial Engineering and supervised jointly to assure necessary knowledge transfer.

Elvis Kojo Kutin-Mensah

May 2021

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Abbreviations

ADP	Abiotic depletion potential
AP	Acidification potential
BCCM	Bern carbon cycle model
CED	Cumulative energy demand
CHP	Combined heat and power
CML	Centrum voor Milieukunde Leiden (*trans*. Centre of Environmental Science)
EP	Eutrophication potential
ETP	Ecotoxicity potential
GHG	Greenhouse gas
GWP	Global warming potential
HHV	Higher heating value
HTP	Human toxicity potential
IPCC	Intergovernmental panel on climate change
ISO	International standard organization
LCA	Life cycle assessment
LCIA	Life cycle impact assessment
LCI	Life cycle inventory
LHV	Lower heating value
MSW	Municipal solid waste
ODP	Ozone depletion potential
SHI	Stepped hearth incinerator
SNG	Syngas, or synthetic gas, or synthetic natural gas

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Finally, I would like to thank the municipal authorities of Vestmannaeyjar for granting me access to project information and data on the proposed incineration facility to be constructed in the region.

1 Introduction

1.1 Background

The transition from continued reliance on fossil-driven energy systems to renewable energy is crucial to reducing energy-related carbon emissions and mitigating global warming (Omoregbe et al., 2020; Sasaki et al., 2009). Fossil fuel dominates the global primary energy supply, accounting for about 85%, and is responsible for about 89% of carbon dioxide (CO_2) emissions in 2018 (Lun et al., 2018; Saidur et al., 2011).

The need to mitigate for climate change, promote local energy security and economic development of rural areas as well as optimize waste resource utilization for bioenergy production has increased the acceptability of biomass (Ptasinski et al., 2009; McKendry, 2002; Staples et al., 2017). According to World Energy Council, biomass provides 10% of global energy use but only 13% is used for combined heat and power (CHP) (World Energy Council, 2016; Saidur et al., 2011). Biomass is rated the fourth global energy source, accounting for about 14% of the final energy consumption. Biomass has been one of the earliest sources of available and inexpensive fuel source for rural communities (Demirbaş, 2005). The renewability of biomass underscores the theoretical foundation of the carbon-neutrality assumption of biomass-derived CO_2 emissions, usually assumed to be compensated by plants through the photosynthesis effect (Darda et al., 2019; Eggleston et al., 2006). However, this convention adopted to estimate the life cycle CO_2 emissions has been critiqued by past and recent literature (For examples, see: Börjesson & Gustavsson, 2000; Searchinger et al., 2009; Searchinger, 2010). The criticisms stem from the underestimation of the climate change impact of bioenergy systems by this accounting convention. Often overlooked in the carbon-neutrality discussion, the CO_2 emissions from permanent and temporary carbon loses add to climate change before they are captured during biomass regrowth. A further argument put forth is that CO_2 emissions from bioenergy may not directly be offset by biomass regrowth, particularly for bioenergy from slower growing biomass, like forest (Cherubini et al., 2011; Heinonen et al., 2015).

Globally, the use of biomass fuel as commercial fuel has not yet achieved substantial commercial viability on a large scale due to undeveloped supply chains and limited supply of biomass feedstock (Saidur et al., 2011). Biomass wastes, however, represent a sustainable option to reduce the reliance on fossil fuel and its associated environmental burdens. The search for sustainable and economical fuel sources has made the use of waste materials useful sources of energy that has simplified the waste disposal process in addition to obtaining inexpensive heat and electricity. Locally sourced waste has become the preferred sustainable biomass application option—than other biomass fuel such as energy crops—towards the avoidance of waste management via landfilling. Moreover, locally sourced wastes, as compared to energy crops, avoid the transportation of fuel from a long distance; local wastes are usually close to the plants and the end-use (Saidur et al., 2011; Capehart et al., 2020).

Several process and technology pathways are available to convert waste to energy. Examples include direct combustion, pyrolysis, fermentation, composting, gasification, anaerobic digestion, etc. Amongst them, the gasification pathway provides the opportunity to convert organic wastes into useful energy such as heat, power, or both through a thermochemical combustion process, thereby diverting the volume of wastes that ought to be landfilled or incinerated. Organic waste streams such as wood, paper, textiles, plastics, food wastes, and yard wastes can be gasified (Arafat et al., 2015). It has been proven that downdraft gasification technology is effective on wood feedstock (Rollinson & Williams, 2016).

Gasification is not a novel technology. The use of CHP downdraft gasifiers has been in existence over the last century to generate energy from locally sourced biowastes (Mamphweli & Meyer, 2009). A renewed interest in utilizing the small-scale gasifier is attributed to the growing realization of this technology type as a sustainable option for effectively managing waste, reducing greenhouse gas (GHG) emissions, supplying uninterrupted energy to off-grid communities due to variations in seasons and weather, and guaranteeing future energy security (Asadullah, 2014). Globally, biomass gasification has seen an upward surge, with an annual growth of 8% in syngas production (U.S. Department of Energy, 2005). Syngas, or synthetic gas, or synthetic natural gas (SNG) is a mixed of fuel gas primarily consisting of hydrogen (H_2), carbon monoxide (CO), CO_2 and methane (CH_4) (Safarian et al., 2019; Demirbaş, 2005; Steubing et al., 2011). The wide acceptability of gasifiers is that it provides on-demand energy and avoid the reliance on storage technology such as batteries. Moreover, gasifiers offer enhanced efficiency paralleled to traditional combustion or steam turbine engines as well as offering a lower nitrogen oxides (NO_X) and particulate emissions due to the suitability of fuel feedstock specification to engine designs (Food and Agriculture & Organization, 1986; Hamilton et al., 2014).

1.2 Research problem and research questions

Globally, the industrial revolution, coupled with population growth, and enhanced socio-economic progress have intensified material consumption and the ensuing volume of waste generation (Schaffartzik et al., 2014). Waste generation and management are major issues confronting urban, small and isolated communities in Iceland. The volume of wastes generated in the country are facilitated by the healthier living standards of the citizens, continuous growth of the economy, and the influx of tourists for holiday making. SORPA, the company responsible for waste management in the Capital area, estimated a total of 76 ± 6 thousand tons of municipal solid waste (MSW) in 2010. This is equated to 222-257 kg of MSW per capita (Sundberg et al., 2010).The organic waste components account for about 60%, of which 43%, 41% and 16%, respectively, constitute mixed paper, timber and wood, and garden wastes. It is anticipated that the volume of MSW will amount to about 100 ± 7 thousand tons by 2030 (Safarian et al., 2020; Sundberg et al., 2010).

Landfilling and incineration have been the main conventional methods for disposing of MSW in Iceland. In the past, EEU regulations regarding refuse incinerators (89/369/EBE and 89/429/EBE) have been implemented into Icelandic laws, with aim of reducing exhaust pollution generated from the opening burning of wastes (Icelandic Ministry for the Environment and Natural Resources, 1999; Icelandic National Audit Office, 2011). Unfortunately, many waste incinerations plants in several locations, including Vestmannaeyjar, did not follow the 89/369/EBE and 89/429/EBE regulations resulting in dioxin pollution within locality of the incineration plants (Icelandic National Audit Office, 2011). For instance, the dioxin emissions from some incinerator sites in the enclave of farmland near Isafjordur have released carcinogenic agents and toxic chemicals into the meat and milk products of the Efri Engidalur farm (SMH, 2015; Halldorsson et al., 2012; Sundberg et al., 2010), consequently, jeopardizing the export of dairy products from Iceland (Icelandic Farm News Paper, 2011). Unfortunately, people exposed to these contaminated products are predisposed to developing specific types of cancer (U. S. Environmental Protection Agency, 2020a; National Human Genome Institute, n.d.). Also, while other incinerators were been found to exceed their emission limits (Umhverfistofnun, 2011), most landfill sites have been operated under unsanitary open dumping conditions (Halldorsson et al., 2012; Sundberg et al., 2010). Moreover, cargo trailers have added to the environmental pollution because of long distance transportation of MSW from local waste collection stations to the to waste incinerator plants (Icelandic Ministry for the Environment and Natural Resources, 1993; Icelandic National Audit Office, 2011). Following these setbacks, several incinerators, however, have been closed down (Halldorsson et al., 2012; Safarian & Unnthorsson, 2018).

In the Vestmannaeyjar region, about 1443 tons of general and painted timber wastes are shipped offshore to the mainland Landeyjarhöfn and subsequently transported by road to be disposed of in the landfill site at Álfsnes. The plastic (116 tons) and cardboard paper (258 tons) waste components are usually exported to Sweden and the Netherlands, respectively (Statistics Iceland, 2018). The waste management and transportation systems have heightened the carbon footprint of the region. Following a public outcry on the environmental and health impacts of improper waste management nationwide (Sundberg et al., 2010), the central government issued a new directive, effective from January 2020, to ban all landfilling activities in Iceland.

In response to the directive, the local government in the region has unveiled a proposal to construct a 2 MWh waste-to-energy incineration system predicated on improving waste management as well as recovering heat energy for district heating. While the proposed stepped hearth incinerator (SHI) is integrated with emission control systems, the overall environmental impacts of incinerators have been found as higher in comparison with other waste-to-energy processes and technologies (Matthews Environmental Solutions Ltd, 2018).

Waste gasification, instead of incineration, thus, represents a viable waste management alternative. Gasification has been found to emit fewer air pollutants than incinerators (Demetrious et al., 2018). Given this, installing a small-scale downdraft CHP pallet gasifier will be a preferred substitute for effective waste management and efficient resource utilization than the current solutions available in the Vestmannaeyjar region. The downdraft gasifier will reduce fossil fuel consumption for the transportation of wastes and potentially decreases the carbon footprint of waste management in the region. The downdraft gasifier can harness environmental-friendly synthetic gas (SNG) which can be combusted in engine generators to produce electricity and heat (CHP) (Safarian et al., 2019; Safarian et al., 2020).

Despite the numerous environmental benefits linked to waste gasification, it remains unclear, however, the GHG outcome of the downdraft CHP pallet gasifier in comparison to the proposed SHI system and the conventional landfill system. The main goal of this study is to evaluate the life cycle GHG emissions and selected other environmental impacts of SNG production and use in comparison with the reference systems in the Vestmannaeyjar region, using process life cycle assessment (LCA) methodology. Specific research questions are stated as follows:

a) Is the alternative CHP pallet gasifier system environmentally superior to the proposed incineration system and the conventional landfill system?

b) Which lifecycle phases and processes of the assessed systems are environmental hotspots?

c) What is the energy recovery/generation potential of the CHP pallet gasifier system in comparison to the reference systems?

1.3 Study scope

Scope of the study refers to the domain and operating parameters of research (Simon & Goes, 2013). The study scope includes geographic, system, and methodological scope. Geographically, the study focuses on Vestmannaeyjar (also Westman Islands) region. The region consists of several islets located 14 km south of Iceland with a population of 4330 residents. System scope covers the small-scale downdraft CHP pallet gasifier owned by the University of Iceland, the proposed incineration facility to be constructed in Vestmannaeyjar, and the sanitary managed landfill with a methane recovery system in Álfsnes. The system boundary of the LCA methodology is a gate-to-gate life cycle. Life cycle stages occurring before the point of waste generation are excluded from the study. In addition, resource extraction to fuel use, material consumptions and environmental impacts related to construction and decommissioning of plants, as well as specific processes (e.g. methane processing, leachate treatment and disposal of ash from the gasifier and incinerator) are not included in the scope. Validity and reliability of inventory data for the study are discussed in the discussion section.

2 Theoretical background

This section reviews the theoretical knowledge underlying the three main waste-to-energy process and technology pathways, the physical composition of woody biomass feedstock, energy recovery, and related environmental impacts, as well as previous empirical LCA studies. The section demonstrates knowledge of past research and the need for further studies on the environmental impacts in the field of small-scale gasifier.

2.1 Waste-to-energy processes and technologies

Several processes are available today to convert biomass to different energy sources. The choice of a particular process is dependent on the available supply of a type of biomass feedstock, the preferred energy carrier (end-use), environmental guidelines, economic considerations, etc. (Saidur et al., 2011). The method of retrieving energy from biomass can broadly be categorized under thermochemical conversion (e.g., combustion, gasification, and pyrolysis), biochemical conversion (e.g., fermentation, anaerobic digestion, and esterification), chemical conversion where energy from biomass is retrieved through various chemical reactions and thermal conversion in the form of bio-diesel. Usually, biochemical conversion is the most preferred route if the biomass has a very high moisture content (Saidur et al., 2011) for easy bacteria decomposition process to occur (Davis & Cornwell, 2008; Demirbaş, 2005; Tchobanoglous et al., 2004). In this thesis, the processes of gasification, incineration and landfill are compared as relevant means of waste disposal for the Vestmannaeyjar municipality. The following subsections shortly review the literature explaining the mechanisms of these processes.

2.1.1 Gasification

Gasification is a thermochemical process of transforming biomass feedstock under high temperature in the presence of controlled oxygen flow into a mixture of combustible gaseous fuels, through a series of chemical reactions (Arafat et al., 2015; Saidur et al., 2011; Van Huynh & Kong, 2013). The gasification process can be divided into four separate stages or zones, namely, drying zone, pyrolysis zone, combustion zone, and reduction (gasification) zone. Different chemical reactions take place in each zone (Safarian et al., 2019a; Casado & Pascual, 2008). Figures 2.1 and 2.2 illustrate the different reactions that occur in each zone and the process flow diagram of a downdraft CHP pallet gasifier, respectively.

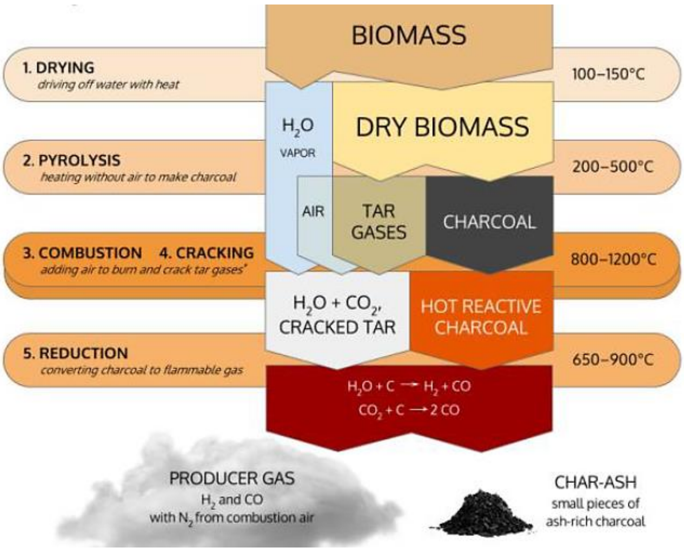


Figure 2.1 Gasification process steps and chemical reactions
Source. All Power Labs (2021)

The gasification process begins with a drying of biomass feedstock to increase the efficiency of the drier biomass. Drying usually takes place under a temperature of 100-200°C to reduce the moisture content in biomass wastes from 5-35% to 5%. This is followed by pyrolysis, where the dried wood pellets are heated in the first section of the reactor under a temperature ranging from 200-700°C in an oxygen-controlled environment to produce either solid products (such as char and charcoal), liquid products (such as pyrolignious acid) and gas gaseous products (such as hydrogen H_2, carbon monoxide CO, carbon dioxide CO_2 and methane CH_4) (Safarian et al., 2019; Demirbaş, 2005; Steubing et al., 2011). The products of biomass pyrolysis are determined by the temperature, heating rate, particle size and catalyst used. The pyrolysis process occurring more rapidly yields more gases than solids. The reaction of the pyrolysis process is Biomass-charcoal + volatile matter (Demirbaş, 2005).

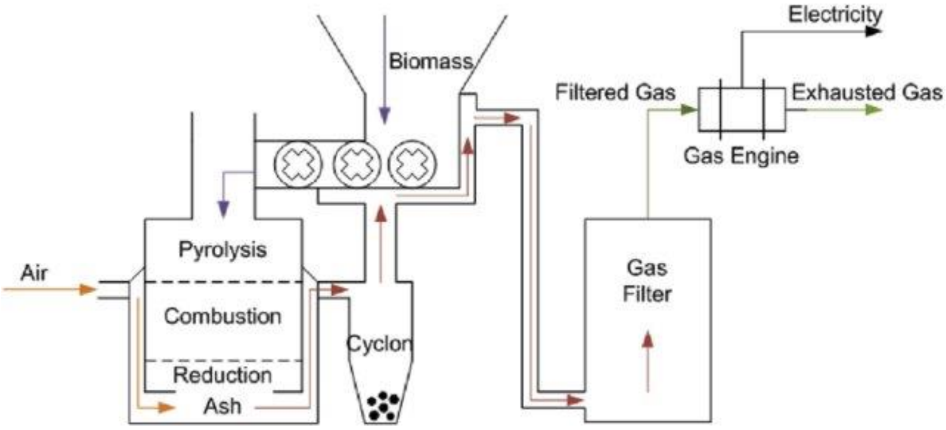


Figure 2.2 A process flow diagram of the CHP pallet gasifier
Source: García-Velásquez et al. (2018)

The unconverted solid biomass (char) and the bed material (olivine) are conveyed to the second section of the reactor where they are fully combusted and gasified at temperatures between 700-1500 °C and 800-1100 °C, respectively, in an interaction process under a presence of air. This reaction occurs concurrently upon the entry of the air into the gasifier. The air combines with the pyrolysis gases produced and at the same time flows downward paralleled with the solids (char and ash) through the oxidation and gasification zones. Hot steam and olivine provide the needed energy for gasification (Felder & Dones, 2007; Safarian et al., 2019; Steubing et al., 2011). The main reactions that transpire in the combustion zone are broken down as follows:

$$C + O_2 \rightarrow CO_2; -393 \text{ MJ/kmol}$$
$$2H2 + O_2 \rightarrow 2H_2O; -242 \text{ MJ/kmol}$$

A subsequent reduction process produces combustible gases—often known as SNG —which has a higher heating value (HHV) ranging from 3.5 – 8.9 MJ/m3 (Rajvanshi, 1986). The composition of SNG is CO_2 (15-20%), CO (20-30%), H_2 (30-40%), CH_4 (10-15%), ethylene (1%), and water (6%). The produced SNG is then cleaned to remove contaminants such as tars, alkalis, ammonia, chlorides, sulfides, and particulates. Alongside the main products produced, tar is also formed during the pyrolysis process. Tars partially break down as they move through the combustion and reduction zones. The tar consists of 10% of the total carbon from the waste, lowering the CO2 emission accordingly and recycled into fertilizers. Downdraft gasifier is generally the most preferred technology choice for reducing a large number of tars in biomass residues. (Arafat et al., 2015; Saidur et al., 2011; Van Huynh & Kong, 2013). The gas yield (efficiency) is contingent on the gasification process temperature and moisture level of the biomass (Felder & Dones, 2007). The reactions are as follows:

$$C + CO_2 \rightarrow 2CO_2; +164.9 \text{ MJ/kmol}$$
$$C + H_2O \rightarrow CO + H_2; +122.6 \text{ MJ/kmol}$$
$$CO2 + H_2 \rightarrow CO + H_2O; +42.3 \text{ MJ/kmol}$$

The most significant factors that determine the level of tar in a downdraft gasifier are residence time, oxidizing agents (steam vs. oxygen or air) (Monteiro Nunes et al., 2007). The relationship between tar content and temperature is inverse; an increase in temperature decreases the tar content in the SNG due to thermal cracking (Han & Kim, 2008). For example, an increase in temperature from about 700 to 820°C reduces the tar content significantly from 15 to 0.54 g/Nm3 in a circulated fluidized bed (Li et al., 2004). Figure 2.3 depicts the mass flow distribution throughput biomass gasification.

The SNG derived is then transferred to the internal combustion chamber which consists of a gas engine/turbine, providing a conducive environment for reaction with air to occur (Safarian & Bararzadeh, 2012; Safarian et al., 2019). The SNG can be combusted directly in the engine generator to generate heat and electricity or be upgraded into various fuels such as hydrogen, ethanol, gasoline, and diesel.

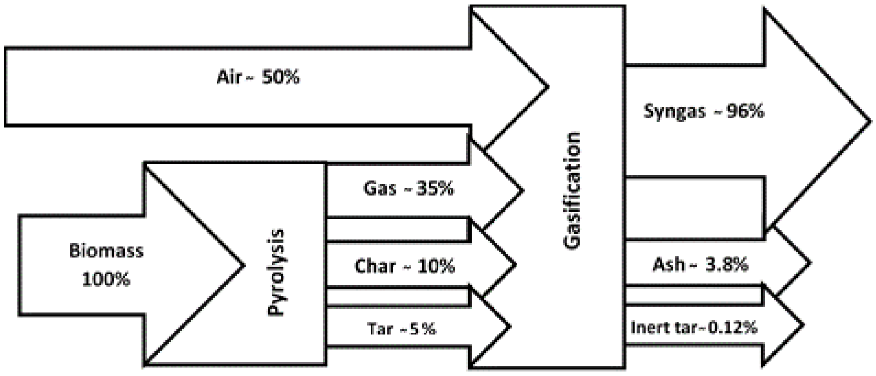


Figure 2.3 A Sankey chart of mass flow distribution throughput biomass gasification. Source. Safarian et al. (2019b)

Globally, Fischer–Tropsch synthesis, electricity generation, hydrogen, ammonia, and methanol constitute the five key applications of commercialized gasification plants exceeding 100 MWe capacity, of which Fischer–Tropsch synthesis and electricity generation from 29% and 24% of the total capacity, respectively (Baukal, 2001; Van Huynh & Kong, 2013). The main products of the gasification process and environmental impacts are summarized in Table 2. 1.

Table 2.1 Main gasification products and associated environmental impacts.

Gas species	Environmental impacts
CH_4	Strong GHG emissions as well as combustible fuel
CO_2	GHG emissions
CO	Toxic gas that causes asphyxiation. Also, a combustible gas
N_2	Eutrophication
HCN	Poisonous gas and explosive at high concentrations
H_2	Explosive gas and combustible fuel. Can cause asphyxiation

Source. Higman & van der Burgt (2008)

2.1.2 Incineration

Incineration is a direct combustion technology wherein the feedstock is converted directly into energy without any further chemical processes (Arafat et al., 2015). The main compounds that are usually released during the incineration process are carbon dioxide and water vapor (Demirbaş, 2005).

The incineration process begins with the shredding of solid waste. The shredded wastes are delivered to the waste reception hopper and then transferred unto the waste conveyor down to the ram loading box system. The waste is cascaded down a series of hydraulically actuated steps (stepped hearth design) to the primary combustion chamber (PCC) where they are combusted. The ash residue is periodically discharged through the sealed ash doors into a water bath (Matthews Environmental Solutions Ltd, 2018).

Products of combustion flow into the floor-mounted secondary combustion chamber (SCC) where the gases are further heated, oxidized, and subject to turbulence to ensure destruction of the combustion emissions. The flue gases in the SCC are dispatched to two medium pressure hot water heat exchangers (MPHWHE) where the gases are cooled under a temperature between 180 to 200°C after treated with air atomized urea solution to reduce the levels of NOx. The gas cooling produces a 135°C heated water/glycol mix, which is pumped through series of pipes for further cooling before it is being pumped back to the heat exchangers at 105°C (Matthews Environmental Solutions Ltd, 2018).

The treated gases exiting the heat exchangers are then injected with a reagent (a blend of sodium bicarbonate and activated carbon) to neutralize the acid content of the gases as well as to adsorb heavy metals and any reformed dioxins/furans emissions before transferred to the abatement plant, where a bag filter captures both the combustion particulates and the injected reagent. These substances are discharged automatically by a blast of compressed air through a series of solenoid valves. The resultant clean gases are monitored by a continuous emission monitoring system and are finally discharged to the atmosphere via the main induced draught fan and exhaust chimney (Matthews Environmental Solutions Ltd, 2018). Figure 2.4 shows the process flow diagram of the stepped hearth incineration system to be installed in the Vestmannaeyjar region.

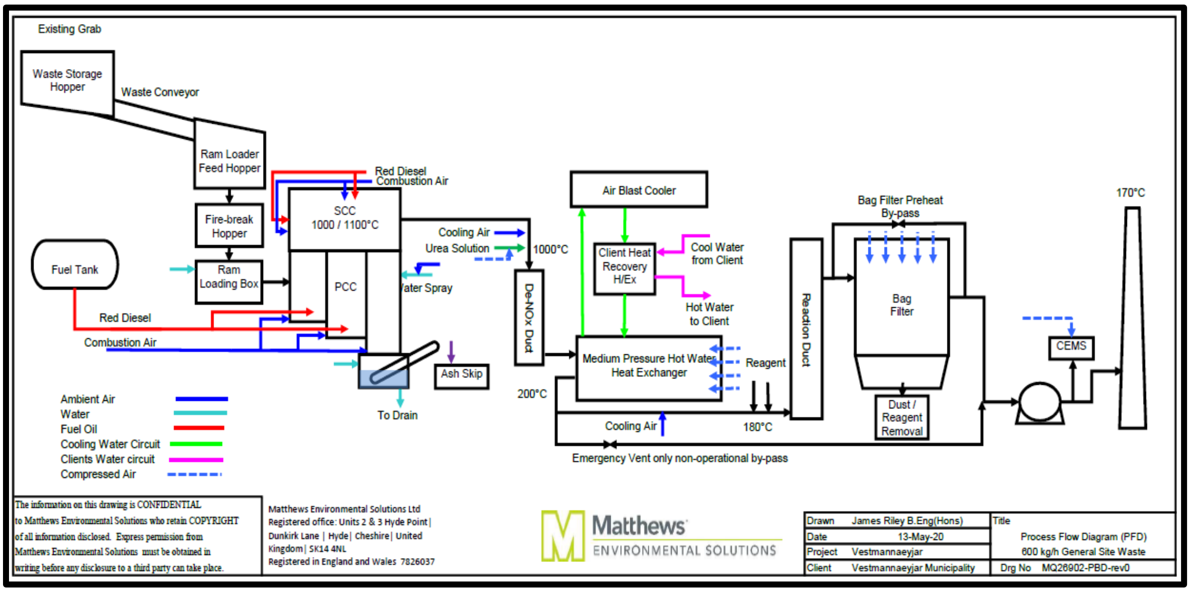


Figure 2.4 Process flow diagram of the stepped hearth incineration system
Source. Matthews Environmental Solutions Ltd (2018)

2.1.3 Landfill

Anaerobic digestion via landfill is a biological process where micro-organisms break down and convert biodegradable material (organic wastes) with water in the absence of oxygen into a mixture of gases, known as biogas. The biogas largely contains methane CH_4 and carbon dioxide CO_2, and a trace amount of hydrogen sulfide (H_2S). Table 3.2 depicts the standard composition of biogas. This bacteria decomposition process has been commercially proven worldwide and is commonly considered for treating wet biomass wastes. The advantage of anaerobic digestion over aerobic digestion is that this process generates less solid sludge. Some landfills are fitted with a bioreactor to speed up the biodegradation and biogas production process (Davis & Cornwell, 2008; Demirbaş, 2005; Tchobanoglous et al., 2004). The formation of the gas starts from the decomposition process of the waste under the anaerobic condition that generates the landfill gas, which mainly comprises methane and carbon dioxide (Lee et al., 2017).

The anaerobic digestion process undergoes four main chemical and biological stages: namely, hydrolysis, acidogenesis, acetogenesis, and methanogenesis. Hydrolysis as the first stage of anaerobic digestion breaks down large organic polymers for example carbohydrates, fats, and proteins into smaller molecules, such as amino acids, fatty acids, and simple sugars. The remaining components are further broken down into chemicals, such as alcohols and carbonic acids through the acidogenesis process. The acetogenesis process then produces H_2, CO_2, and largely acetic acid ($C_2H_2O_2$). Methanogenesis is the final stage, chemicals produced by prior processes are converted by methanogens into CO_2 and CH_4. The four key stages of anaerobic digestion are illustrated in Figure 2.5. The overall chemical reaction is represented by the simplified chemical equation below:

$$C_6H_{12}O_6 \rightarrow 3CO_2 + 3CH_4$$

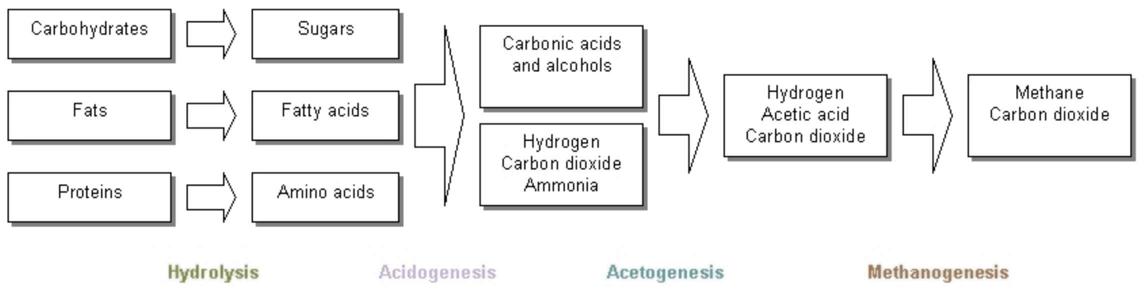


Figure 2.5 The four key stages of anaerobic digestion.
Source. Wordpress.com (2010).

2.2 Compositions of biomass feedstock

Biomass is a generic term given to organic matter obtained from plants and animal resources such as forestry products (e.g. wood), agricultural products (e.g. crops, seaweed, etc.), and wastes (e.g. industrial, plants, human and animal) (Oregon, 2010). Crops and wastes constitute the two main sources of biomass. Crops include agriculture crops and woody crops, whereas wastes include crop wastes, wood residues, municipal solid wastes, sewage, and animal wastes. Biomass has been one of the earliest sources of available and inexpensive source fuel for rural communities (Boyle, 2004; Demirbas, 2004).

Combustion of biomass requires an understanding of the chemical compositions and properties of the fuel feedstock which affects the result of the combustion process. Biomass combustion involves series of chemical processes in which carbon and hydrogen are oxidized to carbon dioxide and water respectively (Saidur et al., 2011). The compositions of biomass are the distinctive structures that determine and characterize the physical properties, quality, applications, and environmental challenges associated with biomass fuel (Vassilev et al., 2010). The organic and inorganic, physical properties and energy content draws out the differences between biomass fuel and fossil fuels. Regarding chemical composition, biomass has less carbon and heating value but more oxygen, hydrogen, volatile and ash content, moisture, and other compounds (Hawaii Natural Energy Institute, 2010). Studies particularly use proximate, ultimate, ash analysis, and higher HHV analyses to understand and characterize different wood products (Arafat et al., 2015; Vassilev et al., 2010).

2.2.1 Proximate compositions

Proximate compositions of woody biomass consist of the percentage of volatile matter, fixed carbon, and ash contents. These compositions determine the combustion potential of woody biomass. High proportions of fixed carbon ad volatile matter augment the heating value of the wood fuels. In contrast, high ash content triggers combustion difficulties. The proximate analysis of different wood products is presented in Table 2.2.

Table 2.2 Proximate analysis of different wood products

	Timber and wood waste	**Wood pellets**	**Wood chip**
Moisture	5.01	0.00	9.20
Volatile matter (VM)	93.06	81.82	77.90
Fixed carbon (FC)	6.38	17.65	20.30
Ash	0.56	0.53	1.80

Sources. Freeman et al. (n.d); Naryanto et al. (2019); Naryanto et al. (2019); Gautam et al. (2011).

2.2.2 Ultimate compositions

The ultimate analysis provides the basis for assessing the proportion of nitrogen (N), sulfur (S) ad chlorine (Cl) in biomass to understand its environmental impacts. Also, it facilitates the computations of carbon, hydrogen, and oxygen fractions to determine the heating value of wood fuels (Vassilev et al., 2010). The ultimate analysis of different wood products is presented in Table 2.3.

Table 2.3 Ultimate analysis of different wood products

	Timber and wood waste	**Wood pellet**	**Wood chips**
C	56.8	49.75	44.00
H	7.28	6.40	5.50
N	0.82	0.09	1.00
Cl	0.82	0.0	0.0
S	0.0	0.09	0.00
O	34.29	43.14	47.70
HHV (MJ/Kg)	18,68	19,26	17,26
LHV (MJ/Kg)	17.75	18.30	16.40

wt. % of *dry basis*

Source. Freeman et al. (n.d); Naryanto et al. (2019); Naryanto et al. (2019); Gautam et al. (2011).

2.2.3 Ash composition

Ash analysis evaluates the amount of potassium, sodium, chlorine, and phosphorous contents in the biomass fuel which are determined by the state of the biomass plants, significantly impacted by nutrients, fertilizers, and weather conditions (Saidur et al., 2011).

2.2.4 Caloric value

The heating value, also known as, caloric value, of biomass is explained by the HHV and lower heating value (LHV). HHV measures the energy content of dry biomass. LHV is determined by subtracting the energy required to vaporize the moisture content from the biomass from the HHV. The presence of oxygen reduces the heating value, while C and H increase it. The caloric value is activated when the biomass feedstock is heated or combusted. One of the well-known challenges associated with biomass fuel is the low conversion efficiency (Boyle, 2004; Khan et al., 2009).

2.3 Energy recovery and environmental impacts of biomass feedstocks

Energy from biomass called bioenergy is derived from the sun in which atmospheric carbon dioxide is converted into other carbon molecules in plants through the process of photosynthesis. Energy recovery from biomass and the ensuing environmental pollution is dependent on the chemical composition of the biomass that influences the properties, quality, and potential applications of a particular biomass fuel type (Saidur et al., 2011). The versatility characteristics of biomass make it a unique and useful energy source from other renewables such as solar and wind. It can be transformed into other fuel types such as solid (e.g. charcoal), liquid (ethanol), and gaseous (e.g. methane) fuels and energy (e.g. electricity and heat) when combusted, making it quite easier to store energy (Kopetz, 2007; Saidur et al., 2011).

In comparison to fossil fuels, biomass is environmentally superior—biomass combustion adds no net addition of CO_2, on the assumption of fast-growing biomass species (e.g. annual crops and lignocellulosic energy crops) sequestrating biomass-related CO_2, emissions in the atmosphere (Cherubini et al., 2011). This unique characteristic supports the theoretical assumption that biomass is a carbon-neutral source of energy. The carbon content in biomass reacts with atmospheric oxygen (O_2) to produce CO_2 which is discharged into the atmosphere. When the woody biomass is burned, the amount of CO_2 emitted is equivalent to the same amount absorbed from the atmosphere. This process is often referred to as carbon sink, carbon cycle, or zero carbon emissions, as depicted in Figure 2.6. Granting the carbon neutrality of biomass to global warming, the fossil fuel-derived GHG emissions related to biomass utilization cannot be disregarded, such as fuel consumption in waste transportation and electricity consumption in gasifier operations. Moreover, the combustion of biomass produces a higher particulate matter when compared to other liquids and gaseous fuels. This gives reasons for environmental concerns regarding the cost-effective measures for reducing, for example, aerosol emissions from biomass combusting plants (De Best et al., 2008; Saidur et al., 2011; Sami et al., 2001). Therefore, harnessing the full potential of biomass-derived SNG as a substitute fuel source for heat and electricity generation requires that combustion occurs under conditions that would produce low nitrogen oxides (NOx) emissions (Van Huynh & Kong, 2013) and availability of special cleaning procedure to remove deposits of particulate matter (Sami et al., 2001). NOx comprises nitrogen oxide species (i.e. NO and NO_2) which have detrimental impacts on both human health and the environment (Baukal, 2001; Van Huynh & Kong, 2013).

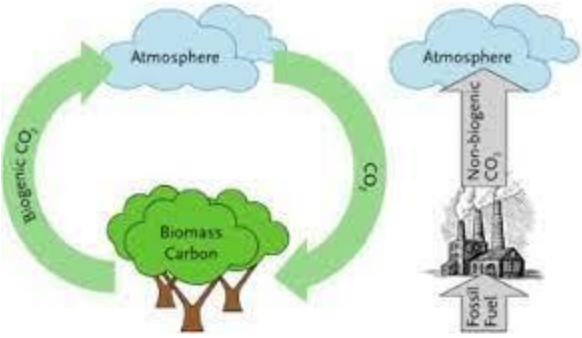


Figure 2.6 Carbon cycle of biogenic CO2 emissions vs. fossil emissions
Source. IEA Bioenergy (2021).

Nonetheless, the biomass gasification process is a relatively matured technology that can be utilized to produce low-carbon ('green') power or fuel in comparison to other thermochemical and biochemical processes. Direct combustion (incineration) of biomass causes significant environmental pollution. MSW contains toxic chemicals including heavy metals and other organic compounds. Burning these chemicals unleashes these toxic emissions into the atmosphere and eventually settles into groundwater that is harmful to human health. Hence, it is appropriate to convert biomass into other forms of fuels such as liquid and gaseous fuels to reduce the environmental impacts (Saidur et al., 2011). However, if well-managed, toxic ashes from incinerators could be useful inputs in cement production, mines reclamation, fertilizer application, biochar (charcoal) production, and industrial tile and road base (Boyle, 2004). Yet, incineration—and other thermochemical processes—are believed to be environmentally superior to landfills because anaerobic decomposition of biowastes releases pollutants such as CH_4, NH_3, H_2S, volatile organic acids, and other chemicals that are far potent. Combusting reduces the environmental impacts of these pollutants as well as avoiding contamination of soil and water that the stored biomass will leach into them (Sami et al., 2001).

2.4 Review of previous LCA studies

LCA studies have been conducted to assess and compare the environmental performance and energy generation on various waste-to-energy processes and technologies, such as gasification, incineration and landfill. Other studies have also compared conventional fossil energy systems to alternative bioenergy systems. The subsections below discuss extant literature under these thematic areas.

2.4.1 Environmental sustainability of bioenergy vs. fossil energy systems

The environmental sustainability of bioenergy systems over fossil energy has been researched extensively in environmental science literature (De Best et al., 2008; Saidur et al., 2011; Sami et al., 2001). In the Adams & McManus (2014) study, the environmental performance of the small-scale CHP biomass gasification plant was compared to fossil fuel-based CHP systems. The results indicate that the overall environmental impacts of the gasifier were relatively lower when compared to the fossil fuel-based CHP systems. Similarly, Aberilla et al. (2019) compared the environmental sustainability of direct combustion (incineration), gasification, and anaerobic digestion to a conventional diesel generator. As compared to the diesel generator, the study reveal that both combustion and gasification were associated with lower impacts, apart from eutrophication, ecotoxicity, and human toxicity. Cambero et al. (2015) on the other hand, evaluated five bioenergy systems in two remote communities in British Columbia, Canada—comprising of openly burning forest residues, landfilling of sawmill residues, biomass gasification cogeneration, biomass boiler electricity generation system, and biomass boiler cogeneration—and compared with conventional oil boiler cogeneration system. The biomass boiler cogeneration system emerged as the most sustainable option. Van Huynh & Kong (2013) assessed and compared NOx emissions from woody SNG combustion to natural gas. Under similar heat conditions, the findings point out that woody SNG produced higher NOx emissions when compared to natural gas, and that the formation of NOx is vital in biomass-derived SNG combustion.

2.4.2 Environmental performance of different bioenergy systems

Studies that have evaluated environmental impacts of different bioenergy systems have produced heterogeneous results, although landfill and gasification have emerged as environmentally beneficial as compared to incineration. For instance, Aberilla et al. (2019) evaluated environmental sustainability of feedstocks from a mixture of agriculture wastes (i.e. rice residues, coconut residues, and livestock manure) in different waste treatment technologies in farming communities in Southeast Asia. The LCA results indicate that anaerobic digestion system has the highest environmental benefits for 14 out of 18 impact assessments. Jeswani & Azapagic (2016) compared the environmental sustainability of electricity generation and co-generation of heat and electricity from incineration and biogas recovery from landfill. The study suggests that co-generation of heat and electricity from landfill was associated with favorable environmental performance in eight of the eleven impact categories, although electricity generation from incineration had a lower impact in all categories, apart from human toxicity. Zaman (2012) found that the overall environmental benefits of sanitary landfill was larger than that of pyrolysis-gasification and incineration. Demetrious et al. (2018) estimated the impacts of climate change, acidification, eutrophication, and photochemical oxidation of residual waste recovery facilities in Sydney, Australia. Landfills were found to have lowest GHG and acidification emissions. Arafat et al. (2015) assessed the applications of six waste treatment processes of gasification, anaerobic digestion, bio-landfills, incineration, composting, and recycling on six individual waste streams, consisting of wood, food, paper, plastic, yard, and textile wastes. The study concluded that anaerobic digestion and gasification emerged as environmentally superior technologies whereas minimum environmental benefit was associated with composting.

The environmental benefits of gasification over incineration have been documented. Safarian et al. (2020) compared environmental assessment of integrated CHP waste gasification with waste incineration in Iceland. In all the impact indicators evaluated, the results suggest that electricity generation from waste gasification is more environmentally beneficially than incineration. In the Zaman (2010) study, gasification environmental burdens are lower in global warming, acidification, eutrophication, and eco-toxicity categories than incineration. Similarly, Zaman (2013) in a later study, reveal that pyrolysis-gasification had a minimal environmental load than incineration in acidification, eutrophication, and aquatic eco-toxicity. Cleary & Caspersen (2015) compared a baseline 211 MWe wood pellet-fired power plant to a reference small 250 KWe CHP wood chip gasification plant. The small-scale gasifier exhibited a superior overall environmental performance due to benefits arising from using a dried feedstock, reduction in biomass pretreatment processes, and shorter fuel shipping distance. In a contrasting study, Burnley et al. (2012) quantified the environmental performance of gasification, incineration, anaerobic digestion, combustion in a dedicated plant, and combustion in a cement kiln using organic (i.e. paper, food waste, and wood), non-recyclable, and refuse-derived fuel wastes. The findings indicate that direct combustion with energy recovery was associated with minimum human toxicity, acidification, aquatic ecotoxicity, and eutrophication impacts.

Studies have also assessed the environmental impacts of gasification from different biomass feedstocks. For instance, Puy et al. (2010) evaluated and compared gasification from post-consumer wood from recycling plants, post-consumer wood from bulky wastes, and forest residues in metropolitan areas in Spain. The study identified gasification of wood from recycling plants as the preferred environmental sustainable option, due to the elimination of some pre-treatment processes and less energy intensity. Van Huynh & Kong (2013) on the other hand, appraised combustion of SNG derived from different biomass feedstocks, such as pine, maple–oak mixture, and seed corn. Each scenario was tested under different oxygen and steam conditions using different heat rates. Amongst the three-biomass feedstock, seed corn yielded the maximum nitrogen content and ammonia concentration in SNG, resulting in the highest NOx emissions between 450–900 ppm as compared to the wood-derived SNG under all the test conditions. The results also suggest that the NOx emissions reduce when the heat rate decreases by the same margin. A study by Adams & McManus (2014) suggests that the overall GHG emissions of a small-scale CHP gasification plant that uses wood waste as a feedstock were very low although toxicities, particulates, and resource depletion emanated from ash disposal, SNG combustion, and fossil fuel, metal, and water consumptions, respectively were higher. The authors concluded that finding a consistent supply of feedstock makes biomass gasification a desirable technology.

2.4.3 Energy generation potential of different bioenergy systems

Energy generation potentials from different waste management facilities have also been assessed and compared. In the Demetrious et al. (2018) environmental life cycle assessment of residual waste recovery facilities in Sydney, Australia, incineration achieved the overall highest electricity generation potential. Gasification-pyrolysis, nonetheless, yielded higher energy as compared to landfill. Arafat et al. (2015) applied six MSW treatment processes on six MSW streams to estimate energy generation potentials. The analysis identified recycling suitable process for paper, wood, and plastics; anaerobic digestion for energy recovery from food and yard wastes; and incineration for energy recovery from textile waste. In Cleary & Caspersen (2015), the baseline 211 MWe wood pellet-fired power plant recovered higher energy than the reference small 250 KWe CHP wood chip gasification plant due to high electrical efficiency.

Qazi & Abushammala (2020) analyzed electricity generation and GHG emission reduction from the MSW sector in Oman. The study suggests that incineration is the most optimum technology for energy generation than gasification and anaerobic digestion. Dalmo et al. (2019) assessed electricity recovery from incineration, gasification and anaerobic digestion in São Paulo State, Brazil. Also, two-hybrid combinations of incineration/anaerobic digestion and gasification/anaerobic digestion were investigated. The incineration/anaerobic digestion combination appeared as the highest theoretical potential of energy production. Segurado et al. (2019) evaluated MSW potential for electrical and thermal energy recovery in Oujda city, Morocco. The analysis indicates that direct combustion is the optimal technology for the maximum potential energy generation, followed by gasification and anaerobic digestion. Burnley et al. (2012) point out that the maximum energy recovery potentials from different bioenergy systems were sensitive to the efficiencies of each technology.

The quest for sustainable waste management practices and resource utilization has instigated research interest in bioenergy systems. Empirical research on the most optimum waste treatment technology for improving environmental performance as well as optimizing energy generation potential has yielded mixed results, although landfill and gasification have come to light as the most sustainable options for waste management whereas incineration has gained a reputation for generating higher energy. Studies evaluating the environmental performance of different waste treatment technologies in the Icelandic context is limited. This research gap, however, provides the basis for assessing and comparing, for the first time, the life cycle GHG emissions and other environmental impacts of a small-scale CHP pallet gasifier with a SHI and methane-recovery landfill systems.

3 Descriptions and modelling of case study systems

3.1 CHP pallet gasification system

The CHP pallet gasifier is part of a gasification lab owned by the University of Iceland. The technology was designed and manufactured by All Power Labs San Francisco. The PP20 Power Pallet model is made up of a GEK Hot TOTTI multi-stage gasifier, spark fired industrial engine, generator head, and electronic controller. The system automatically regulates SNG/air mixture through a wideband Bosch oxygen sensor and removes ash via a mechanical auger. The Process Control Unit monitors and responds to all internal reactor, filter, and engine conditions, and displays the results on an LCD screen. With a biomass conversion efficiency of 35% upwards (i.e. 1.2 kg of feedstock to 1 kWh of electricity), the gasifier can produce 18 kWe of electricity and 20 kWt of heat for district heating. It has a power rating of 15 kW at 50 Hz, 18 kW at 60 Hz. It has a biomass gasification rate of 22kg per hour (i.e. 0.33 m3 of SNG every 3 hours) (All Power Labs, 2021). Figure 3.1 shows The PP20 Power Pallet gasifier model.

Figure 3.1 CHP Power Pallet gasifier (PP20 model)
Source: All Power Labs (2021)

This system is a type of downdraft technology that has two separate chambers: the reactor chamber converts biomass feedstock to SNG while the engine chamber combusts the syngas to generate electricity and heat (Steubing et al., 2011). The technology is classified as a fixed-bed gasifier because the gasifying agents and gas descend through the reactor (Arnavat et al., 2010). Air, steam, oxygen, or a mixture of them are used as agents for the gasification process (Safarian et al., 2020). The tar consists of 10% of the total carbon from the waste, lowering the CO2 emission accordingly, and can be recycled into fertilizers. The small-scale downdraft gasifiers have evolved as the most preferred choice for producing clean gas with minimal dust and tar content as compared to the fluidized bed gasifiers. It is economically competitive for small-scale uses (Safarian et al., 2020). The key drawbacks of this gasifier type are that it is difficult to scale up capacity as compared to fluidized bed gasifiers and vulnerable to transferring heat under consistent high temperatures. More so, this technology usually requires appropriate feedstock specifications, below 20% moisture contents of feedstock, and a moderate even particle size distribution to achieve the desired results. (Safarian et al., 2020).

The number of gasifiers required to combust the available wood pellets in Vestmannaeyjar was estimated as follows:

$$NG = \frac{WP}{GCC * OPH} \tag{3.1}$$

Where:

NG	number of gasifiers required for available wood pellets
WP	available wood pellets
GCC	combustion capacity of 1 gasifier per hour
OPH	available operating hours per gasifier per year

Carbon dioxide and other emissions of the gasification and combustion process are modelled based on the proximate and ultimate compositions of wood pellets in Section 2.2 (i.e. Tables 2.2 and 2.3), using the fuel analysis method (IPCC, 2006), as follows:

$$EF_{CO2} = M_p * C_p * C_t * \eta_{ICE} * (1 - \eta_{CE}) \tag{3.2}$$

$$EF_P = M_p * C_p * \eta_{ICE} * (1 - \eta_{CE}) \tag{3.3}$$

$$M_p = \frac{MW_p}{EW_p}$$

Where:

EF_{CO2}	emission factor for carbon dioxide
EF_p	emission factor for other pollutants
M_p	the ratio of molecular weight of pollutant to the weight of active element (i.e. CO2 to C, CH4 to H, etc.)
C_p	the concentration of an active element in pollutant
C_t	fraction of carbon content of tar
η_{ICE}	the overall efficiency of the CHP gasifier
$1-\eta_{CE}$	control efficiency for nitrogen oxides (NOx) emissions
MW_p	the molecular weight of pollutant
EW_p	elemental weight of pollutant

Electrical and thermal energy generated in the internal combustion engine of the CHP gasifier was calculated as:

$$\mathrm{KWe/yr} = \mathrm{kWe/hr} * \mathrm{OPH} * \mathrm{NG} * 3.6 \tag{3.5}$$

$$\mathrm{KWe/hr} = \frac{\mathrm{SNG}_{fr} * LHV_{SNG} * \eta_{ICE,e}}{3.6} \tag{3.6}$$

$$\mathrm{KWt/yr} = \mathrm{kWt/hr} * \mathrm{OPH} * \mathrm{NG} * 3.6 \tag{3.7}$$

$$\mathrm{KWt/hr} = \frac{\mathrm{SNG_{fr}} * \mathrm{LHV_{SNG}} * \eta_{\mathrm{ICE,t}}}{3.6} \tag{3.8}$$

$$\eta_{\mathrm{ICE,e}} = \frac{\mathrm{KWe_{SNG}}}{\mathrm{HV_{SNG}}} \tag{3.9}$$

$$\eta_{\mathrm{ICE,t}} = 1 - \eta_{\mathrm{ICE,e}} - \mathrm{SL} \tag{3.10}$$

$$\eta_{\mathrm{ICE,e}} = \frac{\mathrm{KWe_{SNG}}}{\mathrm{Fuel_{SNG}}} \tag{3.11}$$

$$\eta_{\mathrm{ICE}} = \frac{\mathrm{KWe_{SNG}} + \mathrm{KWt_{SNG}}}{\mathrm{Fuel_{SNG}}} \tag{3.12}$$

Where:

KWe/yr	the annual electrical energy generated
KWt/yr	annual thermal energy generated
KWe/hr	electrical output per hour
KWt/hr	thermal output per hour
OPH	available operating hours per gasifier per year
NG	number of gasifiers required for available wood pellets
SNG_{fr}	SNG flow rate per hour (m^3/hr)
LHV_{SNG}	lower heating value of SNG fuel (MJ/m^3)
$\eta_{ICE,e}$	the electrical conversion efficiency for the CHP gasifier
$\eta_{ICE,t}$	the thermal conversion efficiency for the CHP gasifier
3.6	the conversion factor from MJ to kWt
KWe_{SNG}	electricity generated per m^3 of SNG
KWt_{SNG}	thermal generated per m^3 of SNG
SL	system losses
η_{ICE}	the overall efficiency of the CHP gasifier

3.2 Stepped hearth incineration system

The stepped hearth incineration system is intended to be built in Vestmannaeyjar. The municipality has commissioned Matthews Environmental Solutions Ltd to build a new general site waste incinerator to burn wastes of the type and rates specified that conforms to the waste incineration directive (WID) 2000/76/EC and now an integral part of the Industrial Emissions Directive (IED) 2010/75/EU to burn wastes of all types and rates specified. This planned facility is yet to be constructed but the process description and technical specification, including plant layout, has already been submitted to the municipality for consideration.

Under normal steady operating conditions, the plant has a design burn rate of 600 Kg/h of general waste at 14 MJ/Kg. It has availability rates of 85.6% and 75% under maximum and normal operations, respectively. The waste types handled by the plant include municipal waste, plastic bags, waste paper, and painted timber. The plant when constructed will comprise several distinct systems, including waste handling/loading systems, combustion equipment, heat recovery/dissipation system, abatement plant, exhaust equipment, and continuous emission monitoring system (Matthews Environmental Solutions Ltd, 2018). A typical stepped hearth IED/WID incinerator is depicted in Figure 3.2.

Figure 3.2 Stepped hearth IED/WID incinerator
Source: Matthews Environmental Solutions Ltd (2021)

The typical wood chip proximate and ultimate compositions in Section 2.2 (i.e., Tables 2.2 and 2.3) was used for modelling GHG emissions from the incinerator using the fuel analysis method (IPCC, 2006), as follows:

$$EF_p = M_p * C_p * \eta_{ICE} * (1 - \eta_{CE}) \quad (3.13)$$

Default emission factors adjusted for control efficiencies were adopted for non-GHG emissions, estimated as:

$$EF_p = DEF_p * (1 - \eta_{CE}) \quad (3.14)$$

Where:

EF_p	emission factor for a pollutant
DEF_p	default emission factors
M_p	the ratio of molecular weight of pollutant to the weight of active element (i.e. CO2 to C, CH4 to H, etc.)
C_p	the concentration of an active element in pollutant
η_{ICE}	the electrical conversion efficiency for internal combustion engine
$1-\eta_{CE}$	control efficiency for a pollutant abatement

The SHI system releases heat energy when the organic portion in the wood chips is combusted directly. The released heat is usually captured in a controlled system (furnace) and is used directly for domestic water heating. Heat energy is recovered with a thermal efficiency of 85% (Sundqvist, 1999). The thermal energy (KWt/yr) that can be recovered from useful heat generated in the internal combustion engine is calculated as:

$$KWt/yr = \frac{\mathrm{LHV}_{WC} * \mathrm{WC}_{yr} * \eta_{ICE}}{3.6} \tag{3.15}$$

Where:

KWt/yr	annual thermal energy generated
LHV_{WC}	the lower heating value of the wood chip
WC_{yr}	annual mass of wood chips incinerated
η_{ICE}	the thermal conversion efficiency for internal combustion engine
3.6	the conversion factor from MJ to kWt

3.3 Methane recovery landfill system

This landfill system is a managed sanitary landfill with the recovery of biogas and a leachate collection system located at Álfsnes, about 30 km north of the capital city in the south-eastern part of Iceland. About 350 ha of land have been earmarked for landfill but the actual landfill area covers around 40 ha including fences, roads, and service areas. The operation of the landfill is governed by EU legislation under the Icelandic Environmental and Food agency permit, which prescribes the procedures for collection, utilization, and flaring of landfill gas (Halldorsson et al., 2005). The landfill area is marked with several lanes, 40m wide, with different lengths. In 2019, 131,000 tons of MSW were landfilled, which represents a reduction of 11.1% and 1.3% in 2018 and 2017, respectively (SORPA, 2020).

The topsoil of each lane is excavated until it reaches the bedrock. The excavated lanes are levelled with clay and compacted. Pipes and shredded tyres are attached to the bottom of the levelled ground as drainage support systems. Wastes organized in bales are dumped at the bottom layer of the landfill through a weighbridge remotely controlled from the Gufunes station until they reach about 26 m high. After compacting the wastes, about 1.0 to 1.5 thickness of peat soil and other materials are placed on top of the wastes to facilitate the settling down of the wastes within 4 to 5m during the first months. The top layer is then removed after the wastes had settled down for further disposal of more wastes, with the view to optimizing land space (Halldorsson et al., 2005).

The formation of the gas starts from the decomposition process of the waste under the anaerobic condition that generates the landfill gas, which mainly comprises methane and carbon dioxide (Lee et al., 2017). Leachate from landfill is no longer released into the sea but collected in the settling chamber. The landfill is fitted with gas collection pipes that function under a technology known as an active system. This technology uses a vacuum to suck and collect the landfill gas from the wells connected to an underground collection network of pipes in a common gas duct and subsequently transported to the methane processing plant. The chemical composition of landfill gas at Álfsnes comprises CH_4 (57%), CO_2 (41%), and other gases (2%). At the processing plant, the collected gas is cleaned and upgraded (i.e., separation of carbon dioxide from the landfill gas result in a high methane purity) to a high-quality gas, known as biomethane or renewable natural gas. The cleaning process depends on the use of water scrubbers. Biomethane is used to power Reykjavik city waste trucks and/or combusted in an electrical generator (gas turbine) to generate electricity (Halldorsson et al., 2005). In Iceland, only 40% of the landfill gas recovered is utilized for energy production. The remaining 60% is combusted to flare without energy recovery (Metan Hf, 2020; Hauksson, 2019). To reduce CO_2 emissions through flaring, SORPA in collaboration with CarbFix (a collaborative research project that pioneered a novel approach to capturing and storing CO_2 in water and its injection into subsurface basalts) have planned to dispose of an estimated 3,000 tonnes of CO_2 annually from SORPA's methane plant (Reykjavik Energy, 2020). The process flow diagram of the landfill methane gas recovery and processing system is shown in Figure 3.3.

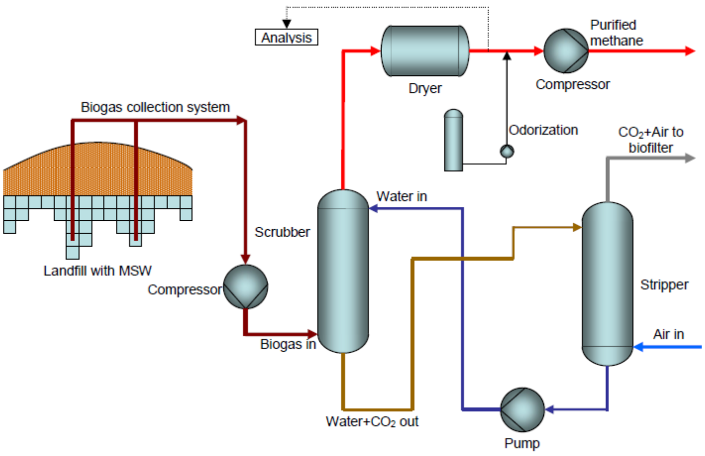


Figure 3.3 Process flow diagram of the landfill methane gas recovery and processing system Source. SORPA (2021).

The GHG emissions and electrical generation were modelled on the methane generation potential of the landfilled waste wood and methane concentration of the methane gas, as reported in the literature (Ayodele et al., 2017; Lee, et al., 2017). Methane gas generation potential, methane, and carbon dioxide emissions are estimated in equations (3.16) to (3.18), respectively, as follows:

$$MGP_{CH_4} = DOC * DOC_F * MCF * F * C_{CH_4} \quad (3.16)$$

$$DOC = DOC_{ww} * w\%, w$$

$$EF_{CH_4} = (MGP_{CH_4} - MGC_{CH_4}) * (1 - OX) \quad (3.17)$$

$$MGC_{CH_4} = MGP_{CH_4} * \eta_{CH_4}$$

$$EF_{CO_2} = MGC_{CH_4} \quad (3.18)$$

Where:

MGP_{CH_4}	methane gas generation potential
DOC	the degradable organic carbon content of landfilled wood waste
DOC_F	fraction of DOC decomposed for wood
MCF	methane correction factor
F	fraction of methane (CH4) concentration in landfill gas at Álfsnes
C_{CH_4}	the carbon content of methane (conversion from C to CH4) [i.e. 16/12]
DOC_{ww}	the degradable organic carbon content of landfilled wood waste
$w\%, w$	weight of wet wood waste
EF_{CH_4}	methane emission factor
MGC_{CH_4}	methane gas recovered/collected
OX	oxidation factor
η_{CH_4}	methane gas collection efficiency for landfilled wood
EF_{CO_2}	carbon dioxide emission factor

Note: EF_{CO_2} represents combustion of total methane gas recovered/collected at Álfsnes (i.e. MGC_{CH_4})

Non-GHG emissions per Kg wood were estimated as:

$$EF_p = M_p * C_p * \eta_{ICE} * \eta_{CH4} \tag{3.19}$$

Where:

EF_p	emission factor for a pollutant
M_p	the ratio of molecular weight of pollutant to the weight of active element (i.e. SO2 to S or HCl to Cl)
C_p	the concentration of the active element in pollutant
η_{ICE}	the electrical conversion efficiency for internal combustion engine
η_{CH4E}	methane gas collection efficiency for landfilled wood

The methane gas collected at Álfsnes is combusted to generate electricity only. The electrical energy (kWh/year) that could be obtained from the combustion of collected methane gas is estimated as:

$$KWe/yr = \frac{MGU_{CH_4} * LHV_{CH_4} * \eta_{ICE}}{3.6} \tag{3.20}$$

Where:

KWe/yr	the annual electrical energy generated
MGU_{CH_4}	methane gas utilized/combusted as fuel at Álfsnes
LHV_{CH_4}	the lower heating value of methane gas
η_{ICE}	the electrical conversion efficiency for internal combustion engine
MGC_{CH_4}	methane gas recovered/collected
UF_{CH_4}	fraction of methane gas recovered/collected utilized as fuel

4 Materials and methods

This section describes the materials and methodological approach adopted for the study. The main materials employed were the case study product systems described separately in Section 3. The method applied is an attributional life cycle assessment (LCA). The theoretical framework of LCA is presented in Section 4.1. The goal and scope definition covering the functional unit, system boundary and allocation method are discussed in Section 4.2. Section 4.3 presents data parameters, assumptions, modelling, and life cycle inventory data, while the environmental impact assessment methods applied are reviewed in Section 4.4.

4.1 Life cycle assessment framework

Life cycle assessment or LCA is a tool used to analyze and quantify the whole life cycle environmental impacts of a product or system (Klöpffer, 1997). This LCA study follows the international standard organization methodological framework as prescribed in ISO standards 14040/14044:2006. The four phases of the ISO framework are goal and scope definition, life cycle inventory (LCI) analysis, life cycle impact assessment (LCIA), and the interpretation of results, as presented in Figure 4.1. Details of each phase are discussed below.

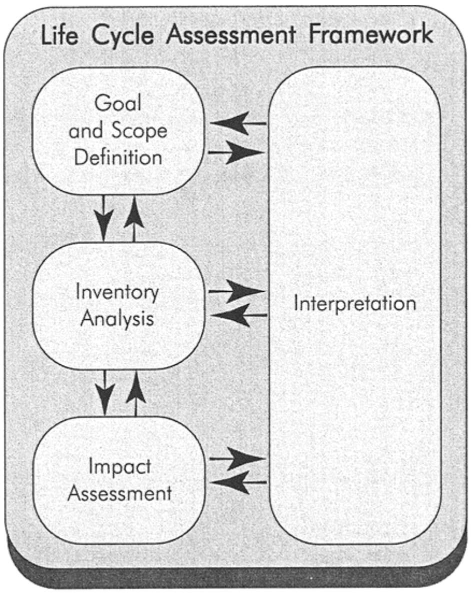


Figure 4.1 Life cycle assessment framework
Source: ISO 14040/14044 (2006)

4.2 Goal and scope definition

The main goal of this study is to evaluate and compare the life cycle GHG emissions and other environmental impacts of both the production and use phase of a small-scale CHP pallet gasifier to two reference systems. i.e., SHI and methane-recovery landfill. The secondary goal of the study is intended to identify environmental hotspots across the life cycle phases of each technology. The life cycle phases considered are from gate to grave.

4.2.1 Functional unit

The functions of the analyzed systems are the production and use of SNG (intermediate product), and electricity and heat (final co-products) for the CHP pallet gasifier; heat (final product) for the stepped hearth incinerator; and electricity (final product) for the landfill. An input-based functional unit is considered for the study, as it is the most appropriate conventional measure when the use of a common functional unit is not representative for different outputs due to the thermodynamic differences of energy quality (exergy). For example, 1 kWh of electricity versus 1 kWh of heat (Karlsdottir et al., 2020). Thus, SNG, electricity, and heat have different exergy properties. Hence, the functional unit adopted is the treatment of 1 Kg of waste wood. This unit will form the basis for comparing the environmental impacts to determine the net environmental benefits, which is calculated as the difference between the impacts caused by the gasifier and the reference systems (Steubing et al., 2011).

4.2.2 System boundary

A system boundary describes the system that is studied and the input flows of materials and energy that pass through the system and output flows of products, waste, and emissions that pass out of the system (Sundqvist, 1999). In this study, the life cycle environmental impacts for the three systems are assessed. Figure 4.2 shows the system boundaries of the systems under study. Each phase of the individual system requires the use of material and energy inputs and the generation of products and emission outputs. The key resource inputs into the systems comprise waste wood, energy, and auxiliary materials, while energy, waste, and emissions represent key outputs flowing in and out of the different unit processes of each product system.

Certain assumptions are made regarding the inclusion and exclusion of some life cycle phases and processes. The life cycle phases are considered from gate to gate, i.e., environmental impacts arising from the operation of the systems are the main concern of the study. In other words, life cycle stages occurring before the point of waste wood generation (i.e. collection and transportation of MSW, sorting, and recycling) are not considered in the study. Literature underscores that the major sources of environmental impacts of biomass-based power plants emanate from the operational phases. i.e., life cycle production and combustion of the resultant biofuel electricity (Hauschild et al., 2018; Liamsanguan & Gheewala, 2008).

The construction, maintenance and end-of-life phases are excluded from the analysis. Typically, the plant's construction and decommissioning phases are inconsequential to most environmental burdens related to the supply of heat and electricity (Hauschild et al., 2018; Liamsanguan & Gheewala, 2008).

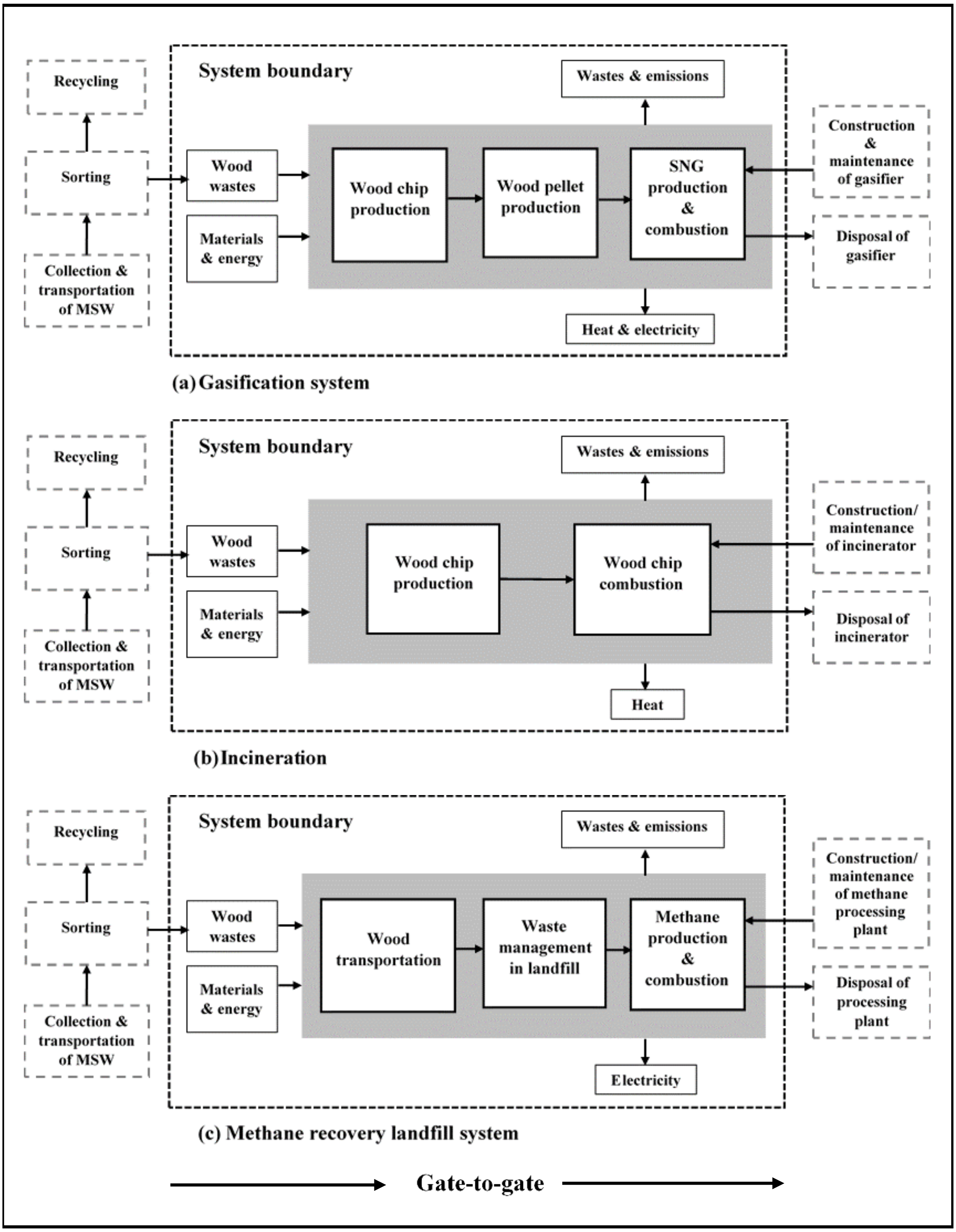


Figure 4.2 System boundary diagram for (a) CHP Pallet gasifier, (b) stepped hearth incinerator, and (c) methane recovery landfill systems

Moreover, the methane processing and SNG cleaning phases as well as leachate treatment and disposal of ash from the gasifier and incinerator are beyond the scope of the study. This is intended to reduce the complexity of the analysis.

Furthermore, common processes are eliminated. For instance, a pre-treatment stage to remove paint from wood is required for all the systems. For life cycle data accounting, either the pre-combustion or post-combustion approach can be applied to deal with the paint portion of the wood. For the latter approach, the painted wood can be combusted, and specific post-combustion treatment of the gas can be carried out to remove impurities. However, the energy content and GHG emissions of painted wood do not vary considerably as compared to unpainted wood and therefore inconsequential, except that paints and preservatives are mainstream sources of heavy metals such as cadmium (Cd) and copper (Cu) (Maïté et al., 2017). This study, nevertheless, adopted the pre-combustion approach where paint is removed before combustion. This is to keep the analysis comparable.

4.2.3 Allocation method

ISO 14044 recommends choosing between a number of allocation methods for multifunctional processes, such as physical allocation (e.g., using energy or exergy), economic allocation, and system expansion (ISO). Finding an equitable allocation method of sharing environmental impacts of co-production processes is a fundamental methodological challenge in LCA, as there is no unanimity regarding a standard allocation procedure of environmental impacts of electricity and heat produced by CHP plants (Heinonen et al., 2015; Karlsdottir et al., 2020).

The system enlargement method was applied to exclusively allocate environmental impacts of 1 kg of waste wood treatment to each system. System enlargement or avoided burdens is one of the approaches for accounting for system expansion (Reap et al., 2008). The equal importance of co-production of both electricity and heat from SNG in the region is assumed. As such, the gasifier system is expanded to include the environmental burdens of the two products. This approach avoids apportioning the overall environmental burden of the multifunctional processes (Lu & El Hanandeh, 2017; Assamoi & Lawryshyn, 2012).

4.3 Life cycle inventory

Life cycle inventory (LCI) details the quantitative units of inputs of materials and energy and outputs of products, wastes, and emissions for each life cycle phase and the corresponding unit processes of the product systems. The LCI data was compiled and computed from plant-specific technical reports, greenhouse gas inventory guidelines, and LCA literature.

4.3.1 LCI of CHP pallet gasification system

4.3.1.1 Wood chip production

As already discussed in Section 2.1.1, it is assumed that before the waste wood is chipped it undergoes a pretreatment phase where the paint portion and other preservatives of the treated timber are removed. It is also assumed that the average thickness of painted timber is 11 mm; the paint portion covers a thickness of 1 mm (i.e., 9%) on the timber, leaving a thickness of 10 mm as the unpainted portion (i.e., 91%). A wood chipper consumes 5 KWh of electricity to chip 100 Kg of waste wood into wood chips, with a conversion efficiency of 98% (Whittaker et al. 2011). Electricity for wood chipping is obtained from geothermal and hydropower resources which are clean energy sources in Iceland (Safarian et al., 2020). With a moisture content of about 5-17%, the wood chip has a net caloric value of 16.2 MJ/Kg (Alakangas, 2005). LCI data of wood chip production is shown in Table 4. 1.

Table 4.1 LCI of Wood chip production per 1 Kg of waste wood

Parameters	Unit	Amount
Resource & energy inputs		
Electricity consumption	kWh	0.04000
Product & emission outputs		
Wood chips	kg	0.98000
Wood dust	kg	0.02000
Particulate Matter (PM)	kg	0.00001
Particulate Matter (PM10)	kg	0.00001
Particulate Matter (PM2.5)	kg	0.00001

4.3.1.2 Wood pellet production

The wood chips are transferred to the pellet mill where they are reduced to smaller particle sizes between 6.4 and 3.2mm and subsequently compacted to form mold wood pellets and cooled down to safeguard their quality and durability (Magelli et al., 2009; Pa et al., 2011). One small pellet mill is required for the number of available wood chips. The mill consumes 7.5 kWh of electricity for every 100 Kg of wood chips pelletized. The pellets have a HHV of 18.34 MJ/Kg (Gautam et al., 2011), with a chip-to-pellet efficiency of 97% (Lu & El Hanandeh, 2017). Table 4. 2 shows the LCI data used in the pelletizing process.

Table 4.2 LCI of wood pellet production per 1 Kg of waste wood

Parameters	Unit	Amount
Resource & energy inputs		
Electricity consumption	kWh	0.07500
Product & waste outputs		
Wood pellets	kg	0.97000
Wood dust	kg	0.03000
Particulate Matter (PM)	kg	0.00047
Particulate Matter (PM10)	kg	0.00050
volatile organic compounds (VOCs)	kg	0.00054

4.3.1.3 SNG production and combustion

The dried wood pellets are transferred to the SNG plant. It is assumed that pre-treatment and gasification facilities are situated close to the waste collection facility in Vestmannaeyjar. Hence, no transportation is required. The downdraft CHP gasifier has a gas composition of 21% CO, 2% CH_4, and 17% of H_2 with energy properties of 12.7, 35.8, and 10.8 MJ/m3 in 1atm giving total energy of 5.219 MJ/m3 for the SNG (Pradhan et al., 2015). Based on data modelling, 5 gasifiers are needed for the thermal conversion of the available wood pellets to

SNG. The gasifier uses 18 kWh of electrical energy for every 22 kg of wood pellets (All Power Labs, 2021). For a startup, the gasifier requires a small quantity of diesel fuel 0.0002 l/Kg (Safarian et al. 2020). With tar and ash content of 5% and 0.4%, respectively, the gasifier has a wood-to- SNG conversion rate of 95% (Gautam et al., 2011) and SNG density of 0.95 kg/m^3 (Mustafa et al., 2017). Combustion of 60m3 of SNG generates 18 kWh_el and 20 kWh_th of electricity and heat energy, respectively. The overall efficiency of 49%, electrical efficiency of 23%, and power-to-heat ratio of 90% were modelled. The LCI data of the gasification process is depicted in Table 4. 3.

Table 4.3 LCI of SNG production and combustion per 1 Kg of waste wood

Parameters	Unit	Amount
Resource & energy inputs		
Electricity consumption	kWh	0.08300
Diesel fuel consumption	kg	0.00018
Water consumption	kg	0.14794
Air consumption	m3	0.00011
Stainless steel	kg	0.00272
Aluminum (wrought alloy)	kg	0.00007
Aluminum (cast alloy)	kg	0.00003
Product, emission, and waste outputs		
SNG produced (3.55841 kWh)	MJ	12.81027
Electricity generated (0.37555 kWh)	MJ	1.35196
Heat generated (0,41727 kWh)	MJ	1.50218
Tar	kg	0.05000
Carbon dioxide (CO2), biogenic	kg	0,84915
Carbon monoxide (CO), biogenic	kg	0,56881
Methane (CH4), biogenic	kg	0,13001
Nitrous oxide (N2O)	Kg	0.00058
Nitrogen oxides (NOx) as NO2	Kg	0.00061
Ash	Kg	0.00440

4.3.2 LCI of Stepped hearth incineration system

4.3.2.1 Wood chip production

Like the gasification process, waste wood is first shredded into chips before the combustion process begins. It is assumed that the same chipping machine is used for both the gasification and the incineration processes. Hence, the same inventory data for wood chip production.

4.3.2.2 Wood combustion

In the combustion phase, wood chips are delivered to the combustion chamber via the waste reception hopper and waste conveyor. The incinerator consumes 117 kWh of electricity, 11 l/h of diesel fuel, 180 l/h of water, and various quantities of other materials to burn 600 kg/h of wood chips (Matthews Environmental Solutions Ltd, 2018). The thermal efficiency has been assumed to be 85% (Ecoinvent data v2., v2007). Table 4.4 shows the LCI data for the wood combustion phase.

Table 4.4 LCI of wood combustion per 1 Kg of waste wood

Parameters	Unit	Amount
Resource & energy inputs		
Electricity consumption	kWh	0.19500
Diesel fuel consumption	kg	0.01650
Water consumption	kg	0.30000
Compressed air consumption	m^3	0.10667
Sodium bicarbonate consumption	kg	0.02233
Activated carbon consumption	kg	0.00090
Urea solution consumption	kg	0.00617
Energy, waste & emission outputs		
Thermal (heat) energy (3.17475 kWh)	MJ	11.42910
Bottom ash (IBA)	kg	0.15333
APCr discharge	kg	0.01867
Carbon dioxide (CO2), biogenic	kg	1,61700
Carbon monoxide (CO), biogenic	kg	1,14333
Methane (CH4), biogenic	kg	0,58800
Nitrous oxide (N2O)	kg	0.03771
Nitrogen oxides (NOx) as NO2	kg	0.03943
Sulfur dioxide (SO2)	kg	0.00200
Particulate matter (PM10)	kg	0.00005
Particulate matter (PM2.5)	kg	0.00001
Volatile Organic Compounds (VOC)	kg	0.00088
Dinitrogen monoxide	kg	0.00001

4.3.3 LCI of Landfill system

4.3.3.1 Wood transportation

Waste wood is shipped offshore in a 24-ton truck trailer (with a volumetric capacity of 60m3) in a RoRo vessel from Vestmannaeyjar to Landeyjarhöfn over 13 Km, with fuel consumption of 45 g/tKm. It is then transported from Landeyjarhöfn by road over 134 Km to the landfill site, with fuel consumption of 210 g/Km (VTT Technical Research Centre of Finland Ltd, 2017); Rome2rio Pty Ltd, 2020; IPCC, 2007). Tables 4.5 and 4.6 present the LCI data for wood transportation by sea and road, respectively.

Table 4.5 LCI of sea transportation per 1 Kg of waste wood

Parameters	Unit	Amount
Resource & energy inputs		
Diesel fuel consumption	kg	0.0455
Emission outputs		
Carbon dioxide (CO2), fossil	kg	0.0035
Methane (CH4), fossil	kg	3.17E-07
Nitrous oxide (N2O)	kg	8.44E-08
Carbon monoxide (CO), fossil	kg	7.91E-06
Total hydrocarbons (HC)(including methane CH4)	kg	2.65E-06
Nitrogen oxides (NOx)	kg	0.0001
Particulate matter (PM)	kg	1.74E-06
Sulfur dioxide (SO2)	kg	2.10E-06

Table 4.6 LCI of road transportation per 1 Kg of waste wood

Parameters	Unit	Amount
Resource & energy inputs		
Diesel fuel consumption	kg	0.21000
Emission outputs		
Methane (CH4), fossil		0.00049
Nitrous oxide (N2O)	kg	0.00003
Carbon dioxide (CO2), fossil	kg	0.11730
Carbon monoxide (CO), fossil	kg	0.00011
Nitrogen oxides (NOx)	kg	0.00042
Particulate matter (PM 2.5)	kg	0.00016
Ammonia (NH3)	kg	0.00009
Non-methane volatile organic compounds (NMVOC)	kg	0.00001
Dinitrogen monoxide	kg	2.30E-06

4.3.3.2 Waste management

Timber waste transported from Landeyjarhöfn is dumped at the bottom layer of the landfill. About 48.3 MJ/t of diesel is consumed for compacting landfilled wood waste using a bulldozer machine (Jeswani & Azapagic, 2016). The landfill is then levelled with an average soil cover of 0.125m3/t to facilitate the settling down of the wastes (Sundqvist, 1999). Table 4.7 displays the LCI data.

Table 4.7 LCI of waste management per 1 Kg of waste wood

Parameters	Unit	Amount
Resource & energy inputs		
Electricity	kWh	0.00035
Diesel fuel consumption	kg	0.00128
Soil cover	m^3	0.00013
Aluminum sulfate	kg	1.17E-07
Chemical, organic	kg	1.51E-09
Emission outputs		
Carbon monoxide (CO), fossil	kg	0.00001
Carbon dioxide (CO2), fossil	kg	0.00061
Nitrogen oxides (NOx)	kg	0.00001
Sulfur dioxide (SO2)	kg	2.00E-06
Methane (CH4), fossil	kg	2.20E-06
Non-methane volatile organic compounds (NMVOC)	kg	1.62E-08
Particulates (PM2.5)	kg	2.87E-07
Total organic carbon (TOC)	kg	0.10100
Dinitrogen monoxide	kg	9.10E-08
Mercury	kg	3.19E-07
Calcium, ion	kg	0.00013
Arsenic, ion	kg	4.21E-07
Lead	kg	0.00003
Ammonium, ion	kg	0.00039

4.3.3.3 Methane gas production and combustion

Immediately after the waste had settled, the anaerobic decomposition process starts, culminating in the generation of 0.173 kg of landfill gas for every kg of waste wood landfilled as modelled. landfilled wood has a methane gas collection efficiency of 67%. A portion of the uncollected methane is emitted into the atmosphere, while the remaining portion is oxidized into carbon dioxide (Lee et al., 2017). landfill gas at Álfsnes has a chemical composition of methane (57%), carbon dioxide (41%), and other gases (2%) (Metan Hf, 2020). Generation of landfill gas simulated assumes a decomposition process of degradable carbon of waste wood remaining in the landfill using IPCC First Order Decay (FOD) method, and default activity data and parameters (Lee et al., 2017). In Álfsnes, 40% of landfill gas collected is combusted to generate electricity. The remaining 60% is combusted to flare (Metan Hf, 2020), to convert the methane to carbon dioxide to reduce the impact of global warming (Lee et al., 2017). Based on the modelling, 1 kg of landfill gas with a heating value of 37.2 MJ/m3 (Ayodele et al., 2017) can generate 0.104 kWh of electricity as calculated. Table 4. 8 presents the LCI data on methane gas production and combustion.

Table 4.8 LCI of methane gas production and combustion per 1 Kg of waste wood

Parameters	Unit	Amount
Resource & energy inputs		
Electricity	kWh	0.05897
Lubricating oil	kg	0.00001
Energy, waste & emission outputs		
Electricity generated (0.24103 kWh)	MJ	0.37323
Carbon dioxide (CO2), biogenic	kg	0,11610
Carbon monoxide (CO), biogenic	kg	0,00101
Methane (CH4), biogenic	kg	0,05146
Sulfur dioxide (SO2)	kg	0.04950
Nitrogen oxide	kg	0.00014
Hydrochloric acid (HCl)	kg	0.00127
Waste mineral oil	kg	0.00027
Dinitrogen monoxide	kg	0.00777
Non-methane volatile organic compounds (NMVOC)	kg	0.00002

4.4 Life cycle impact assessment (LCIA) method

The LCIA methods applied to evaluate and compare the environmental impacts of the analyzed systems were the Centre of Environmental Science (CML 2001) and Cumulative Energy Demand (CED) models. The CML 2001 method was adopted because “it aims to provide best practice for midpoint indicators, operationalizing the ISO14040”. Also, it offers the opportunity to analyze the effects of the various impact categories over different time horizons (Institute for Environment and Sustainability, 2010). The CML 2001 provides a set of ten midpoint categories that serve as environmental indicators. The impact categories considered are climate change, depletion of abiotic resources, acidification, eutrophication, freshwater aquatic ecotoxicity, terrestrial ecotoxicity, marine aquatic ecotoxicity, stratospheric ozone depletion, human toxicity, and photochemical oxidation. Midpoint impact categories quantify the occurrence of changes in the environment (Assamoi & Lawryshyn, 2012). The CED method quantifies the total use of primary energy resources divided into six impact categories: namely, non-renewable (fossil), non-renewable (nuclear), non-renewable (primary forest), renewable (biomass), renewable (geothermal, solar, wind), and renewable (water). Besides, the ILCD 2.0 2018 midpoint method was used to explore the biogenic and fossil sources of global warming impact. The OpenLCA software v.1. 1. 10 and Ecoinvent database v. 3.7.1 were used for analyzing the inventory dataset developed for the study. Tables 4.9 and 4.10 show the summary descriptions of the environmental impact categories used, respectively, in the CML 2001 and CED (adapted from Karlsdottir et al., 2020; Pré, 2014; Safarian et al., 2019b; Safarian & Unnthorsson, 2018).

Table 4.9 Summary descriptions of the impact categories in the CML 2001 method

Impact category	Abbreviation	Environmental impact
Climate change	GWP100	Emission of greenhouse gasses (such as CO2, CH4, and NO2) to air, expressed in Kg carbon dioxide equivalents /kg emission.
Abiotic depletion	ADP	ADP is sub-divided into two categories: ADP for elements relates to the extraction of minerals, expressed in kg antimony equivalents; ADP for fossil fuels relates to the extraction of fossil fuels, expressed in MJ.
Acidification	AP	Emission of acidifying substances to air and the resulting pollution of chemicals (such as SO2, NH3, H2SO4, H2S, HCL, SO3, and NOx) to streams and soils, expressed in Kg SO2 equivalents/kg emission.
Eutrophication	EP	Extreme quantities of macro-nutrients in the environment triggered by emissions of nutrients (such as NOx, NH3, N2, and NO3) to air, water, and soil, expressed in kg PO4 equivalents/kg emission.
Stratospheric ozone depletion	ODP	Impacts of ozone depletion on human health and ecosystems, expressed in kg CFC-11 equivalents/kg emission.
Human toxicity	HTP	Exposure and impacts of toxic substances on the human environment expressed in 1.4-dichlorobenzene equivalents/kg emission.
Freshwater aquatic ecotoxicity	FAETP	Impacts of toxic emissions to air, water, and soil on freshwater ecosystems, expressed in 1.4-dichlorobenzene equivalents/kg emission.
Marine ecotoxicity	MAETP	Impacts of toxic substances on marine ecosystems, expressed in 1.4-dichlorobenzene equivalents/kg emission.
Terrestrial ecotoxicity	TETP	Impacts of toxic substances on terrestrial ecosystems, expressed in 1.4-dichlorobenzene equivalents/kg emission.
Photo-oxidant formation	POCP	Impact of reactive substances (mainly ozone) formation on human health and ecosystems, expressed in kg ethylene equivalents/kg emission.

Table 4.10 Summary descriptions of the impact categories in the CML 2001 method

Impact category	Abbreviation	Environmental impact
Non-renewable, fossil	$CED_{NR, fossil}$	Cumulative energy use is based on the upper heating value of various fossil fuel resources expressed in MJ equivalents/kg emission.
Non-renewable, nuclear	$CED_{NR, nuclear}$	Cumulative energy use is based on the energy value of natural uranium and a nuclear fuel chain expressed in MJ equivalents/kg emission.
Non-renewable, primary forest	$CED_{NR, bio}$	Cumulative energy use based on the upper heating value of wood from primary forest expressed in MJ equivalents/kg emission.
Renewable, biomass	$CED_{R, bio}$	Cumulative energy use based on the upper heating value of wood from sustainable resources, food products, agricultural byproducts, etc., expressed in MJ equivalents/kg emission.
Renewable, geothermal, solar, wind	$CED_{R, g,s,w}$	Cumulative energy use is based on converted solar energy, the kinetic energy of wind, and the amount of geothermal energy delivered to a heat pump, expressed in MJ equivalents/kg emission.
Renewable, water	$CED_{R, water}$	Cumulative energy use is based on the converted potential energy of the water in a hydropower reservoir, expressed in MJ equivalents/kg emission.
Total, non-renewable	$CED_{NR, total}$	Single score results for the sum of all non-renewable categories expressed in MJ equivalents/kg emission.
Total, renewable	$CED_{R, total}$	Single score results for the sum of all renewable categories expressed in MJ equivalents/kg emission.
Total	CED_{total}	Single score results for the sum of all CED categories above expressed in MJ equivalents/kg emission.

4.4.1 Sensitivity analysis and assumptions

To identify the robustness of the result, sensitivity analysis was performed to ascertain the effects of variations of parameters of interest—regarding future technological changes and environment concerns—on environmental performance and energy generation. This exercise satisfies the ISO standard requirement for testing data quality and the effect of different impact parameters. The parameters and assumptions for the sensitivity analysis for each system are discussed below.

4.4.1.1 CHP pallet gasification system

Overall system efficiency. A baseline efficiency of 49% modeled from the exergy properties of woody SNG fuel was used in the analysis. Eriksson & Finnveden (2017) reported that the efficiency of the CHP system for biomass conversion ranges from 27% - 34%. However, the generic biomass conversion efficiency of the gasifier system is 35%. Hence, a value of ±14% was used in the sensitivity analysis (All Power Labs, 2021).

The carbon content of tar. The baseline analysis used 5%. As stated earlier, the downdraft gasifier dumps 50 Kg of tar from 1000 Kg of wood gasified. A rate of ±2% to assess less CO_2 impact was used in the sensitivity analysis.

4.4.1.2 Stepped hearth incineration system

Thermal efficiency. Average thermal efficiency of 85% was used in the baseline analysis. An efficiency rate of 80% is assumed for the sensitivity analysis on the basis that 80% water condensation will amount to the same efficiency rate of heat exchangers of the incinerator. A variation of ±5% of the baseline rate was used in the sensitivity analysis.

Control efficiency for pollutant abatement. An emission control rate of 55% was used in the baseline analysis. The SH incinerator is fitted with emission systems that can reduce emissions further (Matthews Environmental Solutions Ltd, 2018). Control efficiency of ±5% was used in the sensitivity analysis.

4.4.1.3 Methane recovery landfill system

Methane gas generation. In the baseline analysis, the fraction of the degradable carbon decomposed (DOCF) parameter of 0.4 was used. These variables are subject to huge uncertainties that could affect the methane and carbon dioxide concentration in the methane gas (Lee et al., 2017). The upper ends of DOCF (±0.57) as reported in Lee et al. (2017), were used to perform the sensitivity analysis.

Methane gas collection efficiency and oxidation. The baseline analysis assumed that the landfill in Álfsnes is under wet boreal conditions with moderate methane gas collection efficiency of 67% and a default oxidation factor of 10% (See: Lee et al., 2017). In the sensitivity analysis, higher collection efficiency of ±74% is assumed under an active wet boreal condition. Chanton et al. (2009), cited in Lee et al. (2017), reported an average oxidation factor of 36%. The value of ±36% was used in the sensitivity analysis.

It is interesting to emphasize that there is non-exhaustive list of parameters for the respective systems—apart from the parameters discussed above—that are sensitive to the environmental impacts but are not covered in the study. The exclusion of these parameters represents potentially relevant uncertainty aspects of the results. Discussions on the uncertainty and limitations of the study can be found in Section 6.3.

5 Results

The results of the LCIA of each impact category following the selected LCIA methods are presented in Section 5.2. Hotspot analysis of the various impact contribution drivers of the life cycle stages of each analyzed system is presented in Section 5.3, whereas Section 5.4 shows the analysis of the energy recovery potential of each system.

5.1 Environmental impact assessment

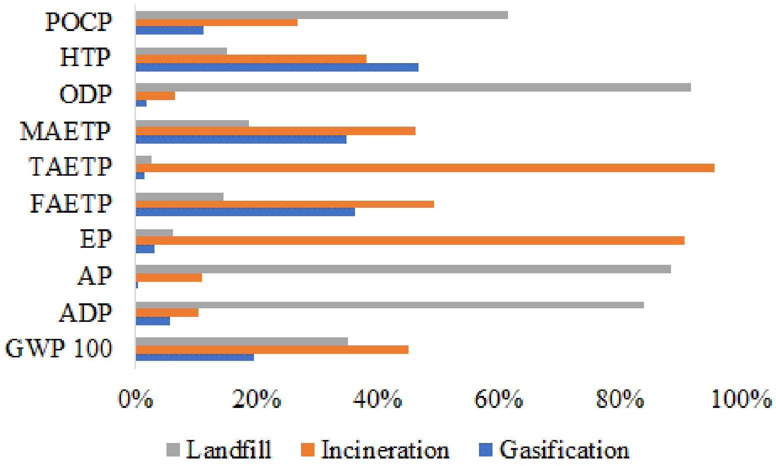
The comparative impact assessment of the baseline gasification technology with the incineration and landfill reference systems has been investigated with both characterization and nominalization impact categories. The results of the impact categories according to CML 2001 and CED are presented in Table 5.1 and Figure 5.1. In addition, CML 2001 impact categories with environmental impacts over different time horizons are presented in Figures 5.3, 5.6 – 5.9. Hotspot analysis of the main relative contributions of the life cycle stages (unit processes) of each system to the impact categories is depicted in Figure 5.2.

The impact categories are grouped into eight main impact categories of *global warming potential (GWP), abiotic depletion potential (ADP), acidification potential (AP), eutrophication potential (EP), ecotoxicity potential (ETP), ozone depletion potential (ODP), human toxicity (HTP) and energy resources (CED)*. The assessment is based on the functional unit of treatment of 1 Kg of waste wood.

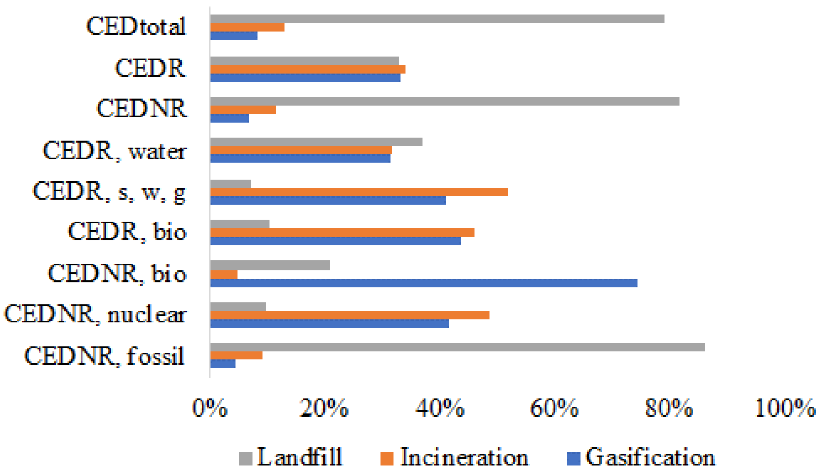
Table 5.1 The impact assessment results for 1 Kg of waste wood treatment

Impact categories	Gasification	Incineration	Landfill	Unit
CML 2001 impact categories				
Global warming	6.3E+00	**1.5E+01**	1.1E+01	kg CO2-Eq
Depletion of abiotic resources	7.1E-04	1.3E-03	**1.0E-02**	kg antimony-Eq
Acidification potential	1.1E-03	2.2E-02	**1.8E-01**	kg SO2-Eq
Eutrophication potential	1.7E-03	**4.7E-02**	3.2E-03	kg NOx-Eq
Freshwater aquatic ecotoxicity	1.6E-01	**2.2E-01**	6.6E-02	kg 1,4-DCB-Eq
Terrestrial ecotoxicity	1.4E-03	**8.2E-02**	2.5E-03	kg 1,4-DCB-Eq
Marine aquatic ecotoxicity	2.5E+02	**3.3E+02**	1.4E+02	kg 1,4-DCB-Eq
Stratospheric ozone depletion	6.0E-09	2.0E-08	**2.9E-07**	kg CFC-11-Eq
Human toxicity	**4.7E-01**	3.9E-01	1.5E-01	kg 1,4-DCB-Eq
Photochemical oxidation	1.5E-03	3.6E-03	**8.2E-03**	kg ethylene-Eq

CED impact categories				
Non-renewable, fossil	1,2E+00	2,5E+00	**2,4E+01**	MJ-Eq
Non-renewable, nuclear	7,2E-01	**8,5E-01**	1,7E-01	MJ-Eq
Non-renewable, primary forest	**2,6E-04**	1,7E-05	7,2E-05	MJ-Eq
Renewable, biomass	7,7E-02	**8,2E-02**	1,8E-02	MJ-Eq
Renewable, geothermal, solar, wind	2,9E-02	**3,6E-02**	4,9E-03	MJ-Eq
Renewable, water	4,6E-01	4,7E-01	**5,4E-01**	MJ-Eq
Total, non-renewable	1,9E+00	3,4E+00	**2,4E+01**	MJ-Eq
Total, renewable	5,7E-01	**5,8E-01**	5,7E-01	MJ-Eq
Total	2,5E+00	4,0E+00	**2,4E+01**	MJ-Eq



(a) CML 2001 impact category



(b) CED impact category

Figure 5.1 Relative changes of LCIA results for 1 Kg of waste wood treatment: (a) CML 2001 and (b) CED impact categories.

Overall, the gasification system environmentally performs better than landfill and incineration, in seven (GWP, ADP, AP, EP, TAETP, ODP, and POCP) of the ten CML impact indicators and two (non-renewable, fossil, and renewable, water) of the six CED impact indicators.

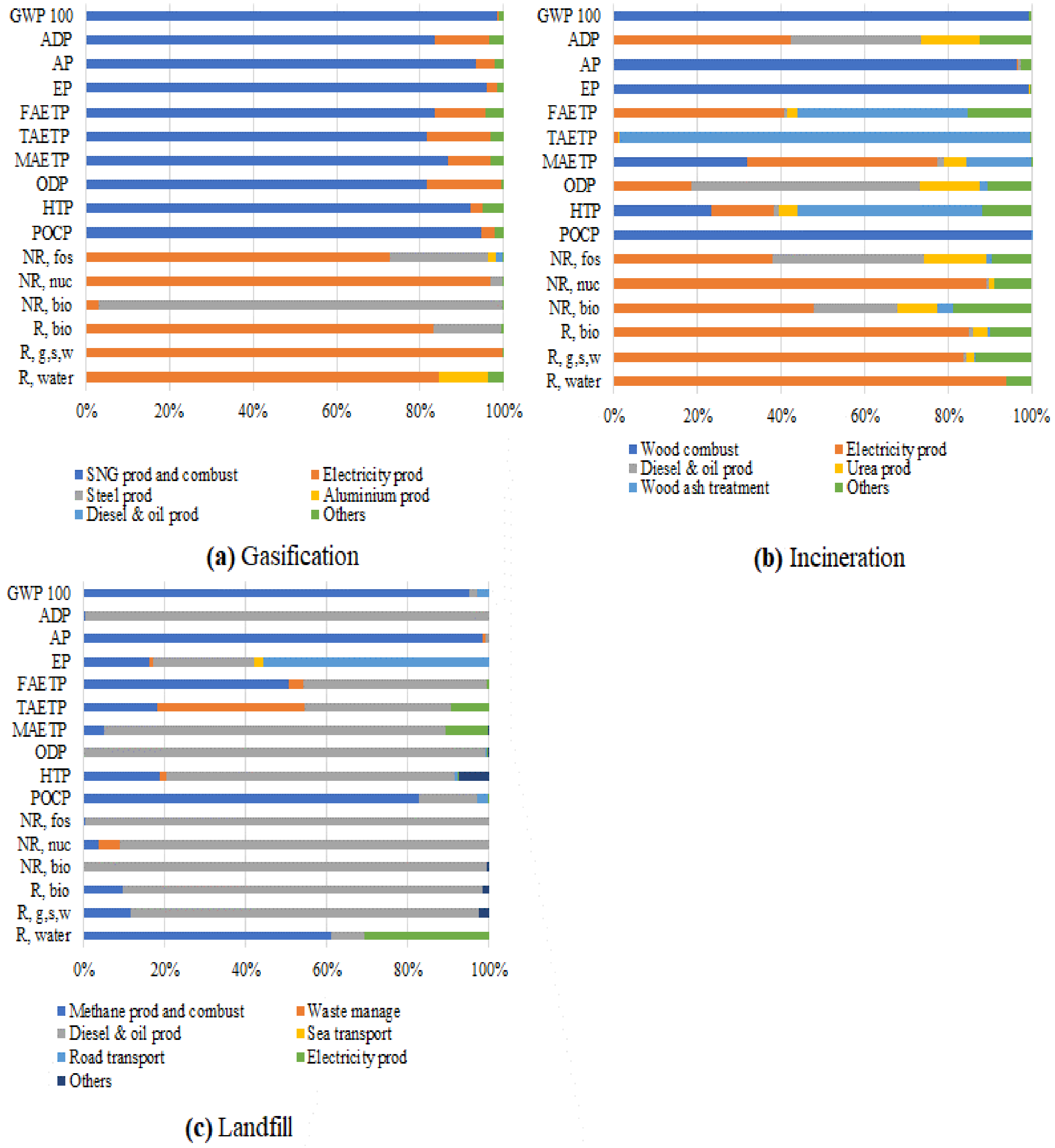


Figure 5.2 Contribution of the main unit processes to the main impact categories of (a) gasification, (b) incineration, and (c) landfill.

The hotspot analysis shows that the SNG production and combustion unit process of the gasification system accounts for a higher burden in all the CML 2001 impact categories, whereas electricity production dominated all the CED impact categories, except for biogenic non-renewable resources (NR, $_{bio}$) which were dictated by steel production.

For the incineration system, impacts are dominated by wood combustion, wood ash treatment, and electricity production, whereas electricity production has higher cumulative energy resource use in all the CED indicators, apart from non-renewable fossil resources (NR, $_{fossil}$) which shared the dominance with diesel and oil production. For the landfill system, GWP100, AP, POCP, and R, $_{water}$ was highly driven by the methane gas production and combustion process. Diesel and oil production was found to be the main impact contributions in ADP, MAETP, ODP, NR, and all the CED indicators but for R, $_{water}$. TAEP was largely found in waste management, and diesel and oil production.

5.1.1 Global warming potential (GWP)

GWP quantifies the amount of energy absorbed by emissions of 1 ton of gas in comparison to the emissions of 1 ton of CO_2 over a specified time (U. S. Environmental Protection Agency, 2020b). Under the Kyoto Protocol and the Paris Agreement, most studies use GWP over 100-year time horizon (i.e. GWP100) as the default metric for measuring the long-term effects of GHG emissions (U. S. Environmental Protection Agency, 2020b). As shown in Table 5.1, gasification has the lowest impacts of GWP100, whereas the impact of landfill is superior to incineration. The main source of the GHG emissions driving the higher GWP impacts of incineration is as a result of direct wood combustion (16.011 kg CO_2-Eq.), electricity production from lignite (0.025 kg CO_2-Eq.), and ammonia production (8.30E-3 kg CO_2-Eq.).

Further analysis is conducted in conformity with the Bern Carbon Cycle Model to understand the variations in the GWP impacts over different time horizons and emission sources. The alternative BCCM considers the amount and time horizon of emission occurrence and natural removal of GHGs from the atmosphere. The GWP20 and GWP500 focus on GHG impacts over shorter and longer time horizons, respectively (U. S. Environmental Protection Agency, 2020b). In figure 5.3, a 100-year, 20-year (i.e. GWP20), and 500-year (i.e. GWP500) time horizons are compared. The GWP over the 20 years has the utmost impact. The dominant environmental effect of GWP20 over GWP100 and GWP500 is based on the relative impact of more energy absorbed per ton of GHGs over a shorter time than a longer time horizon (U. S. Environmental Protection Agency, 2020b).

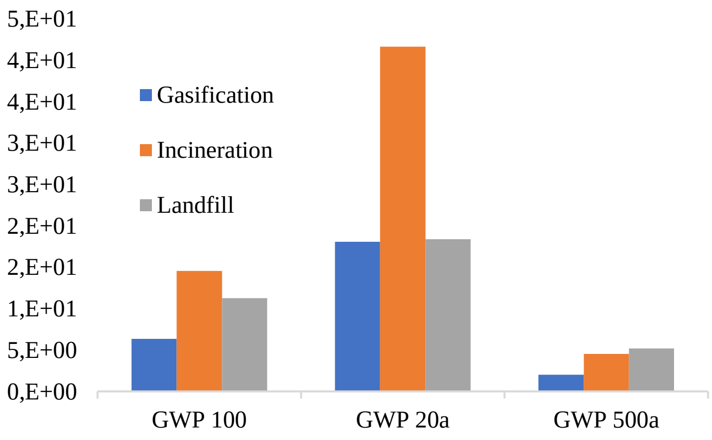


Figure 5.3 Time-horizon GWP impact categories

This phenomenon can be explained by the longevity of GHGs in the atmosphere. Higher climate change impact in the shorter-horizon means that the potency of short-lived gases such as CH_4, is much impactful in the shorter time than in the longer time period. On the contrary, long-lived gases such as CO_2—less potent gas than CH_4—dominates in the longer time period.

The main sources of GWP are depicted in Figure 5.4. GHGs from biogenic sources are the main contribution to GWP, with both gasification and incineration accounting for about 99% of the total GHG emissions of each respective system. In contrast, about 59% of the total GWP could be traced to fossil-based GHG emissions, which are largely driven by fossil fuel use in transportation and landfill operational activities.

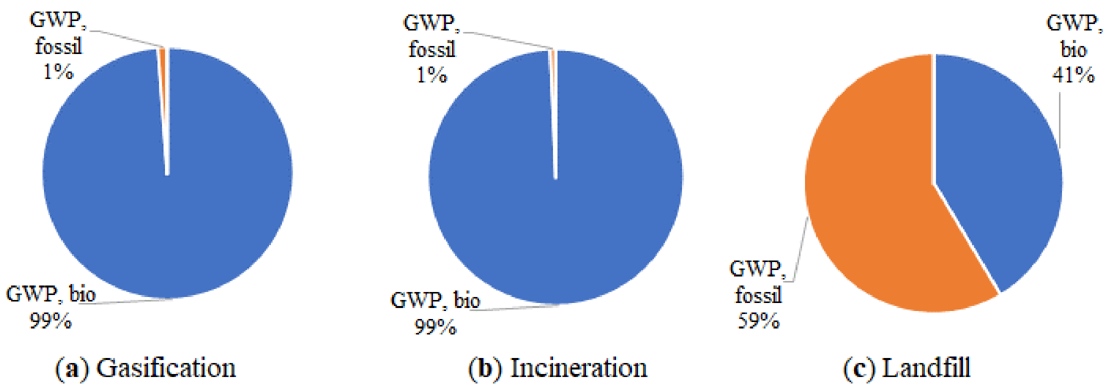


Figure 5.4 GWP sources for (a) gasification, (b) incineration, and (c) landfill.

Regarding the most crucial life cycle stages contribution to GWP impacts of each system, the SNG production and combustion, wood combustion, and methane gas production and combustion unit processes produced the highest GHG emission impacts, about 99%, 99%, and 95% of the total GWP100 of each respective system (Figure 5.2). These impacts are attributed to the main GHG emissions of CO_2, CH_4, and N_2O (Figure 5.4).

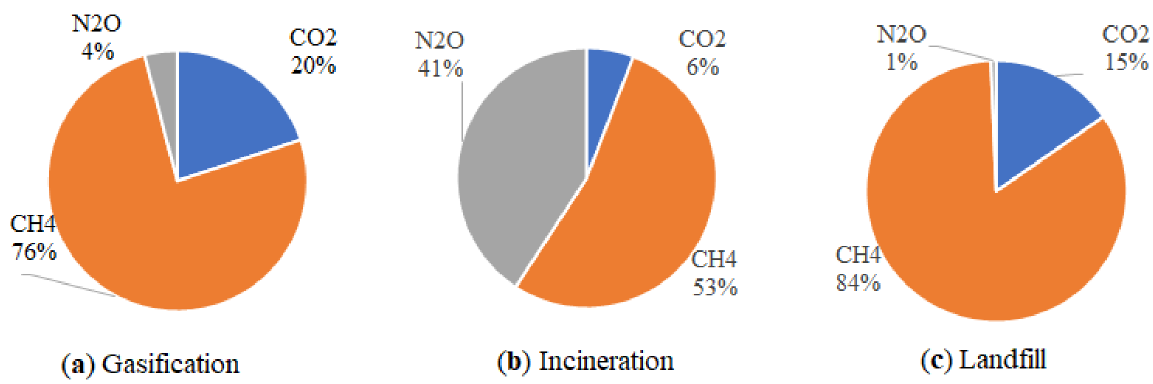


Figure 5.5 Relative contribution of the main GHG emissions of (a) gasification, (b) incineration, and (c) landfill.

For the gasification system, CH_4 emissions account for about 76% of the total GHGs, with contributions of 20%% and 4% from CO_2 and N_2O, respectively. Similarly, CH_4, N_2O, and CO_2 make up 53%, 41%, and 6%, respectively, of the incineration system, while CH_4, (84%), CO_2 (15%), and (1%) constitute the total GHGs of the landfill system. The incineration system dominates all the GHG emissions.

5.1.2 Abiotic depletion potential (ADP)

Landfill accounts for a higher share of ADP (See. Table 5.1). Abiotic resources represent the depletion of mineral resources for use for landfill management as well as the production, collection, and combustion of landfill methane gas. Resource depletion for petroleum and natural gas production accounts for about 85% (8.87E-03 kg antimony-Eq) of the total ADP for landfills. This is mainly due to the high contribution of crude oil extraction for fossil energy and natural gas for electricity generation of some of the system processes within the landfill value chain. Apart from the extraction of petroleum and natural gas, lignite mine operation, hard coal mining is a major source of ADP for gasification and incineration. From the process's contribution analysis, the SNG production and combustion accounts for a prominent ADP impact (84%). Electricity production (42%), and diesel and oil production (31%) hold a larger share for the incineration system. For the landfill system, almost all the ADP impact could be attributed to diesel and oil production (100%) due to the high consumption of fossil fuel for transportation and waste management operations (See. Figure 5.2).

5.1.3 Acidification potential (AP)

Gasification has the lowest AP impact as compared to the worst impact of landfills (See. Table 5.1). The AP impact in landfills is driven by the high emissions of nitrogen oxide and sulfur dioxide gases released to the atmosphere during methane production and combustion (0.178 kg SO_2-Eq.). These emissions eventually contaminate terrestrial and water bodies. Other AP sources include emissions from road transportation (7.1E-4 kg SO_2-Eq.), sea transportation (2.3E-4 kg SO_2-eq.), and heavy fuel oil (1.6E-4 kg SO_2-eq. According to the relative AP impact associated with each stage, the most critical process for all the system is the SNG production and combustion, wood combustion, and methane gas production and combustion stages (94%, 96%, and 99%, respectively) due to the emissions of acidifying substances to air and the resulting pollution of chemicals (such as SO_2, NH_3, H_2S, HCL, SO_3, and NO_x) to streams and soils (Figure 4).

5.1.4 Eutrophication potential (EP)

Gasification is the most favorable system with the least EP impact. Incineration is by far associated with the highest impact of EP, followed by landfill (See. Table 5.1). The major contributors to EP are the emissions from the wood combustion process (0.046 kg NO_X-Eq.), lignite mining (4.4E-5 kg NO_X-Eq.), urea production (7.7E-5 kg NO_X-Eq.), and diesel (1.5E-5 kg NO_X-Eq.). for landfill, nitrate and nitrogen oxides emitted from leachate and combustion of diesel fuel in transport and landfill operations respectively, are the major contributors. Figure 5.5 further shows EP classification by origin. The average EP within the EU region is very high as compared to the generic EP.

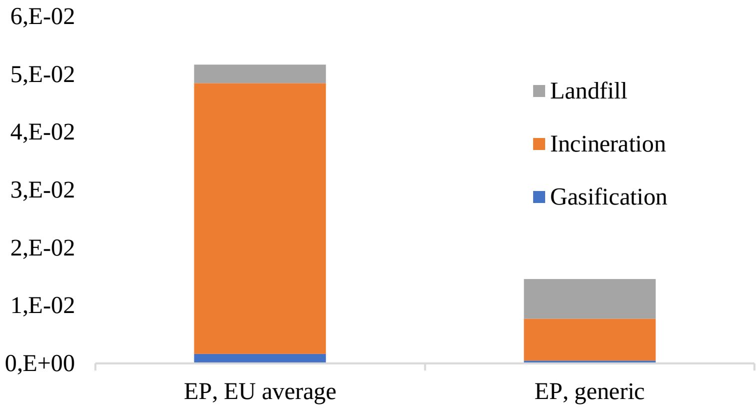


Figure 5.6 EP impact category by origin

Regarding the most critical stage of EP impact, SNG and wood combustion stages (96% and 99%, respectively) entail the highest environmental impact, whereas road transportation, and diesel and oil production are the major impact contributions (See. Figure 5.2).

5.1.5 Ecotoxicity potentials (ETPs)

All ecotoxicity-related impacts (freshwater, marine, and terrestrial) were high for the incineration system. Gasification performed better in TAETP, whereas FAETP and MAETP were the least impactful in landfills (See. Table 5.1). The biggest contributors to all ecotoxicity-related impacts included emissions from lignite mining and wood ash disposal. Other related impacts for each impact category include copper mining for freshwater (FAETP), ammonia production for terrestrial (TAETP), and sulfidic tailing for marine (MAETP).

Figures 5.6 compare the 100-year, 20-year, and 500-year time horizons of each ETP category. In all, the 500-year time horizon has the highest impacts for both TAETP and MAETP. For FAETP, both the 100-year and 500-year time horizons share the same magnitude of impacts. While gasification and incineration dominate FAETP and MAETP impact categories, TAETP is largely dictated by incineration. From the process's contribution breakdown, SNG production and combustion have enormous impacts in all ecotoxicity-related impacts. For the incineration system, the highest process contributions are distributed as follows: FAETP (electricity production and wood ash treatment, 41% each), TAETP (wood ash treatment, 98%), and MAETP (electricity production, 46%; wood combustion, 32%).

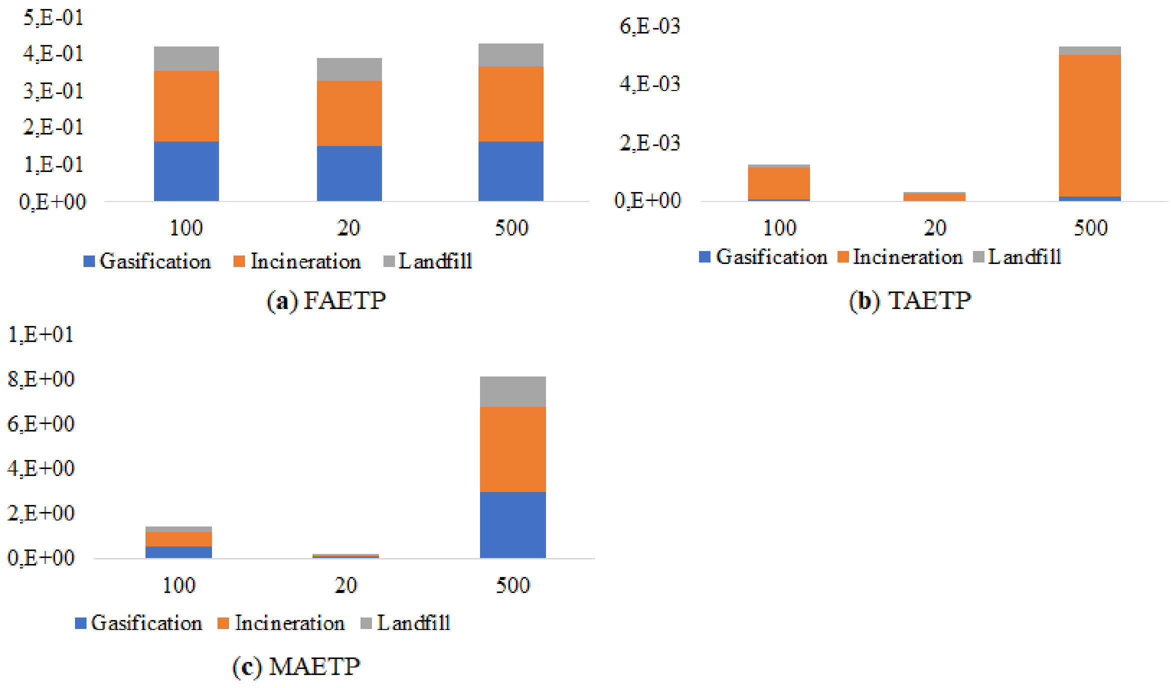


Figure 5.7 Time-horizon ETP impact category of (a) FAETP, (b) TAETP and (c) MAETP

The landfill system is dominated by FAETP (methane gas production and combustion, 51%; wood ash treatment 41%), TAETP (diesel and oil production, 36%; waste management, 36%; methane gas production and combustion, 18%), and MAETP (diesel and oil production, 84%) (See. Figure 5.2).

5.1.6 Ozone depletion potential (ODP)

From Table 5.1, the ODP impact score for landfills is excessively high in comparison with gasification and incineration. The estimated ODP of 2,9E-07 kg CFC-11-Eq for landfill is predominantly caused by the emissions from extraction, production, and use of petroleum and natural gas for transportation and operation of the landfill management and methane gas production and combustion-related activities. Emissions of non-methane volatile organic compounds (NMVOC) are the main contributors to this impact. The different time horizons of ODP impacts are shown in Figure 5.8. The ODP impacts for the respective system do not vary significantly across the different periods. SNG production and combustion, and diesel and oil production process constitute the highest ODP impact of 82% and 99%, respectively. Diesel and oil production, electricity production, and urea production share the ODP burden of 54%, 19%, and 15%, in that order, of the incineration (See. Figure 5.2).

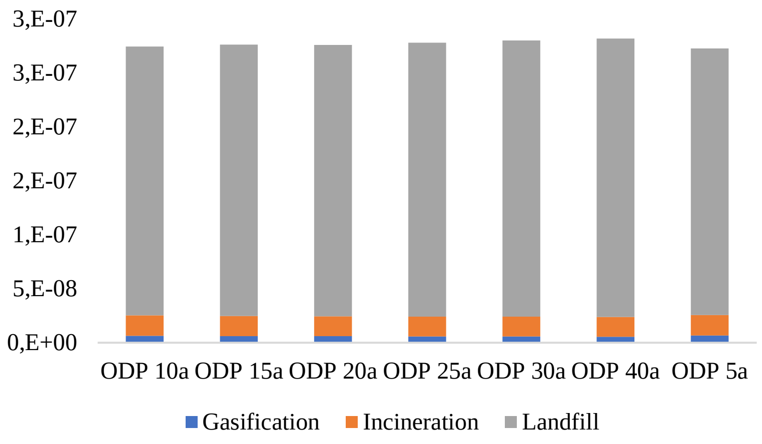


Figure 5.8 Time-horizon ODP impact category

5.1.7 Human toxicity potential (HTP)

Gasification has the highest impact on HTP (See. Table 5.1), mainly caused by the emissions of N_2O and other heavy metals related to alloy production, lignite mining, and sulfidic tailing. Figure 5.9 compares the time horizon of HTP. The HTP impacts across the different time horizons for all the analyzed systems do not vary substantially. The most critical stages are the SNG production and combustion (92%) for the gasification system; diesel and oil production (71%) for the landfill system, and wood ash treatment (44%), wood combustion (23%), and electricity production (15%) for the incineration system (See. Figure 5.2).

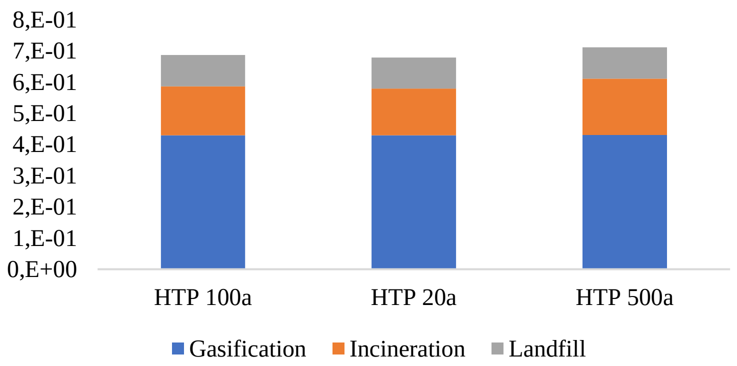


Figure 5.9 Time-horizon HTP impact category

5.1.8 Photochemical oxidation potential (POCP)

The estimated POCP of 4,5E-03 kg ethylene-Eq from incineration is the highest impact score (See. Table 5.1). About 99% of this impact is from the wood combustion process, whilst the remaining impacts are shared among iron sinter production, production, and market for natural gas. The major contributing emission burdens include SO2, VOC, and NOx emissions. The process's contribution breakdown indicates that SNG production and combustion (95%), wood combustion (99%), and methane gas production and combustion (82%) are the key life cycle stages that contribute most to POCP (See. Figure 5.2).

5.1.9 Energy resources (CED)

Overall, landfill is responsible for a higher depletion of total energy resources. For non-renewable energy, the highest impacts of fossil, nuclear, and biomass depletion are shared among landfill, incineration, and gasification, respectively. Similarly, incineration accounts for a substantial depletion of renewable energy resources, such as geothermal, solar, wind and water. While landfill accounts for the biggest depletion of total non-renewable energy resources, incineration constitutes the highest impacts of renewable energy resources (See. Table 5.1). Diesel and oil production were the major contributions to the impact categories of $R_{g,s,w}$ (86%), R_{bio} (89%), NR_{bio} (99%), $NR_{nuclear}$ (92%), and NR_{fossil} (99%). Methane gas production and combustion (61%) and electricity production (31%) responsible for the highest impact in R_{water}.

5.2 Energy recovery potential

The summary of annual total efficiency-adjusted energy recovery potential from wood gasification as compared to incineration and landfill is presented in Table 5.2 and Figure 5.10. Energy inputs account for the consumption of electricity at the various life cycle stages of each respective system. Energy outputs account for energy yields obtained from the combustion of SNG, wood chips, and methane gas, for gasification, incineration, and landfill, respectively. The values of energy outputs were calculated and adjusted for electrical and thermal efficiencies of the internal combustion engines along with their respective wood conversion efficiencies as detailed in the methodology section. For the CHP Pallet gasifier, overall electrical and thermal efficiencies of 23% and 26%, respectively, were calculated. For the Stepped Hearth incinerator, the thermal efficiency of 85% as reported in Ecoinvent data v2. (v2007) was used. The overall electricity production efficiency of 33% was adopted for the methane gas internal combustion engine, as reported in (Ayodele et al., 2017).

Table 5.2 Energy recovery potential

	Gasification	Incineration	Landfill
Energy input (GJ/yr)	579,30	193,28	49,61
Energy output (GJ/yr)	4.481,57	11.286,39	290,29
Net energy (GJ/yr)	3.902,27	11.093,11	240,68

While gasification and landfill of wood provide the overall lower environmental impacts, wood incineration generates optimum energy. The disparities of energy production among the three systems are largely explained by the differences in energy conversion efficiencies of each system´s internal combustion engines.

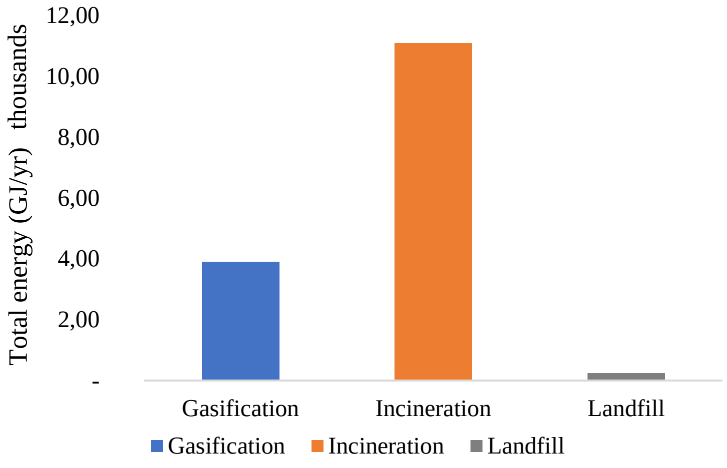


Figure 5.10 Net energy savings by comparison

Moreover, the pretreatment of waste wood is more energy-intensive with the gasification process. As compared to incineration, the added stage of pelletizing wood chips contributes to higher electricity consumption and ultimately decreases the net energy generated. This result draws attention to a major limitation of using gasification for waste wood treatment when energy recovery is the central objective.

6 Discussion

It is generally underscored in environmental literature that the environmental performance of combustion of biomass feedstock is far superior to the combustion of fossil fuel (De Best et al., 2008; Saidur et al., 2011; Sami et al., 2001). Nonetheless, the variations in environmental impacts of different biomass feedstock and the differences in conversion processes provide the basis for comparative study. The main goal of this study is to evaluate evaluates the life cycle GHG emissions and other environmental impacts of SNG production and energy generation from waste wood in comparison with incineration and landfill systems in the Vestmannaeyjar region in Iceland, using process LCA. Specific research questions are stated as follows:

a) Is the alternative CHP pallet gasifier system environmentally superior to the proposed SH incinerator system and the conventional landfill system?
b) Which lifecycle phases and processes of the assessed systems are environmental hotspots?
c) What is the energy recovery/generation potential of the CHP pallet gasifier system in comparison to the reference systems?

The overview of the main research findings relative to the research questions is discussed in Section 6.1. in Section 6.2, the study´s results are positioned and discussed within the context of previous but similar LCA studies. Section 6.3 examines the weaknesses, sensitivity, and uncertainty analyses of the study´s results.

6.1 Main findings and answers to the research questions

The overall results indicate that gasification was environmentally superior to the compared systems, in seven (GWP, ADP, AP, EP, TAETP, ODP, and POCP) of the ten CML impact indicators and two ($NR,_{fossil}$ and $R,_{water}$) of the six CED impact indicators. Apart from HTP and $NR,_{bio}$, incineration performed poorly in all the 16 impact categories assessed. Gasification and incineration had the lowest and highest impacts of GWP100, respectively. Analysis of the time horizon of GWP revealed that GWP20 had the utmost impact because of the relative impact of more energy absorbed per ton of GHGs over a shorter time than a longer time horizon. Regarding sources of GWP, gasification, incineration, and incineration dominated Luluc-GWP, biogenic-GWP, and fossil-GWP, respectively.

The hotspot analysis shows that the syngas production and combustion unit process of the gasification system accounts for a higher burden in all the CML 2001 impact categories, whereas electricity production dominated all the CED impact categories, except for $NR,_{bio}$ which was dictated by steel production. For the incineration system, impacts are dominated by wood combustion, wood ash treatment, and electricity production, although electricity production had higher cumulative energy resource use in all the CED indicators, apart from non-renewable fossil resources (NR, $_{fossil}$) which shared the dominance with diesel and oil production. For the landfill system, GWP100, AP, POCP, and R, $_{water}$ was highly driven by the methane gas production and combustion process. Diesel and oil production was found to be the main impact contributions in ADP, MAETP, ODP, NR, and all the CED indicators but for R, $_{water}$. TAEP was largely found in waste management, and diesel and oil production.

Wood incineration generates the optimum energy, despite the overall lower environmental impacts associated with gasification and landfill. The disparities of energy production among the three systems are largely explained by the differences in energy conversion efficiencies and the energy intensities of the feedstock pre-treatment stages of each system.

6.2 Positioning among previous literature

The overall results of the study indicate that the gasification and landfill systems had the lowest environmental impacts. This finding is generally supported by studies that compared similar bioenergy systems. Arafat et al. (2015) support this claim that gasification and anaerobic digestion of biomass feedstock is environmentally superior to other biomass conversion technologies. Safarian et al. (2020) conclude in their study that electricity generation from waste gasification is more environmentally beneficially than incineration in all the impact indicators evaluated. Similarly, Cleary and Caspersen (2015) identify the overall superior environmental performance of a small-scale gasifier over a wood pellet-fired power plant. According to Cleary and Caspersen (2015), the favorable environmental performance of wood gasification is attributed to benefits arising from using a dried feedstock and reduction in biomass processing.

In conflicting studies, Jeswani and Azapagic (2016) report that electricity or combined heat and power generated from incineration had a lower environmental impact than landfill in most of the impact categories. In a contrasting study, Aberilla et al. (2019) conclude that an anaerobic digestion system has the highest environmental benefits in a study that evaluated the environmental sustainability of direct combustion (incineration), gasification, and anaerobic digestion was compared to a conventional diesel generator. Zaman (2012) reports that the overall environmental benefits of sanitary landfill were larger than that of pyrolysis-gasification and incineration.

The results of previous LCA studies on individual impact categories have yielded mixed findings due to the selection and application of different LCIA methods and assumptions, thereby making direct comparison of results of individual impact categories difficult (Atilgan and Azapagic, 2016). However, the commonest impact category reported by all studies is the GWP (Karlsdottir et al., 2020) and other impact categories such as acidification, eutrophication, toxicity, ozone depletion, and abiotic depletion. Studies that report on similar impact categories were selected for the comparative analysis. The analysis of GWP suggests that the gasification system is the most environmentally sustainable option, albeit landfill performs better than incineration. Safarian et al. (2020) corroborate this finding that climate change impact of gasification is lower than incineration.

Earlier studies contradict this finding. Demetrious et al. (2018) report that the life cycle GHG emissions of landfill is lower than other waste recovery facilities in Sydney, Australia. Likewise, Zaman (2012) states that the GWP of pyrolysis-gasification is least impactful when compared to incineration, but landfill had a favorable environmental impact in comparison to gasification. Cambero et al. (2015) conclude that biomass boiler cogeneration system achieves a net GHG emissions savings than landfilling, direct combustion, gasification, and other alternative bioenergy systems. Burnley et al. (2012) report that combusting biomass waste in a cement kiln yielded an optimal reduction in climate change than gasification, combustion in a dedicated plant, and anaerobic digestion.

The lower impacts in other environmental impact categories—other than GWP—of the gasification system support the results reported in other studies. Jeswani and Azapagic (2016) report that the environmental burdens in gasification are lower in acidification, eutrophication, and eco-toxicity categories than incineration. In a similar study, Zaman (2013) concludes that gasification had a minimal environmental load in acidification, eutrophication, and aquatic eco-toxicity. These results explain that the environmental loads from the release of thermal gas emissions from incineration and disposal of final residues from landfills are higher as compared to gasification (Zaman, 2013). In contrast, Demetrious et al. (2018) state that the AP of landfills is lower than gasification and incineration. Moreover, Burnley et al. (2012) detail in their study that incineration had minimum impacts on human toxicity, acidification, aquatic ecotoxicity, and eutrophication.

Furthermore, the high yield of energy production from incineration is also supported by extant literature. Demetrious et al. (2018) in a similar comparative study, arrived at comparable findings. The authors conclude that incineration of residual waste had the highest electricity generation potential, followed by gasification. The landfill was the least technology for recovery energy. Qazi & Abushammala (2020) suggests that incineration is the most optimum technology for energy generation than gasification and anaerobic digestion. Segurado et al. (2019) indicate that direct combustion is the optimal technology for the maximum potential energy generation, followed by gasification and anaerobic digestion. Arafat et al. (2015) highlight that gasification is not the optimal technology for waste treatment, particularly when energy production is the ultimate consideration. Cleary and Caspersen (2015) also report that a wood pellet-fired power plant recovered higher energy than a CHP wood chip gasification plant. Higher energy production associated with incineration is due to the high energy efficiency of the internal combustion engine of the system (Demetrious et al., 2018; Cleary and Caspersen, 2015; Burnley et al., 2012; Arafat et al., 2015).

6.3 Sensitivity analysis and limitations of the study

A sensitivity analysis was carried out to identify the impact of the variations of the system parameters on environmental performance results. Parameters varied in the sensitivity analysis included: gasification system (overall system efficiency and carbon content of tar), incineration system (thermal efficiency and control efficiency for a pollutant abatement), and landfill system (methane gas generation, methane gas collection efficiency, and oxidation). Environmental impact results of the most sensitive to the varied parameters are summarized in Table 6.1.

Table 6.1 Sensitivity analysis of environmental impacts changes

	Gasification	Incineration	Landfill
GWP 100 (%)	±28.10	±15.41	±17.46
POCP (%)	±28.10	±15.00	±5.76
GWP, bio (%)	±28.57	±15.56	±50.97
GWP, total (%)	±28.23	±15.45	±21.11

The overall sensitivity results reveal that variations in all the parameters yielded lower/higher impact categories. For the gasification system, a variation of the baseline efficiency of 49% to 35% and carbon content of tar from 5% to 2% resulted in a decline of about 28% in the impact categories of GWP 100, POCP, GWP, bio, and GWP, total. Similarly, changes in thermal efficiency of the incineration system from 85% to 80% and a further reduction in control efficiency for a pollutant abatement by 5% produced savings of about 15% across the impact categories. In the landfill system, DOCF, methane gas collection efficiency, and oxidation factor parameters of 0.4, 67%, and 10%, respectively were changed to 0.57, 74%, and 36%. The sensitivity results generated an improvement in GWP100 (17%), POCP (6%), GWP, bio (51%) and GWP, total (21%). From the results, it can be concluded that the global warming impact categories were more sensitive to changes in the parameters.

Limitations relate to the validity and reliability of results and the conclusions drawn from the study (Simon & Goes, 2013). This study, like many other LCA studies, suffers inherent weakness. Data collection and modelling are within the bounds of the scope of a single case study. As such, the results of the study do not represent the precise situation in every municipality in Iceland. Like most waste management LCA literature, this study used an input-based functional unit of 1 Kg of waste wood treatment. Application of output-based functional unit was not appropriate due to the differences in the exergy properties of energy outputs of the analyzed systems (i.e., electricity vs. heat). Also, the study´s scope and data modeling assumptions limit a comprehensive assessment of environmental impacts of the assessed systems due to the exclusion of certain life cycle phases and processes. Moreover, lack of a full-scale uncertainty analysis, such as Monte Carlo simulations, restricts the generalizability of the study´s findings. Therefore, cautious interpretations and comparisons of the results with similar studies that used output-based functional units are required. Notwithstanding these limitations, the case study approach employed to explore the research questions offers the opportunity to understand, thoroughly, a particular phenomenon (such as environmental impacts of existing and alternative technologies) in a particular geographical context (Flyvbjerg, 2011, pp. 314). Such a study will provide an in-depth understanding of the phenomenon for policy decision-making as well as instigating further research into the phenomenon.

7 Conclusions

This study assessed for the first-time comparative life cycle GHG emissions and other environmental impacts of a small-scale CHP pallet gasifier with a SHI and methane-recovery landfill systems in the Icelandic context. Thus, this study provides an assessment of environmental impacts and energy recovery potentials of relatively two matured reference technologies in Iceland and compared them with an innovative small-scale gasifier that can be deployed in small remote communities and Island regions such as Westman Island.

The overall results suggest that gasification and landfill are the most environmentally sustainable options, outperforming incineration in almost all the impact indicators assessed. Analyzing individual impact categories, gasification provides the least environmental burdens in global warming, depletion of abiotic resources, acidification, eutrophication, terrestrial ecotoxicity, ozone depletion, photochemical oxidation, non-renewable (fossil), and renewable (water); whereas landfill is most favored in human toxicity, freshwater and marine aquatic ecotoxicities, non-renewable (fossil), non-renewable (biogenic), and renewable (geothermal, solar, wind).

Therefore, gasification is the most climate-friendly pathway to reduce the global warming impact. A further breakdown of global warming impact over different time horizons suggests that GHG emissions over a 20-year horizon are more potent than either a 100 or 500-year time horizon because of absorption of more energy per ton of GHGs over a shorter time than a longer time horizon. These impacts are attributed to the emissions of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). Biogenic GHGs emissions are the main sources of GWP, while the remaining fossil-based GHG emissions are largely driven by fossil fuel use in transportation and landfill operational activities. Concerning the most crucial life cycle stages' contribution to GWP impacts, over 95% of total global warming impacts of each system could be traced to the final combustion stages.

In addition to environmental impact assessment, energy recovery potentials were evaluated to compare the three technologies. The overall results point to the optimal energy production potential of the incineration technology. The higher energy production output of the incineration system is best explained by the high efficiency of the internal combustion engine of the system and reduced pretreatment stages that save energy consumption.

In summary, the study suggests gasification is an alternative system and the most environmentally sound pathway to dispose of wastes than landfill and incineration in Westman Island, and, perhaps, other small and remote communities in Iceland. In other words, the avoided high concentrations of dioxin emissions from incinerators and carbon emissions from waste transport to landfill sites make the gasification technology environmentally competitive than the incumbent systems available in the region.

The low environmental performance and high energy recovery potential of incineration underscore the trade-off between the technology choice for minimizing environmental impacts and generating more clean energy. No single technology is adequate to achieve these two goals at the same time. Future studies can extend the current work by assessing and comparing the environmental sustainability and energy production of integrated biomass technology pathways with the aims of achieving both environmental sustainability and higher cleaner energy generation.

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