Generation and storage of gas from waste decomposition in municipal solid waste landfills

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Abstract

Generation and storage of gas from waste decomposition in municipal solid waste landfills

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This Ph.D. thesis focuses on optimizing municipal solid waste flows and modeling and managing landfill gas generation from organic wastes. First, it presents a statistical survey of waste flow in New York and Montreal and a calculation of the energy recovery potential of food and yard waste in these cities. The results indicate a low diversion rate from landfills, with significant biogas generation potential from these wastes, contributing to around 2.5% of the energy supply in these cities. Second, it evaluates the current and proposed waste management systems in Montreal, applies a life cycle assessment using the IWM-2 software, and optimizes waste flows using a genetic algorithm to decrease energy consumption, greenhouse gas emissions and costs. The optimized waste flow considers 58% landfilling and shows the importance of further research on landfills.

The following chapters study the generation and storage of gas from waste decomposition in municipal solid waste landfills in the province of Quebec, Canada. The fifth chapter addresses the modeling scenarios of landfill gas generation based on a modified first-order decay model. It uses a genetic algorithm to independently fit parameters to methane and hydrogen sulfide generation models. The results show that differentiating more waste types improves the modeling accuracy, and the changes in waste management strategies within a landfill's decade-long lifetime require various modelling assumptions. Also, the work reveals the importance of considering how different landfill sectors are filled over time. The sixth chapter explores the potential of utilizing stored methane in landfills as an energy source. The study investigates the gas collection system shutdown and restart periods, determining the duration required to maximize collected stored methane. The results show that it takes 0.6 hours to start methane collection and 2.5 hours to reach the maximum collected stored methane. Additionally, the collected stored methane represents 10.5% of landfill gas flow.

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I would like to express my deepest appreciation to my dear supervisor, Professor Ursula Eicker, who supported me continuously during my Ph.D. I'm extremely grateful for her kindness, patience, open-mindedness and professionalism. I would also like to express my deepest gratitude to Dr. Juergen Schumacher. Although he, unfortunately, is not among us anymore, his kindness and encouragement always gave me the energy to continue and reach my goal. I had the pleasure of collaborating with members of the CERC (Canada Excellence Research Chair) at Concordia University, and I want to thank all my friends, especially Mr. Saeed Ranjbar.

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Special thanks to my lovely family: Jaleh, the best mother in the world for fulfilling me in love in each and every moment of my life, Iraj, my amazing father who gave me the power and braveness to move, and Sara and Tayebeh, my wonderful sisters and mentors. Finally, I could not have undertaken this journey without my beloved husband, Shahin Eskandari. I appreciate his continuous love and support, and our stunning baby, Aiden, for his love and joy.

Words cannot express my gratitude to my husband, Shahin, and I dedicate my Ph.D. thesis to him.

Contributions of the Author

Journal papers:

Energy recovery potential from food waste and yard waste in New York and Montreal

Authors	Tahereh Malmir and Ursula Eicker
Publisher	International Journal of Energy and Environmental Engineering (2020), vol. 14 (6), ISNI:0000000091950263, waset.org/Publication/10011244
Abstract	Landfilling of organic waste is still the predominant waste management method in the USA and Canada. Strategic plans for waste diversion from landfills are needed to increase material recovery and energy generation from waste. In this paper, we carried out a statistical survey on waste flow in the two cities New York and Montreal and estimated the energy recovery potential for each case. Data collection and analysis of the organic waste (food waste, yard waste, etc.), paper and cardboard, metal, glass, plastic, carton, textile, electronic products and other materials were done based on the reports published by the Department of Sanitation in New York and Service de l'Environnement in Montreal. In order to calculate the gas generation potential of organic waste, Buswell equation was used in which the molar mass of the elements was calculated based on their atomic weight and the amount of organic waste in New York and Montreal. Also, the higher and lower calorific value of the organic waste (solid base) and biogas (gas base) were calculated. According to the results, only 19% (598 kt) and 45% (415 kt) of New York and Montreal waste were diverted from landfills in 2017, respectively. The biogas generation potential of the generated food waste and yard waste amounted to 631 million m^3 in New York and 173 million m^3 in Montreal. The higher and lower calorific value of food waste were 3482 and 2792 GWh in New York and 441 and 354 GWh in Montreal, respectively. In case of yard waste, they were 816 and 681 GWh in New York and 636 and 531 GWh in Montreal, respectively. Considering the higher calorific value, this amount would mean a contribution of around 2.5% energy in these cities.
Chapters	Chapter 3

Improving municipal solid waste management strategies of Montreal (Canada) using life cycle assessment and optimization of technology options

Authors	Tahereh Malmir, Saeed Ranjbar, Ursula Eicker
Journal, Date, and DOI	Energies (2020), vol. 13, 5701, https://doi.org/10.3390/en13215701
Abstract	Landfilling of organic waste is still the predominant waste management method in Canada. Data collection and analysis of the waste were done for the case study city of Montreal in Canada. A life cycle assessment was carried out for the current and proposed waste management system using the IWM-2 software. Using life cycle assessment results, a non-dominated sorting genetic algorithm was used to optimize the waste flows. The optimization showed that the current recovery ratio of organic waste of 23% in 2017 could be increased to 100% recovery of food waste. Also, recycling could be doubled, and landfilling halved. The objective functions were minimizing the total energy consumption and CO_{2eq} emissions as well as the total cost in the waste management system. By using a three-objective optimization algorithm, the optimized waste flow for Montreal results in 2% of waste (14.7 kt) to anaerobic digestion (AD), 7% (66.3 kt) to compost, 32% (295 kt) to recycling, 1% (8.5 kt) to incineration, and 58% (543 kt) to landfill.
Chapters	Chapter 4

Optimization of landfill gas generation based on a modified first-order decay model: a case study in the province of Quebec, Canada

Authors	Tahereh Malmir, Daniel Lagos, Ursula Eicker
Journal, Date, and DOI	Environmental Systems Research (2023), vol. 12(6), https://doi.org/10.1186/s40068-023-00292-w
Abstract	Landfills will likely remain an essential part of integrated solid waste management systems in many developed and developing countries for the foreseeable future. Further improvements are required to model the generated gas from landfills. The literature has not addressed detailed waste characterization in landfill gas (LFG) modeling by a first-order decay model such as LandGEM while using a genetic algorithm. Additionally, little has been done in the literature regarding H_2S generation modeling. This paper uses a genetic algorithm to independently fit parameters to a CH_4 and H_2S generation model based on a modified first-order decay model. In the case of CH_4 generation modeling, biodegradable organic waste (OW) was segregated into food waste, yard waste, paper, and wood. In addition to optimizing the OW fractions, key modeling parameters of OW, such as CH_4 generation potential (L_0) and CH_4 decay rate (k_{CH_4}), were determined independently for different periods in the landfill's life. Similarly, in the case of H_2S generation modeling, the construction and demolition waste (CD) was classified into fines (FCD) and bulky materials (BCD), and H_2S generation potential (S_0) and H_2S decay rate (k_{H_2S}) of FCD and BCD were determined. LFG collection data from a landfill site in the province of Quebec, Canada, was used to validate the LFG generation model. A range of scenarios was analyzed using the validated model, including fourteen scenarios (two benchmark and twelve optimizing) for CH_4 and two for H_2S modeling. The results showed that the differentiation of more waste types improves the modeling accuracy for CH_4. Moreover, within the decade-long lifetime of a landfill, the waste management strategies change, requiring different assumptions for the modeling. Also, the work showed the importance of considering how different landfill sectors are filled over time. Finally, scenario twelve of optimizing scenarios, which assumed four waste types, constant three periodic waste fractions, and six sectors, had the lowest residual sum of squares (RSS) value. For H_2S generation modeling, both scenarios, with or without separate fits of S_0 and k_{H_2S} for FCD and BCD, predicted the generated H_2S well and had a very similar RSS value. Further data could improve H_2S generation modeling.
Chapters	Chapter 5

Analyzing landfill gas storage in integrating energy supply to photovoltaic power plants: A case study in the province of Quebec, Canada

Authors	Tahereh Malmir, Martin Heroux, Daniel Lagos, Ursula Eicker
Journal, Date, and DOI	Waste Management 171 (2023), 155–162, https://doi.org/10.1016/j.wasman.2023.08.029
Abstract	Landfills are extensively applied to dispose of municipal solid wastes in developed and developing countries. Landfill gas generation from biodegradable organic wastes can be collected and converted to energy. When the gas collection system is shutdown, some of this gas can accumulate and be stored inside the landfill. Using the gas storage capacity of the landfill gets a better management of the landfill site because the collected stored gas could transform the landfill into a cheap gas storage system to provide short-term energy and use the energy when needed. This novel study analyzes the stored methane using the gas collection data of a landfill in Quebec province, Canada, for modulating energy production from landfill gas. Twenty episodes of the gas collection system's shutdown and restart as well as different gas flow durations were studied. The results showed that the collected stored methane is accumulated in an average of 2.5 hours. Additionally, the collected stored methane represents 10.5% of landfill gas flow. Although the results are site-specific, the methodology of this paper can be used on other landfill sites with similar size and collection conditions. Designing new landfills could take into consideration some elements to enhance gas storage capacity. For instance, designing landfill daily covers with more granular materials and higher porosities can be the next step to enhance the landfill as a gas storage system during shutdowns.
Chapters	Chapter 6

Table of Contents

List of Figures

List of Tables

Abbreviations

AD	Anaerobic Digestion
BCD	Bulky materials of construction and demolition waste
C_{BD}	Collected CH_4 from B to D during T_{BD} (volume)
C_{DE}	Collected CH_4 from D to E during T_{DE} (volume)
CH_4	Methane
CO_2	Carbon dioxide
CO_{2eq}	Equivalent CO_2 of GHG emissions
$CO_{2eq.AD}$	Equivalent CO_2 of GHG emissions of anaerobic digestion
$CO_{2eq.C}$	Equivalent CO_2 of GHG emissions of composting
$CO_{2eq.Cbs}$	Equivalent CO_2 of GHG emissions of incineration
$CO_{2eq.L}$	Equivalent CO_2 of GHG emissions of landfilling
$CO_{2eq.R}$	Equivalent CO_2 of GHG emissions of recycling
CD	Construction and demolition waste
E	Energy consumption of each technology
E-wastes	Electronic products
E_{AD}	Energy consumption by anaerobic digestion
E_C	Energy consumption by composting
E_{Cbs}	Energy consumption by incineration
E_L	Energy consumption by landfilling
E_R	Energy consumption by recycling
F_{BD}	CH_4 flow from B to D (flow)
F_{BD_Avg}	Average CH_4 flow from B to D (flow)
F_{DE_Avg}	Average CH_4 flow from D to E (flow)
F_i	Fall of the year i (i = 1, 2, or 3)
FCD	Fines of construction and demolition waste
FDR	Fast decaying refuse
FW	Food Waste
GHG	Greenhouse Gas
H_2S	Hydrogen sulfide
HCV	Higher Calorific Value

k_{CH4}	CH_4 generation rate (*y^{-1}*)
k_{H2S}	H_2S generation rate (*y^{-1}*)
L_0	CH_4 generation potential (*m^3 CH_4/t biodegradable waste*)
LandGEM	Landfill gas emissions model
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCV	Lower Calorific Value
LFG	Landfill gas
MGP	Metal, Glass, Plastic and Carton
MSW	Municipal solid waste
NSGA	Non-dominated sorting genetic algorithm
OW	Organic waste
P_{BD_Avg}	Average atmospheric pressure from B to D (kPa)
P_{DE_Avg}	Average atmospheric pressure from D to E (kPa)
PC	Paper and Cardboard
PWF	Periodic waste fraction
RSS	Residual sum of squares
S_0	H_2S generation potential (*m^3 H_2S/t sulphur*)
S	Collected stored CH_4 (volume)
S_i	Summer of the year i (i = 1, 2, or 3)
SDR	Slow decaying refuse
T_A	Shutdown duration up to A (day)
T_{AB}	Start-up duration from A to B (day)
T_{BC}	Duration of collected stored CH_4 up to the peak from point B to C (day)
T_{BD}	Duration of the whole collected stored CH_4 from point B to D (day)
T_{CD}	Duration of collected stored CH_4 from point C to D (day)
T_{DE}	Duration of stable CH_4 flow from D to E (day)
V_{BD_Avg}	Average vacuum from B to D (pressure)
V_{DE_Avg}	Average vacuum from D to E (pressure)
W_i	Winter of the year i (i = 1, 2, or 3)
x_{AD}	Input waste to anaerobic digestion
x_C	Input waste to composting

x_{Cbs}	Input waste to incineration
x_L	Input waste to landfilling
x_R	Input waste to recycling

Chapter 1. Introduction

1.1 Problem statement

Global Municipal Solid Waste (MSW) generation was 2.01 billion tons in 2016 and will reach 2.59 and 3.40 billion tons by 2030 and 2050, respectively [1]. Organic Waste (OW), such as food waste, yard waste, paper, etc., contributes to a substantial portion of MSW. OW accounts for around one-third of the MSW in high-income countries and half in middle- and low-income countries [1].

Landfilling, anaerobic digestion, and composting are well-known OW management methods. OW management is a significant environmental challenge as its decomposition generates biogas, a Greenhouse Gas (GHG) consisting of more than 50% methane (CH_4). If it is managed improperly, which is a case in most developing countries, it can be a major contributor to global warming. The generated CH_4 is a potent GHG with a global warming potential of 28 times that of CO_2 over a 100-year timeframe [2]. In fact, today, more than 90% of waste in low-income countries is still openly dumped or burned, and gaseous emissions from such dumpsites have been poorly studied [3], threatening the environment and human health. In contrast, waste in developed countries is a resource for energy production, and only 2% is openly dumped in high-income countries [1].

Landfills are large and heterogeneous emitting sites that contribute 20% and 17.4% of national CH_4 emissions in Canada [4] and the U.S. [5], respectively. CH_4 leakage rates in landfills can be influenced by various factors, such as waste composition (specially OW which produces substantial amounts of CH_4), waste management practices (waste compaction, cover materials, landfill design, etc.), landfill age (more CH_4 generation by freshly deposited waste), climate (more CH_4 generation with temperature and moisture increase), and waste management infrastructure (e.g., landfill gas collection systems). A combination of these factors enhances the emissions from landfills. For instance, inefficient waste management practices, inadequate infrastructure, and the lack of proper regulations and monitoring contribute to the release of significant amounts of CH_4. Landfills will likely remain an essential part of integrated solid waste management systems in many developed and developing countries for the foreseeable future [6]. One of their advantages is being cheap. In addition, they can be operated and maintained easily. However, some challenges exist for landfill management, such as Landfill Gas (LFG) leakage and migration. For example in the Canadian province of Quebec, the regulation for landfilling and incineration of residual materials of the Environment Quality Act [7] ordains that landfills must follow the regulation respecting the gas collection system. It indicates that “in the case of landfills having a maximum capacity greater than 1,500,000 m^3 or as soon as a landfill receives 50,000 t or more of residual materials per year, the biogas collection system must have a gas pumping device except if such a device is not warranted because of the nature of the residual materials received and the low quantity of biogas likely to be produced.” (Chapter II – Landfills, Division 2 – Engineered Landfills, Subdivision 32 – Collection and Removal of Biogas).

First-order kinetic models are widely applied to forecast landfill CH_4 generation [8, 9]. For instance, Lagos and Heroux [8] segregated landfilled OW into two types and estimated CH_4 generation in a landfill containing MSW and construction and demolition wastes (CD) using a first-order kinetic model. Further waste segregation into easily, slowly, and hardly biodegradable OW can predict CH_4 generation more accurately. In addition, CD is often landfilled with MSW, producing H_2S, which has rarely been addressed in the literature. According to [10], sulfate-reducing bacteria in these wastes outcompete acetogens and methanogens for electron equivalents (like hydrogen or organic acids). The generated sulfide is inhibitory (reduces CH_4 generation) and leads to odor and corrosion [10].

A main drawback of landfills is occupying considerable land space. A landfill's lifetime is as long as decades, and occupying a vast land area is inappropriate environmentally and economically. Adding renewable energy plants to closed landfills can provide added value to the land use.

1.2 Motivation

Efficient waste management strategies should be based on three objectives: reducing cost, GHG emissions, and energy consumption. Cost is a significant criterion that leads municipalities and companies to choose the most economically feasible waste management option. Due to high costs, investing in modern waste management plants is usually not applicable in developing countries. Therefore, uncontrolled and illegal dumping sites exist in most cities of these countries. Consequently, excessive GHG emissions and lack of energy production dominate in these conditions.

In contrast, developed countries invest in modern waste management technologies to reduce GHG emissions and generate energy. However, some technologies cause high costs in many cases. For instance, Japan's waste management strategy relies on incineration mainly, a significantly expensive technology. In Tokyo, 2.7 million tons waste was generated in 2018 and 88% was incinerated, 2% was recovered and 11% was landfilled [11]. GHG emissions and energy consumption can be decreased by prioritizing waste management systems: waste prevention and source reduction, recycling and material recovery, composting and energy recovery from waste. However, the application of these waste management systems can vary depending on factors like local infrastructure, waste composition, and operational efficiency. In this regard, Life Cycle Assessment (LCA) is a guiding tool for selecting an optimum waste management method by evaluating each approach's potential environmental and economic impacts. LCA studies about waste management methods are applicable in developed and developing countries to help governments and organizations implement the optimum waste management plans.

Although current waste management plans and strategies try to reduce landfilling of OW, the situation has not been improved tremendously. Landfills are the end-of-life of several types of wastes, including organic or inorganic ones, in many developed and developing countries. Hence, it motivates landfill owners to extract energy from landfills. LFG can be collected and used as an energy resource, generating revenue and creating jobs. Additionally, using LFG reduces odors and hazards of LFG emissions, such as fire, temperature increase, and chemical and trip risks. LFG collection prevents CH_4 migration tremendously and decreases landfill contribution to global warming.

Different pathways exist for generated CH_4 in landfills. In the case of considerable CH_4 generation, LFG can be collected and converted to electricity, heat, or biomethane. A portion of CH_4 in the LFG emits to the atmosphere or oxidizes to CO_2 in the landfill cover (reaction of CH_4 with oxygen in the landfill cover soil which results in CO_2 generation). Landfill cover soil temperatures, soil moisture, and CH_4 soil gas concentrations are environmental factors can drastically affect the oxidation capacity [12]. IPCC [13] suggests an oxidation value of 10% for covered well-managed landfill sites to predict diffusion via the cap and escape by cracks or fissures. USEPA [14] assumed a single oxidation factor of 10% for managed anaerobic landfills from 1980 to 2004 and average oxidation factors of 14% (2005-2009), 18% (2010-2016), and 22% (2017-2022). Part of the LFG can migrate laterally from the landfill, and the rest could be stored temporarily inside the landfill. Lateral migration could account for 9-18% of the generated CH_4, where 57-79 kg h^{-1} CH_4 is generated from a landfill with an area of approximately 100,000 m^2 receiving around 2.9 million tonnes of mainly non-combustible waste and soil [15]. Well-engineered landfills are necessary to control emissions and leakages in order to effectively manage the waste by protecting the environment.

Understanding GHG sources and sinks is a significant endeavour, and many countries have committed to reducing their emissions (e.g., UN COP26 2021 [16]). Estimating the produced quantities of LFG and CH_4 can determine the efficiency of an LFG collection system through a mass balance of the LFG generated, collected, and emitted. Higher LFG collection efficiency, more energy recovery and fewer CH_4 emissions are sustainable ways to produce positive outcomes for local communities and the environment.

Additionally, as mentioned previously, landfills are widely used worldwide, and consuming substantial land space is one of their shortcomings. Thus, new renewable technologies, such as solar landfills, could be designed so that photovoltaic power plants and landfills provide stable energy supply. The combination of intermittent photovoltaics and storable landfill gas with cogeneration engines can deliver controllable power to the electrical grid.

Integrating renewable energy generation systems such as photovoltaic power plants on landfill sites can improve land space usage and decrease the costs associated with electrical energy consumption [17]. Several studies combined landfills with photovoltaic power generators ([17], [18]), but further investigation is required to consider a landfill as a financially viable gas storage system (i.e., cheap battery), and there is a limited amount of studies that address gas storage inside the landfill [15, 19].

1.3 Objectives

This research aims to improve sustainable waste management strategies. Using a case study in the Canadian province of Quebec as a basis, the presented studies and methods could be applied globally to enhance sustainability in developed and developing countries.

The first objective is to investigate the LCA of waste management methods to reduce cost, GHG emissions, and energy consumption. Next, this research estimates CH_4 generation from OW and CH_4 storage inside landfills to enhance the gas collection efficiency, storage and recovery, and decrease emissions. This research also predicts H_2S production from CD waste. The research questions are as follows:

> Q1. What is the optimum waste management strategy regarding life cycle cost, GHG emissions, and energy consumption? Does a well-managed landfill site better perform OW decomposition than anaerobic digestion or composting? These questions will be answered using a LCA with simplified waste decay models.
>
> Q2. How to improve the prediction of LFG generation models? Do OW and CD waste characterization improve estimating CH_4 and H_2S generation, respectively?
>
> Q3. How to assess the gas storage capacity of the landfill and the usage of the gas for a stable power supply in combination with other renewables?

The hypotheses are as follows:

H1. A LCA study could suggest an optimum waste management strategy. Due to the cost-effectiveness of landfills and the fact that energy can be recovered and GHG emissions can be controlled from landfills, they can be introduced as the main OW management method.

H2. CH_4 generation can be predicted more accurately by classifying the OW into different biodegradable categories (i.e., easily biodegradable, slowly biodegradable, and hardly biodegradable). Similarly, H_2S generation modeling improves if the CD waste is classified into fines and bulky materials.

H3. A landfill site can be used as a cheap gas storage system. The stored CH_4 could transform the landfill into an energy source to provide short-term energy to complement intermittent generation for example from photovoltaic power plants.

This work's outcomes can help municipalities, waste treatment operators, and policymakers world wide to deliver more sustainable waste management strategies by enhancing landfills' environmental and economic aspects.

1.4 Thesis organization

Figure 1.1 illustrates the structure of the thesis.

Chapter 1 is the introduction. This chapter explains the overall content of the thesis, including the problems, motivation, objectives, research questions, and hypotheses.

Chapter 2 contains the literature review. Organic waste valorization methods, life cycle assessment of waste management, landfill gas generation based on waste composition, description of gas collection systems, landfill gas collection efficiencies and factors affecting them, landfill gas measurement and modeling landfill gas generation are shortly addressed in this chapter.

Chapter 3 describes a statistical survey of waste generation in two case study cities, New York and Montreal. It presents the waste generation trends and flows and calculates energy recovery potential from food waste and yard waste in these cities.

Chapter 4 further develops the work done in Chapter 3 by adding a LCA study to determine the cost, GHG emissions, and energy consumption of waste management scenarios and optimize the waste flow specified for each waste management method.

Chapter 5 includes CH_4 and H_2S generation modeling from OW and CD for a case study in the province of Quebec, Canada, respectively. First-order decay models and parameter fitting are introduced in this chapter to optimize and enhance the existing models in twelve scenarios for CH_4 and two scenarios for H_2S generation modeling.

Chapter 6 assesses landfill gas storage and application regarding energy management for a case study in the province of Quebec, Canada. This chapter reports extensive gas collection data to analyze the CH_4 storage quantities and durations. It investigates the collected stored CH_4 of a municipal solid waste landfill for modulating energy production from LFG for combinations with photovoltaic power plants according to the energy demand.

Chapter 7 is the conclusion of the thesis.

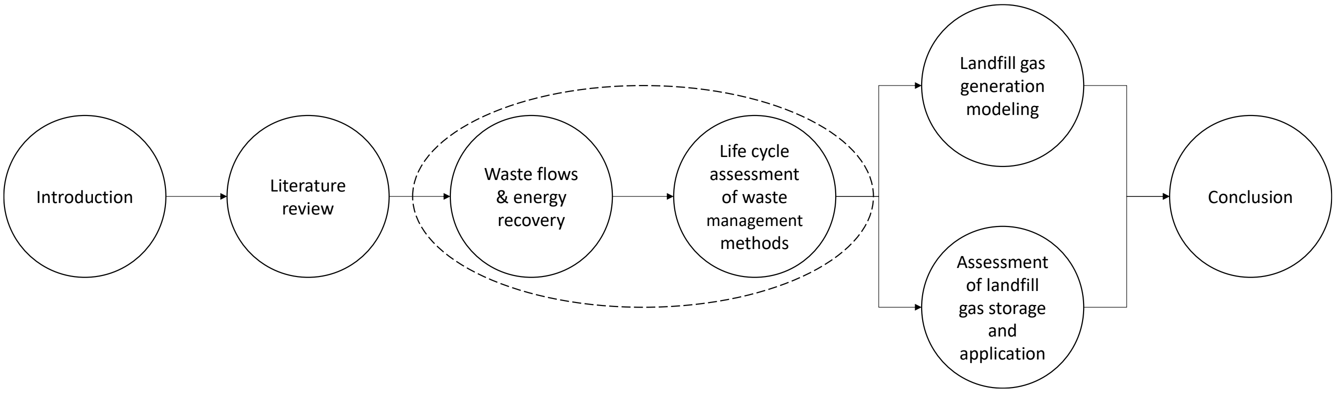


Figure 1.1: Structure of the thesis.

Chapter 2. Literature review

2.1 Organic waste valorization methods

The selection of a waste management method significantly depends on the waste type. Generally, the municipal solid waste contains organic and inorganic materials. Organic Waste (OW) can be biodegradable (e.g., Food Waste (FW)) and non-biodegradable (e.g., plastic). FW is of growing importance. It is the most considerable fraction in municipal solid waste, and its generation is predicted to increase with 44% by 2025 [20]. Hence, it provides a promising source for biorefinery feedstocks.

The combustion of such biomass waste is often energetically inefficient due to its high moisture content and low energy density [21]. According to the World Bank (2012) [22], the worldwide generation of municipal solid waste will increase 1.7 times by 2025 (increasing to 2.2 billion tons in 2025) and will be managed mainly by landfill disposal method. Beside landfilling, biological and thermo-chemical treatment methods could be widely used due to their applicability for biofuel to bioenergy conversion.

Anaerobic Digestion (AD) is a sustainable biological process that converts biodegradable organic matters to stabilized residues by producing biogas and digestate. The produced biogas can be used for heat, power, combined heat and power generation, and compressed natural gas application [23]. Thermo-chemical methods, including pyrolysis and gasification, are currently commercially used [24]. In gasification, the dried biomass is gasified or heated at extremely high temperatures (around 700 °C) in the presence of limited oxygen to produce synthesis gas or syngas (carbon monoxide and hydrogen), which is cleaned and can be used as an energy resource. The AD also generates digestate consisting of nutrients and stable organic carbon, which potentially represent valuable resources to be recovered [25]. In addition to the syngas, biochar is another useful product from the gasification process due to its potential ability to improve soil quality and sequestering carbon [26]. Utilizing these by-products as bio-fertilizer and soil conditioner would provide new ways of combining waste to energy strategies. AD is a useful energy source that can be accessed on-demand, unlike some other renewables such as wind and solar, which are more intermittent [27]. Not only in urban areas but also in remote areas such as farms that can be off the main energy/electricity grids, AD is useful to produce energy [28]. The critical point in applying AD is to use the installed capacity for energy production completely rather than flaring some of the biogas. As an example, 31% of medium-scale AD plants used biogas for energy production, and 69% of them burned the biogas [29]. AD is a promising biological treatment of biodegradable OW that is comprised of several stages, namely hydrolysis, acidogenesis, acetogenesis, and methanogenesis, and produces biogas and digestate. The caloric value of the biogas is about 5.5-6.0 $kWh.m^{-3}$ (corresponds to about 0.5 L of diesel oil) in which approximately 70% of CH_4 is produced by the degradation of acetic acid and about 30% by a redox reaction from H_2 (hydrogen) and CO_2 [30]. Feeds with a high solids content can be treated in batch digesters, continuously stirred tank reactors, and plug flow reactors. The temperature ranges for AD systems include thermophilic (50 °C to 60 °C), mesophilic (30 °C to 40 °C) and psychrophilic (15 °C to 25 °C) classification. Moreover, AD technology includes single-phase, two-phase, and temperature-phased AD. Two-phase systems can achieve an overall improvement of the anaerobic performances compared to the traditional single-phase process with enhanced H_2 and CH_4 production [31]. Also, the treatment of mixed organic matters can be done through anaerobic co-digestion. Studies of the co-digestion of FW generally found that the inclusion of FW was beneficial for CH_4 yield. In contrast, digestion processes with FW as the sole substrate were often found to be unstable [32]. Therefore, anaerobic co-digestion of two or more substrates has been widely studied recently to overcome the drawbacks of digesting single substrates [33]. For AD of the organic fraction of municipal solid waste, an average biogas yield of 100 m^3/t wet biowastes and a CH_4 content of about 60% by volume may be assumed [30]. However, various mathematical models have been advanced to estimate the biogas generation accurately. The most common reactor models are for continuous stirred tank and plug flow reactors.

A mathematical model of anaerobic bioconversion of complex substrates, including carbohydrates, lipids, and proteins, to biogas, was developed by Angelidaki [34]. This model involved hydrolysis of undissolved carbohydrates and undissolved proteins and eight bacterial groups. The bacterial groups were glucose-fermenting acidogens, lipolytic bacteria, long chain fatty acid-degrading acetogens, amino acid-degrading acidogens, propionate, butyrate, valerate-degrading acetogens, and aceticlastic methanogens. For the hydrolytic steps, first-order reaction rates were applied, and Monod type kinetics were used for all the bacterial steps. Monod type ammonia-N co-substrate dependency was also considered. Later, ADM1, as a structured model of AD, was developed to describe biochemical and physicochemical processes by implementing differential and algebraic equation sets [35, 36]. It is a mechanistic model and has been widely used for modeling and simulation of AD of different wastes. A continuous stirred tank reactor has been used as an example for both of these models. [37] integrated ADM1 with Aspen Plus and modified it to reflect ammonia inhibition of acertoclastic methanogenesis by adding an acetate oxidation pathway. [10] omitted composite in the model and modeled phosphorus, sulfur, and iron interactions. [38] investigated the microbial ecology and biological activity in the model. Few studies exist in the integration of this model.

Gasification is another waste-to-energy technology. A classification of solid fuels by their hydrogen/carbon and oxygen/carbon ratios indicates that carbon content plays an important role, and the higher the hydrogen/carbon or oxygen/carbon ratios, the less the heating value of the fuel. For a broad range of biomass, the hydrogen/carbon ratio might be expressed as $(H/C)=1.4125(O/C)+0.5004$ [39]. Gasification is a thermochemical conversion of carbon-based waste to gaseous fuel or chemical feedstock and can be modeling using computational fluid dynamics, heat and mass transfer, kinetics, and thermochemical equilibrium. The main steps of the gasification process are drying, pyrolysis, char gasification, and finally the combustion. The solid phase of gasification contains a carbon content higher than 76% [40]. The gaseous fuel, which is called syngas (mainly H_2 and CO), has a high ratio of H/C, as gasification strips carbon away from the waste and adds H_2. This is done by gasifying agents, including O_2, air, steam, and CO_2, which will rearrange the molecular structure of biomass to convert it to low-molecular-weight gases (like H_2 and CO) or liquids. Different agents have different impacts on the gas products such as changing the H_2 flow rates, CO content or the composition of the gas, heating value of the gas, etc. as well as the amount of tar in the char. For example, [41] found out that steam gasification was superior to the other options, due to capacity and clean by-product gases. However, another study mentioned the heating values for product gas based on air, steam, and O_2 as the gasifying agents are 4-7 MJ/Nm^3, 10-18 MJ/Nm^3, and 12-28 MJ/Nm^3, respectively [39]. The conversion of biomass forms various gaseous products in a gasifier. If the oxygen is used as the gasifying agent, oxygen-driven reactions will be dominant, and CO (low oxygen) and CO_2 (high oxygen) will be produced. And if the oxygen content exceeds a certain amount, gasification will move toward combustion and flue gas will be produced instead of fuel gas. A fluidized bed, moving (fixed) bed and entrained flow are three categories of gasifiers. Updraft and downdraft gasifiers are included in the fixed bed category. In both of them, biomass is fed from the top, but the syngas product leaves the gasifier at the top (updraft) or the bottom (downdraft) of the gasifier. Updraft gasifier is the oldest form of gasifiers, and it is still used. When biomass is fed into a gasifier, heated air is introduced into the bed of the gasifier and drying, pyrolysis, oxidation (combustion), and reduction (char gasification) occurs, which leads to emanating the gases upwards from solids and finally their escape from the gasifier [42]. In this kind of gasifier, the agent passes upward in a direction countercurrent to the flow of biomass. Non-fuel granular solids in fluidized bed gasifiers behave as a heat carrier and a mixer. In these systems, fuel is mixed with bed materials, which enhances the syngas production. The syngas characteristics relate to the gasification efficiency, which is expressed by syngas indicators. These indicators are cold gas efficiency, hot gas efficiency, and lower heating value [43]. In the case of cold gas efficiency, the elevated temperature of syngas that is leaving the system is not considered, and therefore, the syngas will be cooled down before entering the energy production system. In the case of hot gas efficiency, there would be no cooling down of the syngas, and the sensible heat of the gas is considered. Lastly, the lower heating value is the heat of waste combustion in case that the temperature of the syngas is not reduced to the room temperature. The thermochemical conversion of waste is affected by many parameters such as moisture content and composition of waste, temperature, and pressure of the gasifier, etc. The higher carbon content and lower moisture and ash contents of the biomass lead to the better efficiency of the system. The appropriate size of the pellets is significant, too, as the increase in size reduces the H_2 and CO of the product gas. Temperature plays an important role, as well. The reaction rate is sufficiently high, mainly at temperatures over 850 °C [44]. The maximum temperature and the maximum moisture content for moving bed, fluidized bed, and entrained flow gasifiers can be 600 °C, 950 °C and 1500 °C, and 50 wt%, 40 wt%, and 20 wt%, respectively [45]. In case of pressure, a pressure of 30 bar can be suitable for various chemical syntheses and integrated gasification combined cycle power generation, including CO_2 capture [46]. Yard waste can be an appropriate feedstock to the gasification system.

A summary of different waste management technologies is shown in Table 2.1.

2.2 Life cycle assessment of waste management

The increased population in urban areas leads to a significant raise in waste generation. Conventional waste management methods like landfill are amongst the main contributors to GHG emissions in the world. Landfills occupy large areas of lands and potentially pose a risk to human health and the surrounding environment. Therefore, it is crucial to find alternative methods for better municipal solid waste management in urban areas. Annually, 2.01 billion tons of waste are produced globally, and waste to energy technologies provide approximately 1.5% of the final energy consumption in Europe [47].

The EU Landfill Directive in 1999 prevented landfilling the OW and forced the members to reduce the quantity of biodegradable municipal waste sent to landfills to 75% (2006), 50% (2009), and then 35% (2016) compared to 1995 [48]. Based on this directive, the proportion of municipal waste disposed of by landfilling should be reduced to 10 % or less of the total amount of municipal waste generated by 2035 but most of the European countries could not achieve this target [49]. In 2018, 247 million tons of municipal solid waste (MSW) were treated in the EU by landfilling (23%), recycling (30%), composting (17%) and incineration (or combustion) with or without energy recovery (47%) [50].

Table 2.1: Summary of different waste management technologies [51].

Technology	By-product	Constraint	Energy generation
Incineration	Heat, ash	Ash production	By heat
Pyrolysis	Heat, biochar, bio-oil	High initial cost	By pyrolysis
Gasification	Heat, syngas, tar	High initial cost	By syngas
Anaerobic digestion	Biogas, digestate	High power cost due to aeration	By biogas
Composting	Heat, compost	Time-consuming	By heat

Life cycle assessment (LCA) is an appropriate tool to investigate the environmental impacts of waste management systems. LCA of wastes starts from the point that materials enter the waste stream, and it continues until the recycling, recovery, or disposal stage [52]. IWM-2 (integrated waste management II), WRATE (waste resources assessment tool for the environment), EASEWASTE (environmental assessment of solid waste systems and technologies), ORWARE (organic waste research), and WISARD (waste–integrated systems for evaluation of recovery and disposal) are examples of LCA software tools in waste treatment systems [53]. LCA tools evaluate different waste management scenarios environmentally, make helpful comparisons, and guide for selecting the best options effectively.

LCA evaluates the potential environmental impacts of products. IWM-2 is a life cycle inventory (LCI) model for integrated waste management that predicts the environmental burdens of a specific waste management system [54]. Numerous studies have focused on the LCA of diverse waste management systems. [47] compared recovery methods, which are beneficial compared to disposal options and concluded that thermal treatment and anaerobic digestion (AD) might be favorable over composting. Composting usually require large areas, is highly affected by weather condition and has odor problems. [55] criticized large-scale centralized composting due to enhanced environmental impacts and therefore, considered it a temporary solution. Decentralized waste management systems decrease transportation requirements significantly. [56] found out that the presence of a composting plant at 10 km from the municipality would decrease 65% of the environmental impacts due to the external transport. Connecting agriculture and waste is beneficial in terms of the reduction of GHG emissions. Still, there are some challenges regarding increased costs and acceptance for the use of digestate as a fertilizer (e.g., legal restrictions on the use of digestate produced from sewage sludge). [57, 58] reported that N_2O emissions of the application of liquid and solid portions of the digestate were the most significant contributors to global warming among all the life cycle stages. In another study, [59] coupled AD with composting to reduce the drawbacks associated with the direct soil distribution of anaerobic digestates such as the emission of CO_2 and obtained stable products to be safely used in soils without affecting their N- and P-fertilizing capacity. According to [60], one way to improve the utilization efficiency of biomass is the use of waste and production residues, and a vast majority of waste to energy technologies have lower GHG emissions when compared to fossil fuels.

Not only the long-term compost application on farmlands is beneficial in terms of nutrient supply, carbon sequestration, soil biodiversity, and soil workability, but also the overall global warming potential for composting varies between significant savings (-900 kg CO_2-equivalents/t wet weight) to net environmental impacts (300 kg CO_2-equivalents/t wet weight) [61]. [62] targeted AD, in-vessel composting, incineration, and landfill. AD was ranked as a system with the lowest environmental impacts, in-vessel composting as the least sustainable option environmentally, incineration as the most sustainable option per ton of waste treated and with the lowest life cycle costs (£71/t), and landfilling as the costliest option (£123/t). Another study assessed the sustainability of these scenarios for household FW treatment in 2030, and recommended the highest AD share that recovers both heat and electricity as the most sustainable option [63]. This study added that a modern landfill with energy recovery could have a lower global warming potential than composting [63]. Conducting an LCA of several waste management systems using an LCA tool like IWM-2 is important. IWM-2 evaluates the recycling of paper, plastics, glass, steel, and aluminum; composting and AD of paper, yard waste and FW; and combustion and landfilling of all waste components. Finally, it estimates the energy consumed (or produced) and the emissions to air, water, and land associated with different waste management systems.

2.3 Landfill gas generation based on waste composition

Human-related activities such as fossil fuel production, domestic livestock ranching/farming, manure management, rice cultivation, biomass burning, and waste management cause 60% of global CH_4 emissions compared to natural CH_4 emitters such as wetlands, termites, oceans, freshwater bodies, and wildfires [64]. The waste sector presents an appreciable potential for emissions reduction, particularly in developing countries where emissions from waste can account for 15% of total country GHG emissions due to the higher content of biodegradable OW [65]. If sanitary landfills are used and managed well, and LFG collection efficiency improves, emissions from landfills can be decreased tremendously.

Originating from waste decomposition, LFG mainly contains CH_4, CO_2, and trace amounts of H_2S as an inhibitory (reducing CH_4 generation), odorous and corrosive gas [10]. LFG production occurs in five phases. Aerobic condition in the first phase takes hours to weeks, the anoxic condition in the second phase takes 1-6 months, and the subsequent phases are anaerobic and take several months to years [66].

Anaerobic degradation of biodegradable OW (e.g., food waste (FW)) generates CH_4. Many studies have reported FW characteristics, and their values do not differ markedly from each other (e.g., 75.0% moisture, 10.0% ash, 40.0% carbon, and 2.0% nitrogen) [67].

There is as low as approximately 0.1% sulphur in MSW [68]. Sulphur-containing materials such as CD (bulky materials of construction and demolition (BCD) and fines of construction and demolition (FCD) wastes) can produce H_2S in an anaerobic environment. Compared to MSW, around 1.5-9.1% of CD is sulphate [69]. These wastes are landfilled with MSW in many cases. The by-product of CD processing facilities is screened materials termed FCD (soil and building material, including drywall). These fines are often used in MSW landfills as alternative daily cover [70] or final cover. Drywall comprises about 90% gypsum (calcium sulphate ($CaSO_4$)) and 10% paper on the back and front. According to [69], typical CD components could be concrete and mixed rubble (40-50%), wood (20-30%), drywall (5-15% or 5-30% [71]), asphalt roofing (1-10%), metals, bricks, and plastics (1-5% each).

2.4 Landfill gas collection systems

LFG can be directly discharged into the atmosphere or collected actively or passively using extraction wells (perforated or slotted collection pipes), blankets (sand, gravel, or geosynthetic blanket), or ventilated trenches within and around the perimeter of a landfill. LFG generation rates could increase by more than one order of magnitude due to bioreactor technology (up to around 2000 m^3/h) [66].

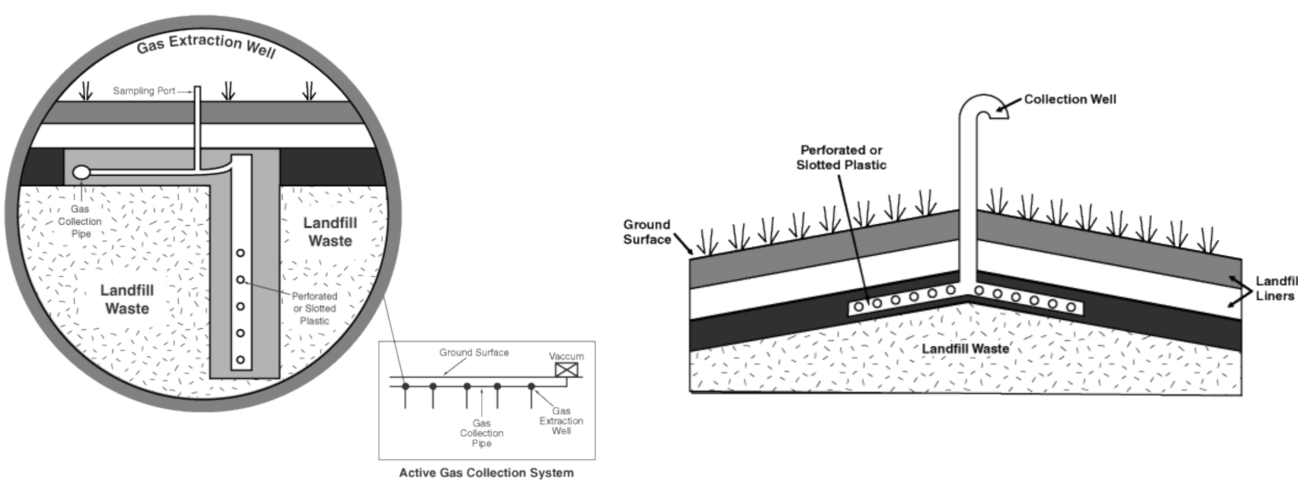
Figure 2.1 shows active and passive landfill gas collection systems. According to [66], active LFG collection systems use a blower system linked with a network of vertical LFG extraction wells and/or horizontal LFG collection trenches installed into the waste to suck on and collect the LFG. Then the collected LFG is transferred through a network of pipes to an extraction plant. Passive LFG collection systems offer a controlled method of permitting migrating LFG to emit from the soil without active mechanical systems and should only be installed and used in addition to an active LFG extraction system. Passive venting includes the installation of horizontal trenches filled with coarse granular fill/geocomposite and/or installing vertical augered wells equipped with riser pipes surrounded by gravel pack. Usually, they are located in the soil surrounding the landfill close to the edge of the waste.

2.5 Landfill gas collection efficiencies

Several studies used different methods to calculate the gas collection efficiency of landfills, which is the ratio of collected (or recovered) to generated gas [72]. Historically, gas collection efficiencies have been typically estimated to be 50-75%, based on measured gas extraction rates divided by modeled gas generation rates. However, actual gas collection efficiencies depend on the quantification of all the pathways in the CH_4 mass balance (Eq. 2.1, all units in mass time^{-1}) [19]. In this regard, [19] assessed the overall CH_4 mass balance in field cells with various designs, cover materials, and gas management strategies and obtained 35-90% collection efficiency.

$$CH_4\ generation = CH_4\ emitted + CH_4\ oxidized + CH_4\ recovered + CH_4\ migrated + \Delta CH_4\ storage \qquad \textbf{Eq. 2.1}$$

Air dispersion models (e.g., AERMOD [73]) use field measurement of CH_4 concentration data and meteorology data to calculate CH_4 emissions from the landfill. A study [74] developed a measure of collection efficiency using readily acquired surface CH_4 concentrations and the U.S. EPA's Industrial Source Complex (ISC, later replaced with the AERMOD) model and showed that the surface CH_4 concentration (ppm) from a landfill is directly proportional to the collected CH_4 flow (m^3/h). The study used the ISC model to predict the landfill surface CH_4 concentration reductions achieved by LFG collection. Considering the meteorological data, landfill cover oxidation effects, etc., the results indicated an efficiency approaching 95% or more.



(a) (b)

Figure 2.1: Landfill gas collection systems: (a) Active, and (b) Passive[1].

[1] Reference: https://www.atsdr.cdc.gov/hac/landfill/html/ch5.html (accessed 29 March 29, 2022)

Fecil [75] validated a first-order decay model for CH_4 generation with annually collected gas data and obtained 98% gas collection efficiency using a CH_4 mass balance. The CH_4 mass balance considered quantities of CH_4 collected, oxidized in the final cover, and emitted to the atmosphere. The study aimed at establishing a relation between CH_4 concentration on the ground and CH_4 flow. It used two methods to estimate CH_4 concentration on the ground: 1) Instantaneous Surface Monitoring (ISM) method, a portable Flame Ionization Detector (FID) to instantaneously measure the CH_4 concentration at a landfill surface divided into grids (wind less than 4.5 m/s and at 5-7 cm above the ground), and 2) a funnel as a static flux chamber coupled with FID. To predict the CH_4 flow, it applied a dynamic flux chamber coupled with FID (wind less than 4.5 m/s). They found out that the CH_4 concentrations on the ground measured by the ISM method were correlated with CH_4 flow. The correlation factor was $0.3647\ \mu g/(m^2.ppmv.s)$.

2.6 Landfill gas measurement

Based on chapter Q-2, r. 19, Regulation respecting the landfilling and incineration of residual materials of Environment Quality Act [7], landfills in Québec must follow the regulation respecting the gas collection system. Chapter II – Landfills, Division 2 – Engineered Landfills, Subdivision 32 – Collection and Removal of Biogas indicates that "in the case of landfills having a maximum capacity greater than 1,500,000 m^3 or as soon as a landfill receives 50,000 t or more of residual materials per year, the biogas collection system must have a gas pumping device except if such a device is not warranted because of the nature of the residual materials received and the low quantity of biogas likely to be produced."

Various LFG measurement techniques for quantifying CH_4 emissions from landfills measure against different scales, i.e., from the landfill surface to several kilometers away. These methods are surface flux chambers, ISM, eddy covariance, radial plume mapping, tracer gas dispersion, differential absorption LiDAR, and aerial mass balance (Unmanned Aerial Vehicles (UAV) in a red rectangular) (Figure 2.2). Most of them lack the complete surface coverage of landfills and miss localized hotspots (e.g., flux chambers do only point measurements). Accurate, whole-site CH_4 emission quantifications are best done using methods measuring downwind of the landfill, such as tracer gas dispersion and differential absorption LiDAR (instruments include the Fourier Transform infrared spectroscopy and cavity ring-down spectroscopy) [2]. However, UAVs bring a cheaper solution with the site's comprehensive coverage than these methods. For instance, CH_4 emissions were quantified to be around 156 to 592 m^3/h by flux chamber, or 178 to 432 m^3/h by tracer gas dispersion method [2].

2.7 Modelling landfill gas generation

Life cycle assessment tools provide coefficients to model the energy recovery from OW straightforwardly. For instance, [76] calculated the produced biogas' energy content from anaerobic digestion by multiplying the FW's biochemical CH_4 potential (450 m^3CH_4/t volatile solids) with the quantity of volatile solids fed to the reactor. They also assumed that the conversion factor of 1 m^3 CH_4 equals 10 kWh of electricity and used the energy conversion efficiencies of 38% for electricity and 48% for heat to convert biogas into heat and electricity in the CHP unit. [77] considered 120 m^3/t LFG potential for fresh matters. Using EASEWASTE Life cycle assessment tool, [78] and [79] assumed 2%, 8%, 70%, and 16% generated gas and 0%, 75%, 75%, and 0% [78], and 0%, 90%, 90%, and 0% [79] collected gas from a conventional landfill with energy recovery after 2, 3, 35, and 60 years, respectively. These various percentages represent typical or possible gas generation and collection measures to show a different level of environmental protection. [80] took into account the LFG collection efficiency to be

50% and in-situ burning in flares to be 25%. They estimated the collected LFG's energy content to be 3.12E+09 MJ (biogas lower heating value of 17.73 MJ/m^3) and 2.43E+08 kWh electricity production (1.38 kWh electricity per m^3 biogas), burning the LFG in turbines with 28% efficiency. [81] modelled LFG generation by EASETECH Life cycle assessment tool using the first-order decay model in which reactive concentration during a period depends on its concentration at the beginning of the reaction and degradation rate. Compared to no-treatment, GHG emissions reductions were as high as 50%-76% with LFG flaring's sole inclusion.

Moreover, the Buswell equation applies to the anaerobic digestion process. Buswell and Mueller [82] discussed the chemical mechanisms that may produce CH_4 by bacteria in nature via anaerobic fermentation. They developed an empirical equation to calculate the substrate's complete conversion to CH_4 and CO_2 to get 95-100% yields [82]. Landfill technology is a type of digester [83]. A landfill could be inside the same waste management complex as the digester, similar to real situations [84]. Therefore, the Buswell equation can also calculate the gas generation potential from landfills. [85] used the Buswell equation to achieve carbon and nitrogen mass balances of biodegradable solids conversion in a bioreactor landfill. They concluded that 45% of the carbon and nitrogen was transferred into the liquid and gas phases within two years. [86] estimated the biogas composition of anaerobic digestion by the Buswell equation. [87] used the Buswell equation to obtain the substrates' theoretical CH_4 potential.

The polynomial regression models explain the relationship between the cumulative biogas yields as a function of anaerobic digestion time through OW such as FW. The biogas production potential is determined by fitting the experimental data based on the cumulative biogas production and the kinetic parameters of the models employed to analyze the FW degradation rate [88]. Several studies applied the kinetic models, including the Cone model [89], the first-order kinetic model [88-95], and the Modified Gompertz model [88, 89, 91-95]. However, various studies' kinetic parameters (e.g., the maximum biomethane production potential of the substrate, hydrolysis rate constant, maximum specific CH_4 production rate, and lag phase time) vary significantly. These differences make them case-specific rather than general and limit their application in other cases, leading to the need for a general model.

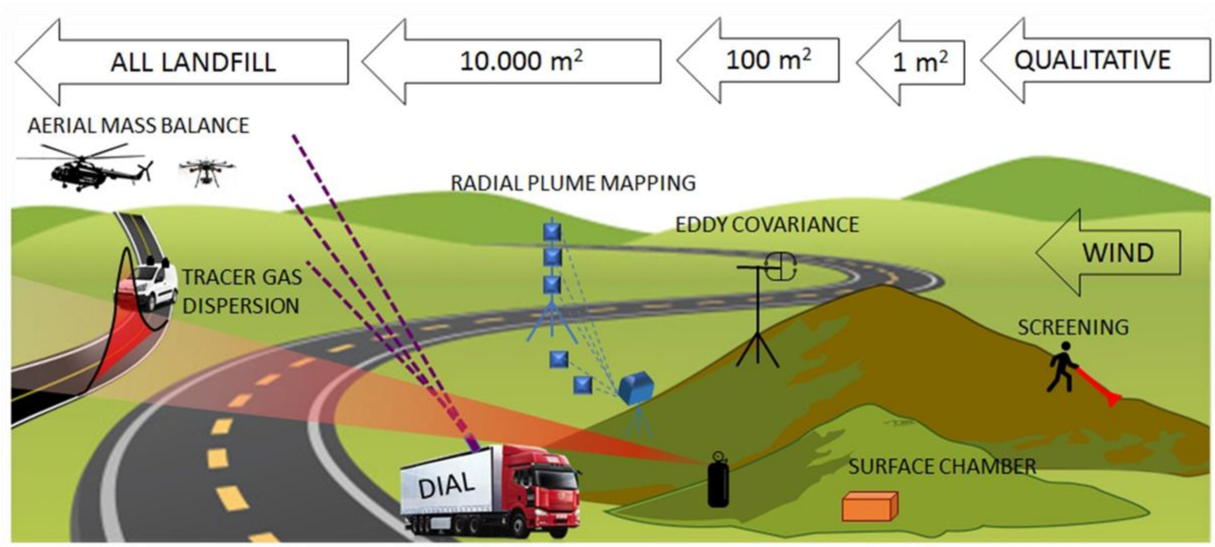


Figure 2.2: Overview of methods to identify and quantify CH_4 emissions from landfills [2].

[34] developed a mathematical model of anaerobic bioconversion of complex substrates, including carbohydrates, lipids, and proteins, to biogas. This model involved hydrolysis of undissolved carbohydrates and undissolved proteins, and eight bacterial groups. The bacterial groups were glucose-fermenting acidogens, lipolytic bacteria, long-chain fatty acid-degrading acetogens, amino acid-degrading acidogens, propionate, butyrate, valerate-degrading acetogens, and aceticlastic methanogens. First-order reaction rates were applied for the hydrolytic steps, and Monod-type kinetics were used for all the bacterial steps. Monod type ammonia-nitrogen co-substrate dependency was also considered. Later, ADM1, as a structured model, was developed to describe biochemical and physicochemical processes by implementing differential and algebraic equation sets [35, 36]. It is a mechanistic model and has been widely used to model and simulate various wastes' anaerobic digestion [38]. A continuous stirred tank reactor has been used as an example for both of these models. [37] integrated ADM1 with Aspen Plus and modified it to reflect ammonia inhibition of acetoclastic methanogenesis by adding an acetate oxidation pathway. [10] omitted composite in the model and modelled phosphorus, sulphur, and iron interactions. [38] investigated the microbial ecology and biological activity in the model.

The first-order kinetic model (e.g., Landfill Gas Emissions Model (LandGEM)) is the most widely applied model to forecast landfill CH_4 generation [8, 9]. The U.S. EPA [96] developed LandGEM, which considers CH_4 generation potential, L_0 (m^3/t wet waste), and CH_4 generation rate associated with waste decomposition, k_{CH4} (y^{-1}) (Eq. 2. 2).

$$Q_{CH_4} = \sum_{i=1}^{n} \sum_{j=0.1}^{1} k_{CH_4} L_0 \left(\frac{OW_i}{10}\right) e^{-k_{CH_4} t_{i,j}} \qquad \textbf{Eq. 2. 2}$$

where Q_{CH_4} (*m^3/y*) is the annual CH_4 generation after *n* years, L_0 (*m^3 CH_4/t biodegradable waste*) is CH_4 generation potential from biodegradable waste, k_{CH_4} (y^{-1}) is the CH_4 generation rate, OW_i (*t*) is the quantity of biodegradable OW landfilled in year *i*, $t_{i,j}$ is the age of the j^{th} section of landfilled OW at the i^{th} year, j = 0.1 year time increment and *n* is the number of years calculated (year of calculation - initial year of waste acceptance).

Although LandGEM models CH_4 generation from heterogeneous wastes effectively, some inputs modifications could significantly enhance its accuracy. For instance, [97] concluded that the LandGEM overestimated CH_4 generation. One reason could be that LandGEM only includes one type of waste. To overcome this deficiency, [8] reformulated the model by involving two types of biodegradable organic wastes (fast decaying refuse (FDR) and slow decaying refuse (SDR)) and optimized the OW fractions and key modelling parameters (L_0 and k_{CH4}) independently for periods in the life of a landfill. A study by [98] categorized the biodegradable OW into three types: easily (e.g., FW), slowly (e.g., paper), and hardly (e.g., wood, textiles, and leather) biodegradable wastes. IPCC [13, 99] further divided the biodegradable OW into rapidly degrading waste (FW, sewage sludge), moderately degrading waste (other (non-food) organic putrescible, garden and park waste), slowly degrading waste (paper/textile and wood/straw), and bulk waste. Besides, LandGEM does not address the interactions of different waste components, interfering with the biogas generation rate. Although sulfate-reducing bacteria help to maintain pH within a reasonable range for methanogenesis (under neutral pH conditions, not acidic ones [100]), methanogens and sulfate-reducing bacteria compete for common substrates (i.e., hydrogen and acetate) to generate CH_4 and H_2S, respectively. According to [10], sulfate-reducing bacteria outcompete acetogens and methanogens for electron equivalents (e.g., hydrogen or organic acids) leading to sulfide production which is inhibitory and causes odor and corrosion. [8] modelled CH_4 generation in a landfill, containing MSW and CD, yet neglected the H_2S generation effect from CD. However, low concentrations of H_2S

could act as an inhibitor on the growth of microorganisms and suppress the CH_4 forming processes in the presence of sulfate and sulfate-reducing conditions effectively [100]. In addition to CH_4, H_2S generation is best modelled with a first-order decay equation, similar to LandGEM [70]. Therefore, a first-order kinetic model, similar to LandGEM, could evaluate H_2S generation from sulphur-containing wastes: BCD and FCD. An optimization algorithm such as a genetic algorithm could also be implemented to estimate H_2S generation potential, S_0 (m^3/t sulphur), H_2S generation rate, k_{H2S} (y^{-1}), and sulphur content, *SC* (%), of BCD and FCD.

[101] used the LandGEM model to calculate landfill emissions and CH_4 generation. The model was a first-order decay model in which CH_4 generation depended on the CH_4 generation constant, CH_4 generation potential, and the mass and age of waste. They concluded that for 146,000 t MSW landfilled, CH_4 generation equaled 17,948 t, and CO_2 emission reached 49,246 t. Another study [102] applied the LandGem model and achieved higher LFG generation with higher CH_4 generation potential. Their results confirmed the dependence of landfill impacts on waste composition, in particular, on the amount of biodegradable OW. [103] compared the LandGEM model results with other models and proved this model's ability to estimate GHG emissions from MSW landfills.

Chapter 3. Energy recovery potential from food waste and yard waste in New York City and Montreal

3.1 Introduction

Looking through the current practices and opportunities of megacities in developed countries, sustainable waste management is still a challenge. The world's largest 27 megacities contribute to 12.6% of waste production of the total global quantity [104] and landfilling is still the predominant waste management method. Efforts have been done globally in order to reduce landfilling of biodegradable fraction of waste, but the reduction amounts were not satisfying so far.

Food Waste (FW) is an easily biodegradable Organic Waste (OW) [98]. It contributes almost half of the total municipal wastes in most countries [105] and has a great potential to be used for energy purposes. However, it is directly landfilled in many cases. For instance, landfilling of over 97% of FW in the U.S. was reported in 2010 [106] and the situation did not improve significantly later. In 2014 and 2015, FW accounted for 38 [107] and 39 million tons [108] in the U.S. respectively and three quarters of these amounts were landfilled [107, 108]. In the last years, different strategic plans for waste diversion from landfills were developed to increase energy generation and material recovery from waste.

In New York City as the most populous and the most densely populated city in the U.S. with around 8.5 million inhabitants [109], it is planned, by 2030, to achieve 75% diversion of solid waste from landfills [110] and a 90% reduction in total waste disposed in landfills relative to 2005 [111].

A second case study is the agglomeration of Montreal (Montreal), which is made up of 16 cities including the City of Montreal, which in turn is divided into 19 boroughs. The City of Montreal is the largest city in the Canadian province of Quebec (24% of the population) [112] and the second-most populous municipality in Canada with around 2 million inhabitants [113, 114]. Currently most of the OW in Quebec is landfilled or incinerated and it is planned to ban the disposal of OW and reach 60% diversion from landfill [115, 116]. Moreover, Montreal has a Waste Management Master Plan firmly anchored in the targets of the Quebec Residual Materials Management Policy - 2011-2015 Action Plan of the Government of Quebec. According to this plan, the recovery target for recyclables, OW, and construction and demolition (CD) waste is 70%, 60% and 70%, respectively [113]. It is also planned, by 2030, to increase the bioenergy production by 50% through various methods such as bio-methanization of OW [117].

Currently most waste in New York City is disposed in landfills [118, 119] and mishandling of OW in Montreal or Quebec has been reported by several studies [112, 116, 120]. In this paper we carry out a statistical survey on waste flow in New York City and Montreal and theoretically estimate the energy recovery potential from FW and yard waste for each case.

3.2 Methodology

3.2.1 Data collection and analysis

Figure 3.1 shows a typical waste flow. As can be seen, generated waste divides to two categories of recyclables and non-recyclables. Non-recyclables are disposed directly into landfills or used for energy recovery (e.g., by incineration). Recyclables which are comprised of OW (FW, yard waste, etc.), paper and cardboard (PC), metal, glass, plastic and carton (MGP), textile and electronic products (E-waste) and other materials have two routes. One is the same as non-recyclables, and the other route is being used for energy and material recovery, or in case of OW, being composted or recovered for energy (e.g., by anaerobic digestion, gasification, etc.).

Data collection and analysis for the waste flow of New York City and Montreal were mainly done based on the reports published by the Department of Sanitation in New York City [118] and Service de l'Environnement in Montreal [113, 121].

3.2.2 Case study of New York City

According to DSNY [118], the report characterized waste collections from residential properties of all sizes and styles, and a small number of institutional and agency costumers. The residential characterization included four residential curbside collection streams including OW (FW, yard waste and food-soiled paper), PC, MGP and refuse (non-recyclables in Figure 3.1). Samples were selected randomly from trucks identified by DSNY collection route and tonnage data to reach a 90% confidence of statistical significance. 45 kg (100 lb) of material per sample for OW, PC and MGP, and 91 kg (200 lb) per sample for refuse were collected for sampling. Totally 660 samples were collected (79 OW, 148 PC, 187 MGP and 246 refuse). Samples were hand sorted by the study team into 70 main sort categories and all of them were fully sorted into the same set of categories.

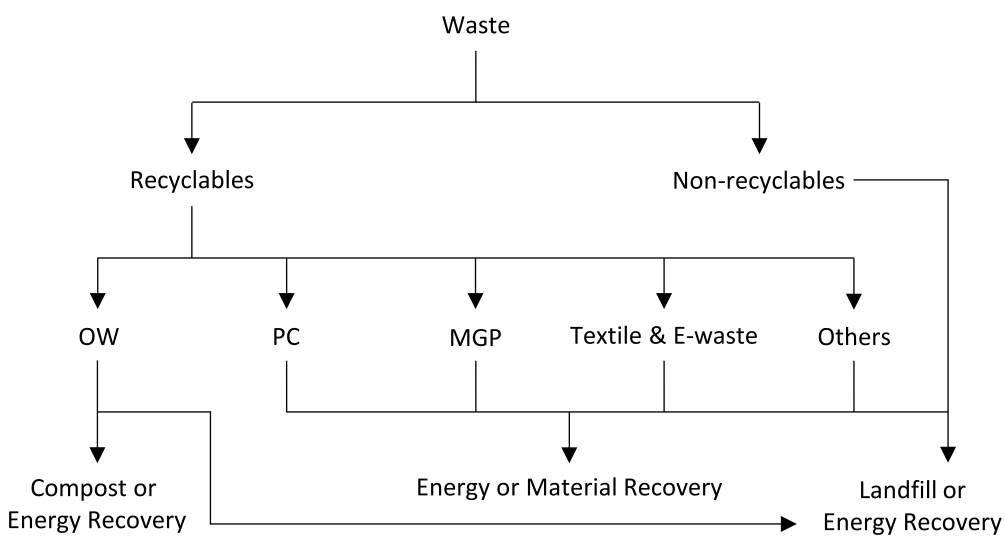


Figure 3.1: Typical waste flow.

3.2.3 *Case study of Montreal*

According to the Service de l'Environnement in Montreal [113], the report characterized waste collections from three sources including residential, institutional and commercial ones. The generated waste included OW (FW, yard waste and mixed residue), PC, MGP, residential CD, harmful household products (e.g. paint, pesticide, mercury devices, etc.), textile, e-waste, and household waste. Collection and disposal of waste is handled by the municipalities in Montreal in different ways and separation of materials is done in sorting centers. Curbside collection service collects the household waste and recyclables and partially OW. OW consisting of FW and yard waste is collected in most of the buildings of 8 or less dwellings in Montreal and then transformed into compost. Seven ecocenters in Montreal collect CD, wood, metal, tire, polystyrene and textile, harmful household products, e-waste, yard waste (gardening and weeding residues, leaves and Christmas trees) and other reusable materials. CD is also collected on street or as a result of resident calls. Household waste and non-recyclable CD are sent to the landfills.

3.2.4 *Waste to energy calculation method*

The biogas generation potential from biodegradable OW depends on the composition of the OW which is characterized by proximate and ultimate analyses. Proximate analysis determines the total solid, volatile solid, pH, volatile fatty acids, and soluble chemical oxygen demand of OW. Ultimate analysis determines the chemicals, such as carbon, hydrogen, oxygen, nitrogen, and sulfur. The characteristics of FW have been reported in many studies, and their values do not differ markedly from each other [67]. Zhang and Matsuto assumed 75.0% moisture, 10.0% ash, 40.0% C and 2.0% N in FW [67]. Sharma reported 44.2% C and 49.8% O in raw yard waste [122]. As stated before, OW in New York City is comprised of FW, yard waste and food-soiled paper and in Montreal is made of FW, yard waste and mixed residue. Samples of OW were not obtained in this study, and their characteristics were assumed as 42.7% C, 9.1% H, 1.97% N and 46.2% O in FW and 46.2% C, 5.8% H, 1.03% N and 47.0% O in yard waste with no uncertainty [123]. The percentage of sulphur, S, was assumed to be zero. Moreover, the moisture content of FW and yard waste was assumed to be 75% [67] and 63% [124] respectively.

In order to calculate the gas generation potential of OW, the Buswell equation [82] was used in which the molar mass of the elements (C, H, N and O) was calculated based on their atomic weight and the amount of FW and yard waste in New York City and Montreal (Eq. 3.1).

$$C_nH_aO_bN_c + \left(n - \frac{a}{4} - \frac{b}{2} + \frac{7c}{4}\right)H_2O \rightarrow \left(\frac{n}{2} + \frac{a}{8} - \frac{b}{4} - \frac{3c}{8}\right)CH_4 + \left(\frac{n}{2} - \frac{a}{8} + \frac{b}{4} - \frac{5c}{8}\right)CO_2 + cNH_4^+ + cHCO_3^- \qquad \textbf{Eq. 3.1}$$

The higher calorific value (HCV) and lower calorific value (LCV) of the FW and yard waste (solid base) and biogas (gas base) were calculated using Eq. 3.2 and Eq. 3.3 [24] and Eq. 3.4 and Eq. 3.5 [125].

$$HCV\ (MJ/kg\ TS) = (34.1 \times C + 102 \times H + 6.3 \times N + 19.1 \times S - 9.85 \times O)/100 \qquad \textbf{Eq. 3.2}$$

$$LCV\ (MJ/kg\ TS) = high\ heating\ value - 2.454 \times (W + 9H) \qquad \textbf{Eq. 3.3}$$

where C, H, N, O and S refer to carbon, hydrogen, nitrogen, oxygen and sulphur content (%TS) in the feedstocks, respectively, and W represents the moisture in fuel (wt.%).

$$HCV\left(\frac{MJ}{m^3}biogas\right) = 0.3989 \times methane\ content\ (\%) + 0.0213 \quad \textbf{Eq. 3.4}$$

$$LCV\left(\frac{MJ}{m^3}biogas\right) = 0.3593 \times methane\ content\ (\%) + 0.0192 \quad \textbf{Eq. 3.5}$$

All simulation models are implemented as modular blocks in the integrated simulation environment Insel4D, which is under development at Concordia University (www.insel.eu). The goal is to develop scenarios, where biogas generation from OW is integrated into the urban energy system via a gas network or cogeneration strategy.

3.3 Results and discussion

3.3.1 Trend in waste generation

Figure 3.2 illustrates the generated waste in New York City (2005 to 2017) and Montreal (2012 to 2017). In New York City, total waste generation was 3,418 kilotons (kt) in 2005 and around 3,095 kt in 2013 and 2017 which may be due to changes in consumption patterns, such as the decline in print newspaper sales, and to the evolution in product design to favor more lightweight packaging. MGP and PC had the highest amount in 2005 (1,213 kt) but OW dominated from 2013 (976 kt) to 2017 (1,062 kt). Non-bottle rigid plastics were added to the MGP recycling program in 2017 which led to an increase in their recovery rate. Glass packaging was declined as it was replaced by lighter weight plastic options. Less printed matter and more online shopping accounted for the decline and grow in recyclable PC respectively. Moreover, small quantities of cartons and aseptic boxes were misplaced which caused recycling of 8% of cartons with paper instead of with MGP. CD was 178 kt in 2005 and decreased to 138 kt in 2017. Textile, e-waste & harmful household product increased from 196 kt in 2005 to 218 kt in 2017.

In Montreal, total waste generation decreased from 970 kt in 2012 to 931 kt in 2017. The average amount of OW, MGP and PC, CD, and textile, e-waste & harmful household product was 361 kt, 286 kt, 234 kt and 8 kt, respectively. “Various factors affected the decrease in waste quantities such as replacement of printed newspapers by digital editions, eco-design of products which reduces the weight of containers, reduction of over-packaging and reduction of consumption.”

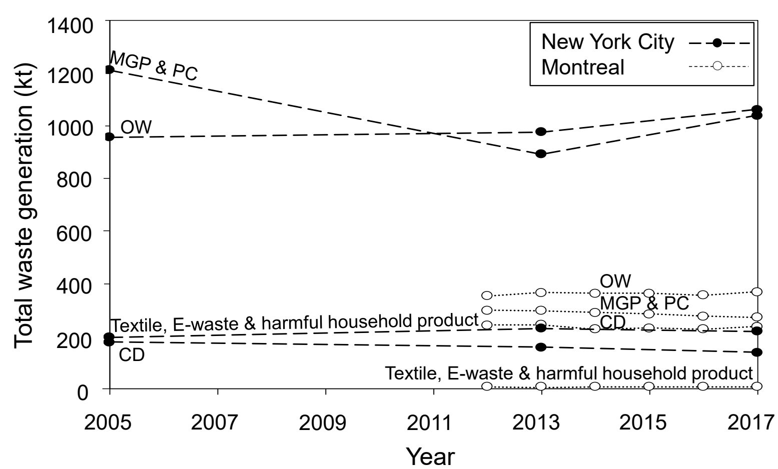


Figure 3.2: Total waste generation in New York City [118] and Montreal [113, 121].

Many products that used to be made from recyclable materials changed to multi-layered flexible packaging that are not accepted for recycling in New York City and Montreal. For example, rigid plastics are designated as recyclable in New York City, but film, flexible or foam plastics are not. In Montreal plastic #6 (polystyrene), different kinds of plastic bags and films are not considered as recyclable items. Worldwide efforts have been done to develop the public policies on plastic carrier bags [126]. Introducing the degradable plastics as the environmentally friendly alternatives to the market can decrease the huge amounts of plastics that are landfilled. For instance, Malmir used solvent casting method to prepare biodegradable films of poly (3-hydroxybutyrate-co-3-hydroxyvalerate) with cellulose nanocrystals which has capability for applications in the industry of food packaging [127] and achieved well-dispersed bionanocomposites with improved mechanical and barrier properties [128].

3.3.2 Waste flow

The waste flow of New York City and Montreal in 2017 is shown in Figure 3.3. Based on this figure, from a total of 3,090 kt generated waste in New York City (comprising 77% recyclables including OW, PC, MGP, textile, plastic shopping bags, harmful household product and e-waste, and 23% non-recyclables including CD and other materials) only 19% was diverted and 81% was mainly landfilled. OW with one third contribution to the total generated waste accounted for 1,062 kt from which approximately 13 kt was recovered and the vast majority was landfilled. Curbside OW collection to collect source separated food scraps, yard waste and food-soiled paper was planned to be introduced to all the neighborhoods in New York City, but it could not reach this target [129]. Plastic films and foam made up 7.5% of the waste stream including garbage and recycling bags (2.5%) and smaller plastic shopping bags (1.9%). Contractors or fee-for-service workers are responsible to dispose their commercial CD; however generated CD from do-it-yourself projects can be disposed in DSNY refuse collection. Therefore, CD is considered as non-recyclable in Figure 3.3 and records a small quantity (4.5%).

In Montreal, 931 kt waste was generated. This amount is comprised of 95% recyclables (OW, PC, MGP, CD, textile, e-waste and harmful household product) and 5% non-recyclables (non-recyclable CD and other materials) and the portion of diverted and landfilled waste was 45% and 55%, respectively. OW in Montreal accounted for 369 kt from which around 85 kt was recovered. The recovery ratio of OW increased from 11% in 2012 to 23% in 2017 but is still far from the 60% recovery target in 2011-2015 Action Plan of the Government of Quebec. The recovery ratio of PC and MGP, and CD was 60%, and 68%, respectively. To compare, household waste collected from urban and rural sectors of Saguenay in the Canadian province of Quebec comprised of 53% to 66% OW, 4% PC, 15% MGP and 5% textile [130]. The waste composition of Tokyo comprised of 27% OW, 48% PC and 20% MGP in 2018. The waste flow of Tokyo in 2018 is shown in Figure 3.4 as an example. According to that, from 2.7 million tons waste, 88% was incinerated mainly due to hygienic reasons, 2% was recovered and 11% was landfilled in Tokyo in 2018.

The waste flow also shows the percentage of FW and yard waste for the OW of New York City and Montreal. In case of Montreal, this percentage was not available for 2017 and we assumed the same percentage in 2016 [113]. Accordingly, FW accounted for 21% (641 kt) and 9% (81 kt) and yard waste was 6% (170 kt) and 14% (133 kt) in New York City and Montreal, respectively. The rest of the OW was 8% (251 kt) food-soiled paper in New York City and 17% (155 kt) mixed residue in Montreal.

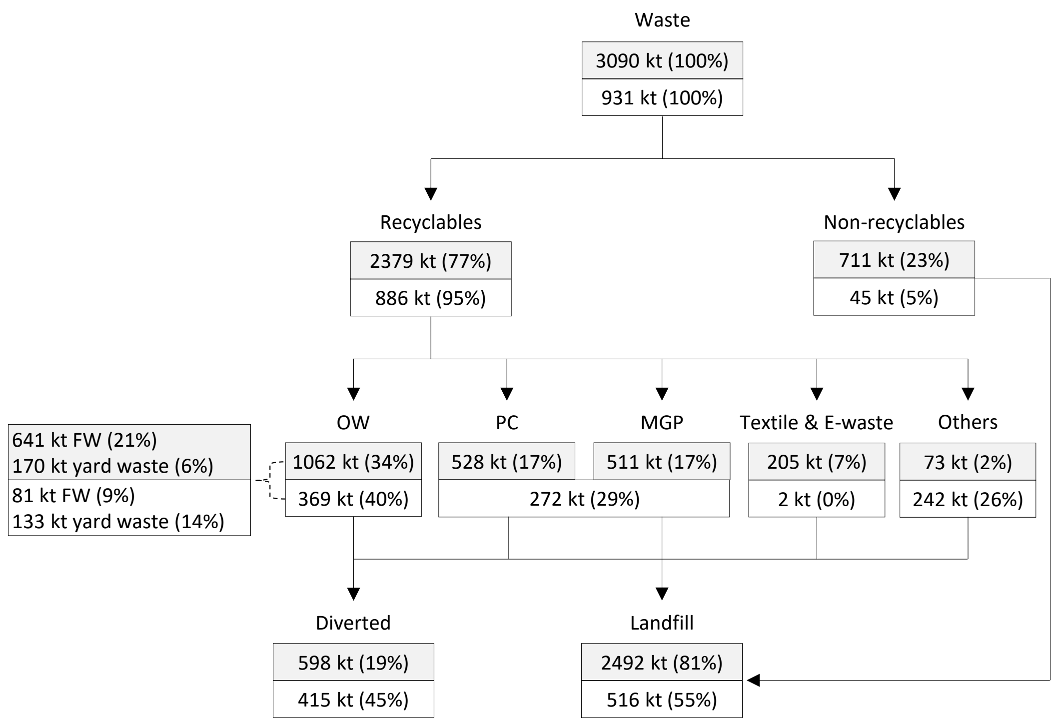


Figure 3.3: Waste flow of New York City (upper values) [118] and Montreal (lower values) [113, 121] in 2017.

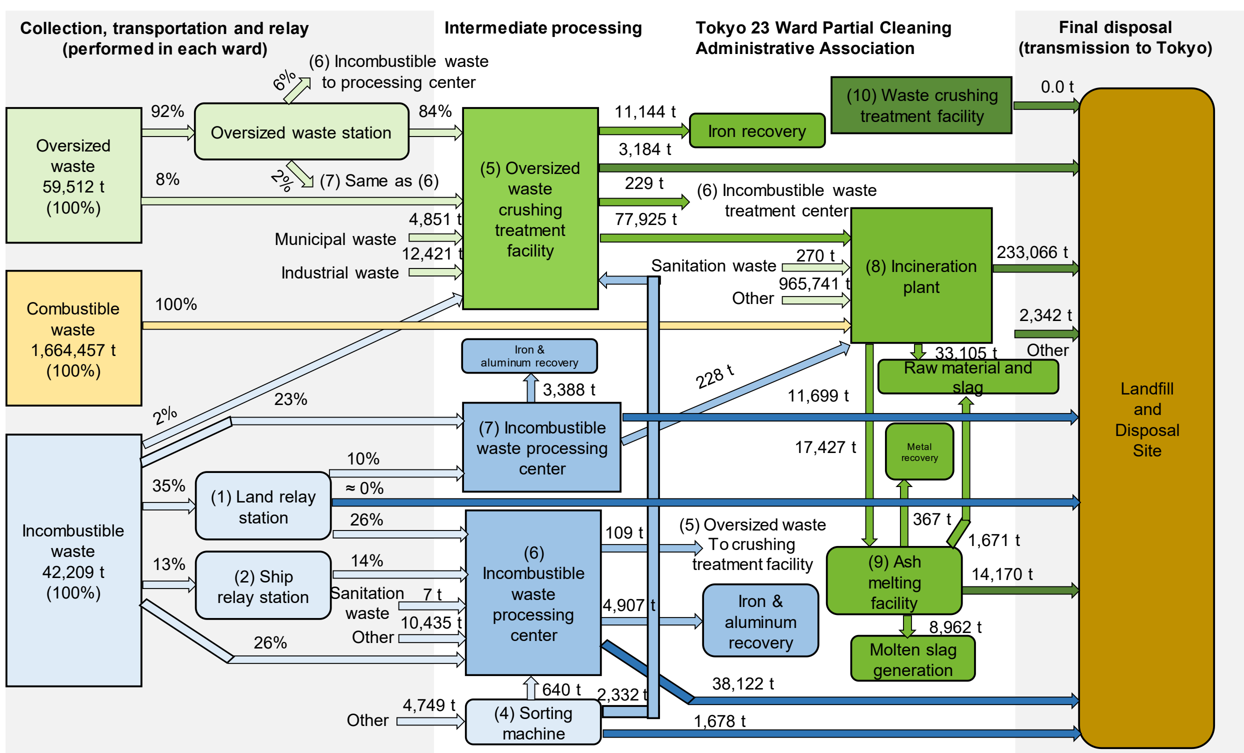


Figure 3.4: Waste flow of Tokyo in 2018 [131].

3.3.3 *Energy recovery, conversion and benefits*

Using the Buswell equation and the molar ratios assumed, the biogas generation potential from 641 kt FW and 170 kt yard waste in New York City and 81 kt FW and 133 kt yard waste in Montreal was calculated. According to the result, the biogas generation is 487 million m^3 from FW and 143 million m^3 from yard waste in New York City and 62 million m^3 from FW and 112 million m^3 from yard waste in Montreal. Table 3.1 shows the HCV and LCV of FW and yard waste (solid base), and biogas (gas base) in New York City and Montreal. According to that, HCV and LCV from FW are 3,482 and 2,792 GWh in New York City and 441 and 354 GWh in Montreal respectively. In case of yard waste, HCV and LCV are 816 and 681 GWh in New York City and 636 and 531 GWh in Montreal respectively. Gas base calculation of HCV and LCV results are close the mentioned solid base values. Totally, HCV from FW and yard waste or biogas is more than 4,000 GWh in New York City and more than 1,000 GWh in Montreal. The total electricity consumption in New York City was 156,370 GWh in 2017 [132] and in Montreal was about 41,613 GWh in 2016. Considering the HCV, this amount would mean a contribution of around 2.5% energy (assuming 100% conversion efficiency) in New York City and Montreal.

3.4 Conclusion

The work presents a comparison on the waste flow in the two cities New York City and Montreal and estimates the potential of biogas generation from their FW and yard waste. It shows the huge potential of energy recovery from FW and yard waste in these cities instead of landfilling them as is the current OW management method. From a total of 3,090 kt generated waste in New York City in 2017, only 19% was diverted and 81% was landfilled. OW with one third contribution to the total generated waste accounted for 1,062 kt from which approximately 13 kt was recovered and the vast majority was landfilled. In Montreal in the same year, 931 kt waste was generated and the portion of diverted and landfilled waste was 45% and 55%, respectively. OW in Montreal accounted for 369 kt from which around 85 kt was recovered. Using the Buswell equation and the molar ratios assumed, the biogas generation potential was 487 million m^3 from FW and 143 million m^3 from yard waste in New York City and 62 million m^3 from FW and 112 million m^3 from yard waste in Montreal.

Table 3.1: HCV and LCV of FW and yard waste (solid base) and biogas (gas base) in New York City and Montreal in 2017 (GWH).

	Solid base				Gas base			
	New York City		Montreal		New York City		Montreal	
	HCV	LCV	HCV	LCV	HCV	LCV	HCV	LCV
FW	3482	2792	441	354	3337	3005	423	381
Yard waste	816	681	636	531	750	675	584	526
Total	4298	3473	1077	884	4087	3681	1007	907

Chapter 4. Improving municipal solid waste management strategies of Montreal (Canada) using life cycle assessment and optimization of technology options

4.1 Introduction

The performance of waste management methods depends on the waste composition and climate conditions. The agglomeration of Montreal (Montreal) in the province of Quebec (Canada) was chosen as the case study. Due to the severe weather condition in this province, the energy consumption of Quebec's residents is one of the highest in the whole world. Currently, about half of the energy demand in Quebec is supplied through renewable sources and the Quebec government has ambitious plans of increasing this amount to 60.9% in 2030 [117]. One of the key targets in this plan is increasing bioenergy production by 50% by 2030. Montreal as the biggest city in this province plays a vital role in achieving those targets. Therefore, in this study the authors focused on the current waste flow in Montreal and presented different possible scenarios for municipal solid waste management in this city. The challenges to achieve optimized waste flows are discussed. The focus of the present study is on municipal solid waste, which includes residential, commercial, and institutional waste, and excludes industrial, construction, and hazardous waste [133]. To the best of the authors' knowledge, there are no LCA studies of waste, and especially OW management for the chosen case study city.

The chapter is organized as followed: First, the waste composition in Montreal, and the current waste flow in the city is presented. Then, different municipal solid waste management scenarios are defined and their environmental performance are compared using IWM-2 LCA methodology. In the next section, based on the preliminary LCA results, a mathematical model is developed and the waste flow is optimized to minimize the equivalent CO_2 of GHG emissions, total energy consumption in waste management system, and total cost of the system. In the final section, the challenges in the way to reach to this optimized waste flow are discussed. The aim of this study was LCA of the current waste management system and waste management systems with new technologies in Montreal.

4.2 Current status of waste management in Montreal

The work uses the Canadian city Montreal as a case study. As mentioned in Chapter 3, Montreal has around 2 million inhabitants and the city plans to prohibit the disposal of OW and divert 60% of OW from landfill. Therefore, the environmental assessment of the current and proposed waste management systems is essential regarding their impact on energy consumption and emissions. Data collection and analysis for the waste flow of Montreal and waste generation and flow of this city were explained in Chapter 3. As explained previously, 931 kt waste was generated in 2017 and the whole amount of this waste was the subject of the research. Adding mixed paper from OW (139 kt) to PC and MGP (272 kt), recyclables could account for 357 kt. 369 kt OW was generated and about 85 kt of it was recovered. Also, the percentage of FW and yard waste was not available for 2017, and we assumed the same percentage in 2016. Hence, FW accounted for 9% (81 kt), and yard waste was 14% (133 kt). The rest of the OW was 17% (155 kt) mixed residue.

4.3 Methodology

4.3.1 LCA methodology

IWM-2 was used as the LCA methodology to predict the environmental burdens of integrated waste management systems [134, 135]. The scope of the environmental analysis model was defined to include the major components of residential waste, including paper, plastic, glass, aluminum and steel, FW, and yard waste. Other types of wastes were considered as components which could be treated through energy recovery and landfilling options [135]. Goals, functional unit, and system boundary, life cycle inventory, and life cycle impact assessment in the following sections were based on IWM-2.

Goals, functional unit and system boundary

LCA of the current waste management systems and waste management systems with new technologies in Montreal was considered. These technologies were based on composting of FW, and energy recovery from mainly FW and yard waste. The total waste generated in Montreal in 2017 was considered as a functional unit in the mentioned systems. The model evaluates the environmental burdens associated with waste management from the point at which a material is discarded into the waste stream to the point at which it was either converted into a useful material or, it was finally disposed [134]. Accordingly, waste collection, waste transfer, sorting of recyclable materials at a materials recovery facility, reprocessing of recovered materials into recycled materials, composting, energy recovery and landfilling were evaluated by the model through recycling of paper, plastics, glass, steel, and aluminum, composting and AD of paper, yard waste and food waste, and incineration and landfilling of all waste components [135]. However, in this study, only AD and composting of FW and incineration of yard waste was considered.

According to IWM-2 [135], "the life cycle of a waste starts when a product is discarded into the waste stream and terminates when the waste material has either been converted into a resource (recycled material or recovered energy) or, when it has been finally disposed. Recovery processes such as recycling, energy recovery, composting and anaerobic digestion which result in the production of a usable material can be viewed as fulfilling two functions: the management of waste (the waste management function) and, the production of useful material or energy (the production function). The recovered material/energy can be used in place of conventional material/energy (e.g., virgin raw materials, energy production from fossil fuels combustion, etc.). A life cycle study of a waste material must therefore take into account the avoided environmental burdens associated with the production of the displaced conventional material/energy. This approach allows the intrinsic environmental value of recovered material and energy to be accounted for in the same way that their economic value is considered by accounting for material and energy revenues. The life cycle boundaries for each of the waste management processes evaluated by the model are described below.

The system boundary for recycling used in the environmental analysis model starts at the point at which a recyclable is set out at the curb for collection and ends when a recycled material (which can be used as a substitute for a conventional material) is produced. Energy and emissions associated with the production of conventional material (e.g., virgin raw materials) that can potentially be replaced by the recycled material are estimated and accounted for as avoided emissions.

The system boundary for anaerobic digestion extends from the collection of organics or co-mingled wastes at the curbside through to the production of products including – heat and electrical energy and compost. Secondary materials are sent to a material recovery facility for further processing, residues are sent to landfill and emissions are discharged into the atmosphere and hydrosphere. The anaerobic digestion process typically employs combusting the treated biogas to result in heat and electricity. Some of the energy produced is used on-site to operate the facility and the remainder is exported. Energy and emissions associated with the production of grid electricity that can be replaced by burning biogas are estimated and accounted for in a fashion paralleling the procedures employed in the model for energy recovery from burning landfill gas. Net burden can be calculated from the combustion of the biogas and production of conventional energy."

Regarding the emissions of waste management systems:

- In the case of landfilling, IWM-2 assumes three sources of air emissions: a) produced landfill gas, b) landfill gas combustion and c) dust from landfilling operation.
- In the case of recycling, IWM-2 calculates the environmental burdens associated with the reprocessing that recovered materials have to undergo in order to be usable as substitutes for conventional materials (deinking, re-pulping, de-tinning, etc.). It also evaluates the burdens avoided as a result of displacing virgin material.
- In the case of anaerobic digestion, IWM-2 assumes two sources of air emissions: a) combustion of the produced biogas and b) emissions that occurs during the curing of the organic digestate which is typically dewatered and then placed into windows for the final curing.

Hence, increasing the CO_{2eq} and energy saving due to an increase in recycling or anaerobic digestion can be explained by considering the entire life cycle of waste management and the specific context of the study. For example, various stages of reprocessing the recovered materials and considerations of displacing virgin materials are taken into account for recycling. Additionally, transportation plays an important role in emissions and energy changes of waste management system (e.g., the distance from transfer station to material recovery facility or anaerobic digestion plant).

Life cycle inventory

The analysis of all the material and energy inputs and outputs for each stage in the life cycle could be combined to give the overall life cycle inventory [54]. The overall estimation of energy consumption and emissions of the waste management systems in this study was conducted with the help of the IWM-2 model and its pre-defined standard data in Microsoft Excel for Office 365 MSO version 16 and Microsoft Visual Basic for Applications version 2012.

Life cycle impact assessment

The model estimated the energy consumed (or produced) and the emissions to air, water, and land associated with different waste management practices [134]. The specific indicator parameters evaluated, and the environmental effects associated with these parameters are shown in Table 4.1.

4.3.2 Waste management scenarios

LCA has been conducted for the current waste management systems in Montreal (Sc1) and three proposed scenarios (Sc2, Sc3, and Sc4) in which the energy consumption and emissions have been determined. All the proposed scenarios considered the maximum amount of recycling rates. They also fed all the yard waste to incineration technology because lignin does not undergo AD, and cellulose and hemicellulose are degraded slowly in comparison [47]. In the case of FW, Sc2 specified all the FW for the AD technology, Sc3 assumed all the FW for the composting technology, and Sc4 allocated half of the FW for the AD technology and the other half for the composting technology. Figure 4.1 shows the amount of input waste for waste management scenarios in Montreal.

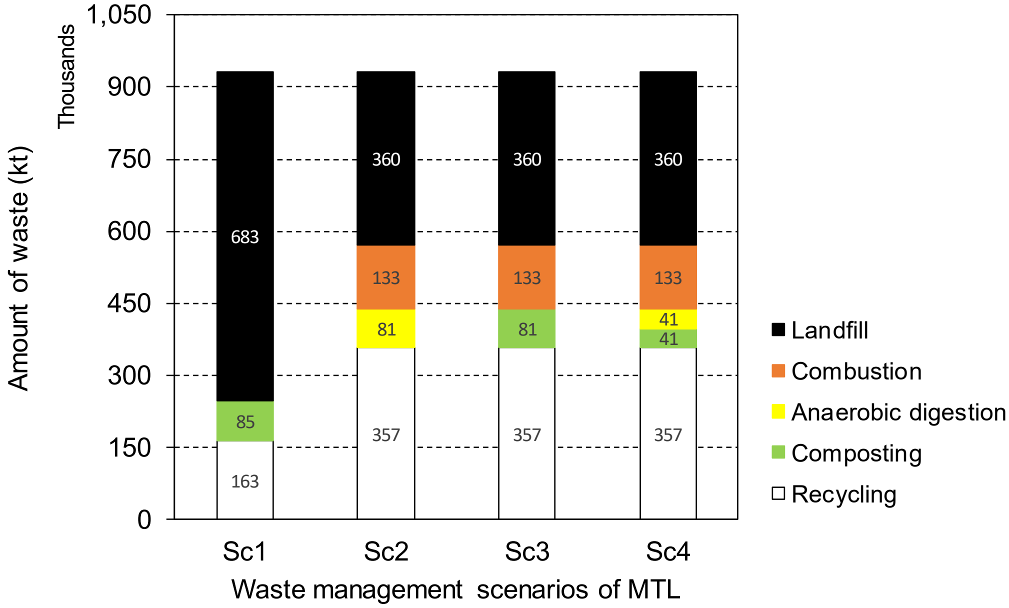


Figure 4.1: Amount of input waste for waste management scenarios in Montreal.

To better understand the IWM-2 LCA model, further preliminary (non-optimizing) scenarios were defined by changing only one waste management method and run using the IWM-2 LCA model for comparison.

a) Scenario a: Recycling scenario (357 kt recycle, 85 kt compost, 489 kt landfill)
b) Scenario b: AD scenario (163 kt recycle, 85 kt AD, 683 kt landfill)
c) Scenario c: Combustion scenario (163 kt recycle, 85 kt combustion, 683 kt landfill)

4.3.3 Optimization

LCA results are useful to develop empirical correlations for the energy and environmental performance of a waste management system. The IWM-2 software is a helpful tool for conducting LCA for a waste management system as it includes all different parts of the system, including transportation, sorting, and energy recovery. The results obtained from IWM-2 included the energy consumption and the CO_2 equivalent of GHG emissions from each waste management technology that were good indicators of waste management system performance. Based on the results derived from the different proposed scenarios, the ranges of the waste sent to each technology was defined. According to these values, the amount of waste sent to each section was changed to obtain the equivalent CO_2 of GHG emissions (CO_{2eq}) and energy consumption of each technology (E) for the specified waste amount. By using these results, a curve fitting tool was applied to develop a second-order mathematical relationship for both energy consumption and CO_{2eq} of GHG as a function of waste input [52]. To achieve a better fit, the data for all technologies were normalized. The general form of the Equation for each technology is as follows.

$$E = a\left[\frac{x - \mu}{\sigma}\right]^2 + b\left[\frac{x - \mu}{\sigma}\right] + c$$ **Eq. 4.1**

$$CO_{2eq} = a\left[\frac{x - \mu}{\sigma}\right]^2 + b\left[\frac{x - \mu}{\sigma}\right] + c$$ **Eq. 4.2**

where E is the energy consumption in each technology, CO_{2eq} is the equivalent CO_2 of GHG emissions from each technology, x is the amount of waste sent to each section, and μ and σ are the mean and variance of the data obtained for each technology, respectively, and a and b and c are constants. The values of μ and σ are derived from the curve fitting tool utilized for developing the Equations.

Objective functions

The total energy consumption and CO_{2eq} emission will be calculated from the following equations [52].

$$E_{total} = E_{AD} + E_C + E_{Cbs} + E_L + E_R$$ **Eq. 4.3**

$$CO_{2eq.total} = CO_{2eq.AD} + CO_{2eq.C} + CO_{2eq.Cbs} + CO_{2eq.L} + CO_{2eq.R}$$ **Eq. 4.4**

where $E_{AD}, E_C, E_{Cbs}, E_L, E_R$ are energy consumption by AD, composting, incineration (or combustion), landfilling, and recycling units, respectively. The energy consumption from waste facilities, including AD and incineration, are calculated from the electrical energy generated minus the energy consumed. The saved energy by using recovered material is subtracted from the energy consumed for recycling the material for the recycling unit. $CO_{2eq.AD}, CO_{2eq.C}, CO_{2eq.Cbs}, CO_{2eq.L}, CO_{2eq.R}$ are equivalent CO_2 of GHG emissions of the AD, composting, incineration, landfilling, and recycling units, respectively. These quantities show the total emissions of each technology in their life cycle based on CO_{2eq}. These two quantities are functions of waste input in each technology. In this optimization procedure, two objective functions are minimizing the total energy consumption and CO_{2eq} of GHG emissions.

Cost is a vital factor in designing an integrated waste management system. Table 4.2 contains the estimated cost function for each waste management technology. The costs are categorized into the initial capital and operating costs, and the parameter x denotes the annual waste input of each technology. In addition to the total energy consumption and CO_{2eq} emission, the waste management system's total cost will be also considered an objective function to be minimized [52].

$$Cost_{total} = Cost_{AD} + Cost_C + Cost_{Cbs} + Cost_L + Cost_R$$ **Eq. 4.5**

Consider *X* to be a vector containing the waste input of each technology ($X = (x_{AD}, x_C, x_{Cbs}, x_L, x_R)$), and the arrays of this vector are decided by the optimization constraints. Then, *X** is optimal in space S if energy, CO_{2eq} and total cost were minimized.

$$E_{total}(X^*) \& CO_{2eq.total}(X^*) \& Cost_{total}(X^*) \\ \leq E_{total}(X) \& CO_{2eq.total}(X) \& Cost_{total}(X) for\ all\ X \in S$$ **Eq. 4.6**

Constraints

Based on the results obtained from IWM-2, the best scenario was chosen. Therefore, the optimization constraints could be determined according to the chosen scenario. The lower bounds for the waste input of landfill and recycling units were the current amount of waste sent to these units in Montreal.

Method

An optimization algorithm was used to find the best waste flow for the waste management system in Montreal. This optimization algorithm was a multi-objective one as the proposed system should be both environmentally friendly and economically feasible.

GA is a popular option for solving such constrained multi-objective optimization problems. GA has been evolved into different forms that each of them is different from the original GA. One of these evolved forms is a non-dominated sorting genetic algorithm (NSGA) developed by Srinivas and Deb [136]. The difference between NSGA and original GA is only in how the selection operator works while crossover and mutation operators remain the same.

In this study, an improved form of NSGA, meaning NSGA-II [137] was used to minimize the energy consumption, CO_{2eq} of GHG emissions, and cost of the system. This improved algorithm was less complicated in terms of calculations, and the solutions were more diverse compared to original NSGA [138].

Figure 4.2 illustrates the procedure of optimization. The first step was an initialization, which included defining objective functions, input variables, and constraints. The mass input of the five technologies (AD, compost, incineration, landfill, and recycling) were taken as input variables. In the next step, a set of values was assigned to the defined input variables as the initial population and a fitness function was found for each set of answers in the next step. Then, using this initial population, the values of the objective functions were calculated to identify the answers that minimize them. Next, the non-dominated sorting was done to order the answers based on their fitness functions. In step 4, parent chromosomes were chosen among the ordered initial population. The crossover process was used to generate children for the chosen parents. In the following step, the mutation operator was utilized for the children. Unlike the crossover process where the children have the same characteristics as the parents, after mutation, some of the children gain characteristics that belong to neither of parents. Then, these mutated children are mixed with other children, and again non-dominated sorting will occur, and children are chosen for the next generation. Finally, the stop criterion of the algorithm was checked. The steps 2 to 8 were repeated until this criterion is met.

Table 4.1: Indicator parameters [52, 134].

Indicator parameter	Indicator of	Unit
Energy		
Total energy consumed	Resource depletion	GJ
Emissions to air		
Greenhouse gases	Climate change	t
Carbon dioxide (CO_2)		
Methane (CH_4)		
Acid Gases	Acidification, health risk	t
Nitrogen oxides (NOx)		
Sulphur dioxide (SO_2)		
Hydrogen chloride (HCl)		
Smog precursors	Urban smog formation, health risk	t
Volatile organic compounds (VOC)		
Nitrogen oxides (NOx)		
Particulate matter (<10 microns) (PM-10)		
Heavy metals	Health risk	kg
Lead (pb)		
Cadmium (Cd)		
Mercury (Hg)		
Trace organics	Health risk	g
Dioxins & Furans (TEQ)		
Emissions to water		
Heavy metals	Health risk, environmental degradation	kg
Lead (pb)		
Cadmium (Cd)		
Mercury (Hg)		
Trace organics	Health risk, environmental degradation	mg
Dioxins & Furans (TEQ)		
Biochemical Oxygen Demand (BOD)	Water quality, environmental degradation	kg
Emissions to land		
Residual solid waste	Land use disruption	t

4.4 Results and Discussion

4.4.1 LCA Results

Greenhouse gases (carbon dioxide (CO_2), methane (CH_4) and nitrogen oxides (NO_x), and the CO_{2eq} in kt of CO_2), acid gases (nitrogen oxides (NO_x), sulfur oxides (SO_x) and hydrochloric acid (HCl)), smog emissions (NO_x, particles (PM) and volatile organic compounds (VOC)), energy consumption and remaining amounts in the current status of waste management and the proposed scenarios in Montreal are presented in Table 4.3. In Montreal, Sc1 consumes the most energy (6,892 TJ saving) and emits the most CO_{2eq} (158 kt). Sc3 and Sc4 are the better waste management scenarios in which 14,027 TJ and 14,043 TJ energy is saved, and 127 kt and 144 kt CO_{2eq} is emitted, respectively. Each waste management system has its own residuals, for example ash in incineration, or residuals that could not being further recycled in recycling etc. These residuals are usually sent to landfill.

With Sc1 as the reference scenario of today's waste management system, Sc2 uses anaerobic digestion, Sc3 uses compost, and Sc4 uses anaerobic digestion and compost. The equivalent CO_2 is calculated using $CO_{2eq} = CO_2 + 21 * CH_4 + 310 * NO_x$. In the AD module (Sc2), emissions originate from biogas combustion (GHG contributor), aerobic composting (GHG contributor), and water (leachate) from the process. All the CH_4 produced is typically combusted, and the resultant CO_2 emissions are not counted.

Biogas combustion emits NO_x too, which contributes to equivalent CO_2 emission. Therefore, emissions of CH_4 + NO_x and equivalent CO_2 are higher for the scenarios with a higher AD ratio. The model does not consider offsetting the combustion of fossil fuels rather than biogas and, consequently, emission saving. Process emissions of composting include only one direct emission, which originates from aerobic composting. GHG emissions of composting are thus lower than AD.

Table 4.2: Costs in Canadian dollars per ton of annual waste input (x).

Technology	Capital Cost (CAD/t)	Operating Costs (CAD/t)	Reference
Recycling	190 x	190 x	[139]
Composting	4000 $x^{0.7}$	7000 $x^{-0.6}$	[140]
AD	35000 $x^{0.6}$	17000 $x^{-0.6}$	[140]
Incineration	5000 $x^{0.8}$	700 $x^{-0.3}$	[140]
Landfilling	6000 $x^{0.6}$	100 $x^{-0.3}$	[140]

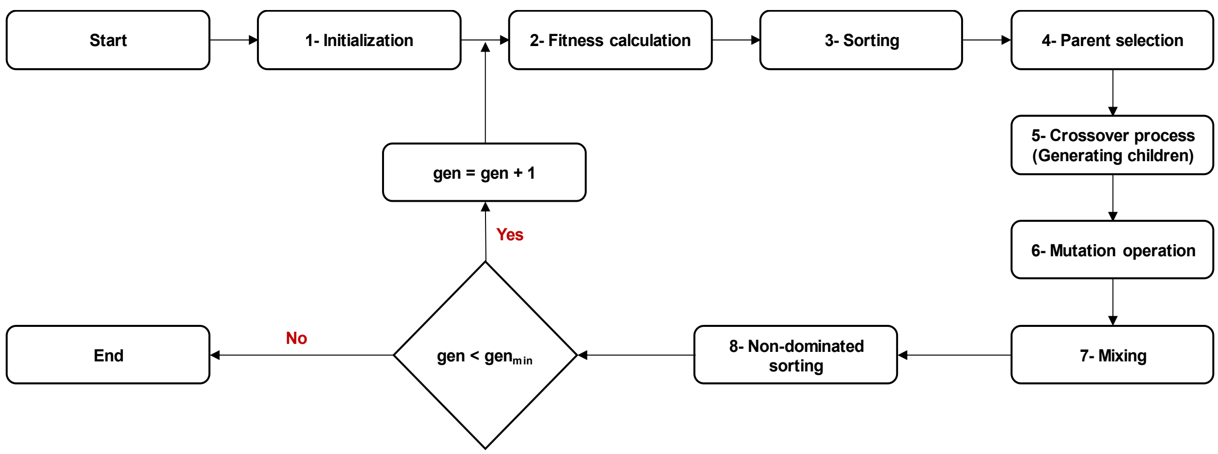


Figure 4.2: Procedure of optimization.

Among the proposed alternative scenarios, Sc2 indicates that using the AD unit as the sole FW treatment technology has the lowest efficiency in Montreal. One of the reasons can be attributed to the FW quantity in Montreal which does not seem high enough for the maximum simultaneous reduction of energy consumption and CO_2 production by AD. It is worth mentioning that IWM-2 estimates the material-specific AD yields of CH_4 and CO_2 based upon the lab studies of AD of MSW in landfills. Accordingly, various kinds of AD set-ups are neglected, and hence, further LCA studies are required based on lower FW quantity feeding to AD units in order to find out the energy consumption and CO_2 emissions of each set-up. Studies showed that although there were 688 centralized AD plants for biowaste treatment (average capacity 31,700 ton/year) in EU in 2016, small scale AD (5.2 ton/year) can be technically viable with potential biogas production performance like large scale AD (3,372 ton/year) [141].

4.4.2 Optimization Results

Based on the results obtained from IWM-2, the best scenario is when all the FW is divided between AD and compost, all the yard waste is sent to an incinerator, and all the recyclable materials are recovered. Therefore, the optimization constraints can be determined. As mentioned previously, the lower bounds for the waste input of landfill and recycling units are the current amount of waste sent to these units in Montreal.

$$0 \leq x_{AD} \leq 81 \quad \textbf{Eq. 4.7}$$

$$0 \leq x_{C} \leq 81 \quad \textbf{Eq. 4.8}$$

$$x_{AD} + x_{C} = 81 \quad \textbf{Eq. 4.9}$$

$$0 \leq x_{Cbs} \leq 133 \quad \textbf{Eq. 4.10}$$

$$360 \leq x_{L} \leq 683 \quad \textbf{Eq. 4.11}$$

$$163 \leq x_{R} \leq 357 \quad \textbf{Eq. 4.12}$$

Table 4.4 shows the parameters of Eq. 4.1 and Eq. 4.2 for each waste management technology, obtained by using MATLAB curve fitting tools. To explain more, based on the result of the fourth LCA scenario (Sc4), the waste input to each technology was changed and the amount of CO_{2eq} of GHG emissions and energy consumption of each technology was recorded. Then, two curves were fitted for unit based on their waste input (one for CO_2 and one for Energy). These values were used for doing the optimization. The results of two-objective and three-objective optimizations are shown in Figure 4.3.

Table 4.3: The emissions in the current status and proposed scenarios of waste management in Montreal.

Case	GHG emissions (kt)			Acid gases emissions (kt)			Smog emissions (kt)			Residuals (kt)	Energy (TJ)
	CO_2	CH_4+NO_X	CO_{2eq}	NO_X	SO_X	HCl	NO_X	PM	VOC		
Sc1	8	6	158	0.08	0.02	0.00	0.08	0.22	0.04	695	-6,892
Sc2	9	4	142	0.19	0.04	0.02	0.19	0.13	0.05	427	-13,530
Sc3	9	3	127	0.17	0.04	0.02	0.17	0.15	0.04	417	-14,027
Sc4	11	3	144	0.20	0.04	0.02	0.20	0.14	0.06	423	-14,043

We have three independent functions. The functions for CO_2 and energy consumption for each technology are in equations Eq. 4.1 and Eq. 4.2. Based on the LCA results, these equations were derived using MATLAB's curve fitting tool. The summation of CO_2 and energy consumption of all technologies gives total equivalent CO_2 emissions and energy consumption, respectively. The third function is the total cost of the system. The cost functions of different technologies are found in the literature (Table 4.2).

The optimization gives 500 solutions and a Pareto front as the set of solutions, that are optimal, as no objective can be improved without sacrificing at least one other objective for the optimization problem. For example, in Table 4.5, the best three combinations from these 500 solutions have been considered, that correspond to the minimum carbon emissions and the lowest energy consumption.

The number of iterations for the optimization tool was 5000. Two separate Pareto fronts were considered, one for cost versus energy and one for cost versus emissions. The following figures are the scatter plots of the total CO_2 equivalent of GHG emissions versus total cost and total energy consumption versus total cost.

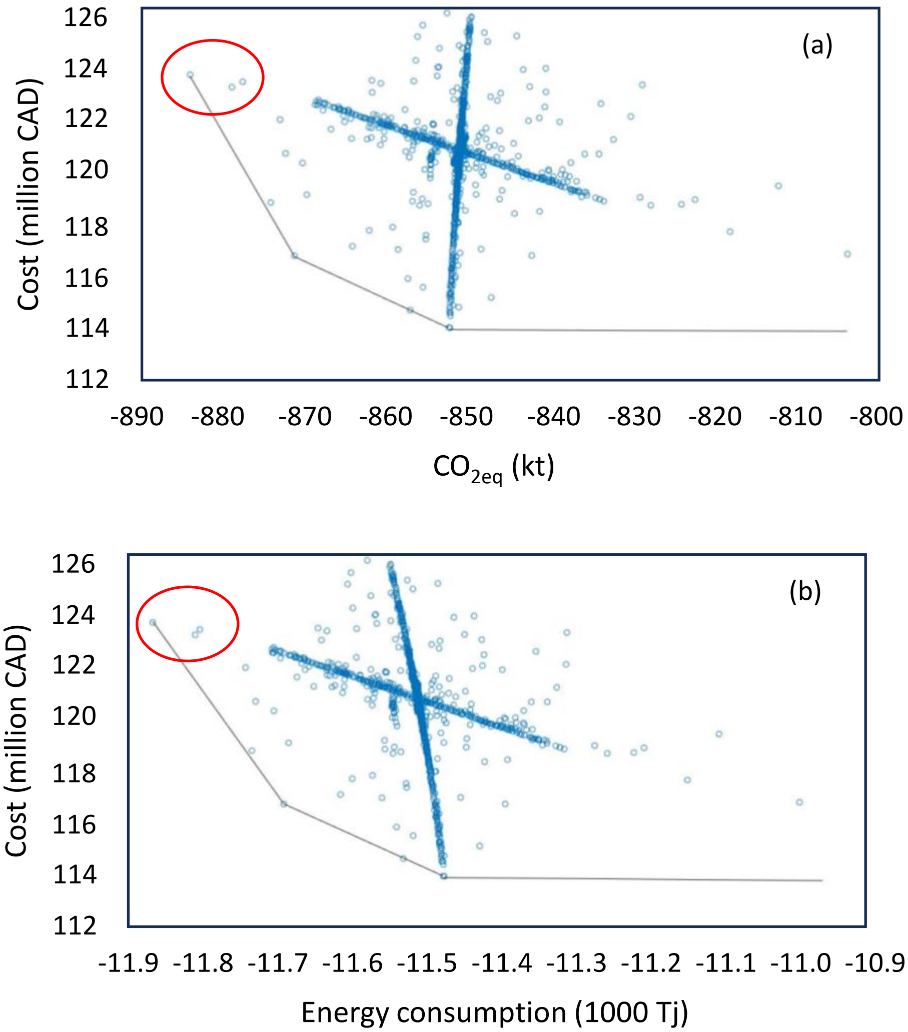


Figure 4.3: Pareto front analysis: a) cost versus CO_{2eq} and b) cost versus energy consumption (points with red circles are design numbers in Table 4.5).

This three-objective optimization for the integrated waste management system of Montreal considers the cost as an objective function in addition to total energy consumption and CO_{2eq} emissions of GHG. Table 4.5 shows the best three optimum waste flows for Montreal, that result in the lowest emissions and energy consumption.

In this lowest emission scenario, 58% of total waste should be sent to the landfill unit (540 kt). Recycling unit has the highest share after landfill (32%) equal to 295 kt. Taking FW into account, 14.7 kt should be sent to AD and 66.3 kt to composting unit, which accounts for 2% and 7% of the total waste, respectively. The incineration unit has the lowest share in the system (1% equal to 8.5 kt) of total waste (Figure 4.4).

The results of preliminary (non-optimizing) scenarios indicated that the CO_{2eq} and energy consumption of scenarios a to c (Recycling, AD and Combustion) were 122, 169, and 260 kt, and -15,163, -6,981, and -7,074 TJ, respectively. These results indicate that recycling has the most significant impact in terms of energy savings due to virgin material displacement credit. Recycling scenario also has the least CO_{2eq} emission and hence, maximum recycling should be the preferred waste management method. Waste to energy scenarios in scenarios b and c had similar energy consumption. However, in terms of CO_{2eq}

emission, combustion contributes to GHG emissions more than AD because combustion generates CO_2 not methane, and therefore, is less sustainable. In other words, methane combustion process is more efficient than just pure combustion of waste. Based on IWM-2 [135], "Allocation of greenhouse gases has been done on the basis of the carbon content of the different materials, and the combustion module assumes that approximately 5,090 m^3 of flue gas is produced per tonne of municipal solid waste combusted."

Moreover, the cost of scenarios a to c based on the equations in Table 4.2 is 162, 113, and 125 million CAD, respectively. It means that combustion is more expensive than AD and recycling is the most expensive method.

4.4.3 Discussion

It is clear from the results that the main benefit of having an integrated municipal solid waste management system is a significant reduction of energy consumption and emissions. LCA provides a comprehensive, consistent and transparent overview of flows in the waste management systems with quantification of the environmental profile [142]. Based on the optimization results, the amount of recycled waste in Montreal should increase by 87 kt per year. Also, adding incineration and AD units to the waste management system of the city would increase the share of energy produced from renewable sources. The results are consistent with the study by [143] with increasing focus on recycling. Also, [144] assessed the environmental and economic benefit of the substitution of energy crops with food waste in AD and concluded that a reduction of 42% in the carbon footprint of the electricity produced from the biogas plant can be obtained. Moreover, installing new units like AD and incineration creates more jobs in the city which is a social benefit of this municipal solid waste management system. A study by [145] showed that new jobs could be created in the various processing centers and between transportation nodes of the waste management system.

Table 4.4: Equation parameters for different waste management technologies.

Technology	Input		μ	σ	a	b	c
Recycling	x_R	Energy	2.6e5	6.2e4	-115	-1.2e6	-1.1e7
		CO_2			-10	-1.1e5	-9.4e5
Composting	x_C	Energy	1.3e4	1.4e4	23	1658	3.3e4
		CO_2			-0.03	160	8104
AD	x_{AD}	Energy	5.9e4	1.4e4	22	-5e4	-1.6e5
		CO_2			6	712	1.8e4
Incineration	x_{Cbs}	Energy	6.6e4	4.3e4	-3.6e4	-3e5	-3.8e5
		CO_2			284	7989	1.2e4
Landfilling	x_L	Energy	5.2e5	1e5	-243	3025	3.1e4
		CO_2			-974	8597	9.6e4

Table 4.5: The best three waste flows resulted from optimization.

Design number	x_{AD} (t)	x_C (t)	x_{Cbs} (t)	x_L (t)	x_R (t)	C_{total} (kt of CO_2)	$Cost_{total}$ (CAD)	E_{total} (Tj)
1	14,710	66,393	8,580	540,166	294,955	-879	123,000,607	-11,809
2	14,525	66,393	7,880	556,522	294,861	-877	123,203,356	-11,802
3	14,684	66,393	8,271	542,802	297,906	-884	123,464,210	-11,866

Employment opportunities by waste to energy include the collection and sorting of waste, waste transportation, waste plant construction, and plant operation. On average, a Waste-to-Energy plant in Europe can create 62 direct jobs, and the total direct and indirect jobs in 2011 was 56,000 in Europe [146]. However, it should be noted that all these results have been driven based on this assumption that all types of wastes are separated completely which is not possible in real life. The following bullet points summarize different challenges present in the way to achieve this optimized waste flow:

1- As mentioned above, the biggest challenge is complete separation and sorting of different types of the waste. Creating sorting units for the whole amount of waste is an expensive solution. Therefore, another solution is encouraging people to be more interested in source separation of waste.
2- Hydro Quebec is the only supplier of electricity in the province of Quebec. Unfortunately, there is no specific policy about buying self-generated electricity from private suppliers. Therefore, it affects the interest from external investors to help to construct expensive units like AD and incineration.
3- The other existing challenge is the public awareness. People should become aware of the hazards of landfilling the municipal waste and realize what an important role they play in different waste management scenarios.
4- Another challenge is the location of new AD and incineration units and whether there should be one central unit or several distributed units across the city. Although a study by [141] concluded the advantages of a fully decentralized AD systems, the authors believe that more detailed LCA studies are needed to find the solution for this problem.
5- The severe weather condition in Montreal during its long winters is another challenge for utilizing organic waste management facilities like AD and composting units. Further thermal energy would be required to keep the system in an optimum temperature condition. Especially in case of AD and composting, cold weather might slow down the degradation process. Putting the AD in a greenhouse has been suggested and is recommended. Study by [147] showed that an AD could 49% less heat energy by being housed in a greenhouse.

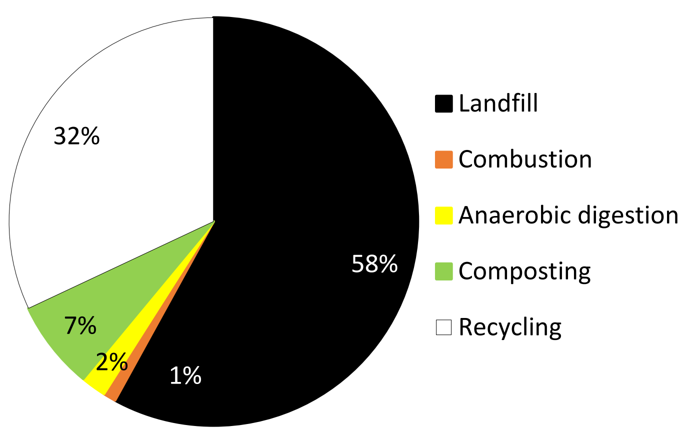


Figure 4.4: The optimized waste flow of Montreal.

4.5 Conclusion

The work presents an analysis of waste flow including OW, PC, MGP, CD, textile, E-waste, harmful household product, and other materials in a case study city Montréal. It shows the huge potential of energy recovery from FW and yard waste instead of landfilling them, as is the current OW management method. In Montreal in 2017, 931 kt waste was generated, and the portion of diverted and landfilled waste was 45% and 55%, respectively. OW in MTL accounted for 369 kt from which around 85 kt was recovered.

Four scenarios were analyzed to assess the greenhouse gas emissions and costs of different waste management strategies. With the current waste management system as the reference scenario 1, the proposed scenarios Sc2, Sc3, and Sc4 feed all the yard waste to incineration. Moreover, all the food waste goes to anaerobic digestion in Sc2 and to composting in Sc3. Sc4 considers 50% of FW for AD and composting.

The LCA study showed that in Montreal, Sc3 and Sc4 are the better waste management scenarios in which 14,027 TJ and 14,043 TJ energy is saved, and 127 kt and 144 kt CO_{2eq} are emitted, respectively. In Sc3, all the FW is managed by composting which has a high energy saving (14,027 TJ). Additionally, this scenario has less CO_{2eq} emission (127 kt) compared to Sc4. In Sc4, half of the FW is managed by composting and the other half by AD. The energy saving is slightly higher than Sc3 (14,043 TJ) because the quantity of FW is not very high. Additionally, AD causes more CO_{2eq} emission (144 kt). Sometimes, the energy inputs for anaerobic digestion, such as the energy required for maintaining the optimal temperature, can be higher than anticipated, offsetting the energy gains from methane production. As the energy consumption for the whole AD process is higher than the energy generation, it could lead to more energy saving by composting compared to AD. According to IWM-2 [135], "composting requires energy for: i) the collection and transportation of waste materials; ii) process energy to aerobically biodegrade the waste; and, iii) the production and delivery of the fuels and electricity used in (i) and (ii). On the other hand, the AD module calculates the amount of energy consumed by the processes including the electricity needed to operate sorting equipment and de-watering apparatus and the energy consumed while maintaining proper operating temperatures within the digester. A default value of 22% energy consumption has been included and is based upon experiences reported for Canadian facilities. In some plants the amount of energy produced on site is less than the amount consumed by plant operations. In such cases, supplemental energy will be required from the local power grid".

Based on the results obtained from LCA studies, NSGA-II was used as an optimization algorithm to optimize the waste flow in MTL. The objective functions were minimizing the total energy consumption and CO_{2eq} emission of GHG and the total cost in the waste management system. The optimized waste flow for Montreal by using a three-objective optimization algorithm is sending 2% of waste (14.7 kt) to AD, 7% (66.3 kt) to compost, 32% (295 kt) to recycling, 1% (8.5 kt) to incineration, and 58% (543 kt) to landfill. This scenario provides the lowest emissions and lowest energy consumption. Based on the optimization results, the benefits of this integrated municipal solid waste management system are significant reduction of energy consumption and equivalent CO_2 emissions. The other benefits are increasing the share of renewable energy production and creating more jobs through construction of AD and incinerations units. However, this should be noted that in all these scenarios it has been assumed that different waste types are completely separated. Therefore, proper separation and sorting of recyclable material, food waste and yard waste is a big challenge. Another challenge is the lack of a specific policy for buying self-generated electricity, which reduces the interest from external investors to invest into the construction of AD and incineration units in the city. The other challenge is low public awareness about the dangers of landfilling and their important role in having an efficient municipal solid waste management system. Finally, the severe weather conditions during the long winters of Montreal could affect the efficiency of AD and composting and these units would need further thermal energy to operate properly.

The final conclusion is as follows.

- Energy Recovery: Anaerobic digestion can generate energy in the form of biogas, which can be used for electricity generation or as a fuel source. Similarly, some recycling processes can recover and use energy from certain waste materials. This energy recovery can offset the need for fossil fuels, leading to a net energy savings.
- Material Savings: Recycling can also lead to material savings by reducing the need to produce new items from raw materials. Manufacturing products from recycled materials often requires less energy and resources compared to using virgin materials, which contributes to energy and resource savings over the entire product life cycle.
- Transportation and Processing: The specific waste management processes and transportation involved in recycling and anaerobic digestion may be more energy-efficient than traditional waste disposal methods like landfilling or incineration. The reduced energy requirements for collection, transportation, and processing can contribute to energy savings.

Chapter 5. Optimization of landfill gas generation based on a modified first-order decay model: A case study in the province of Quebec, Canada

5.1 Introduction

Today, more than 90% of waste in low-income countries is still openly dumped or burned, and gaseous emissions from such dumpsites have been poorly studied [3], threatening the environment and human health. In contrast, waste in developed countries is a resource for energy production [148], and only 2% is dumped in high-income countries [1]. Landfills are large and heterogeneous emitting sites that contribute 20% and 17.4% of national methane (CH_4) emissions in Canada [4] and the U.S. [5], respectively. Landfills will likely remain essential to integrated solid waste management systems in many developed and developing countries for the foreseeable future [6]. Understanding Greenhouse Gas (GHG) sources and sinks is a significant endeavour, and many countries have committed to reducing their emissions (e.g., UN COP26 2021 [16]). CH_4 is a potent GHG with a global warming potential of 28 times that of carbon dioxide (CO_2) over a 100-year timeframe [2].

Human-related activities such as fossil fuel production, domestic livestock ranching/farming, manure management, rice cultivation, biomass burning, and waste management cause 60% of global CH_4 emissions compared to natural CH_4 emitters such as wetlands, termites, oceans, freshwater bodies, and wildfires [64]. The waste sector presents an appreciable potential for emissions reduction, particularly in developing countries where emissions from waste can account for 15% of total country GHG emissions due to the higher content of biodegradable Organic Waste (OW) [65]. If engineered sanitary landfills are managed correctly and Landfill Gas (LFG) collection efficiency improves, emissions from landfills can be decreased.

Originating from waste decomposition, LFG mainly contains CH_4, CO_2, and trace amounts of hydrogen sulfide (H_2S) as an inhibitory (reducing CH_4 generation), odorous and corrosive gas [10]. LFG production occurs in five phases. The aerobic condition in the first phase takes hours to weeks, the anoxic condition in the second phase takes 1-6 months, and the subsequent phases are anaerobic and take several months to years [66]. The anaerobic phases are the ones that LFG generation is usually addressed. Under anaerobic conditions in landfills, CH_4 generation starts and increases, and CO_2 generation decreases (third phase). The trend continues until it reaches a steady state in the fourth phase and finally approaches zero in the fifth phase.

Anaerobic degradation of biodegradable OW (food waste, yard waste, etc.) generates CH_4. There is as low as approximately 0.1% sulphur in Municipal Solid Waste (MSW) [68]. Sulphur-containing materials such as construction and demolition waste (CD) (including bulky materials of construction and demolition (BCD) and fines of construction and demolition (FCD) wastes) can produce H_2S in an anaerobic environment. Compared to MSW, around 1.5-9.1% of CD is sulphate [69]. These wastes are landfilled with MSW in many cases. The byproduct of CD processing facilities is screened materials termed FCD (soil and building material, including drywall). These fines are often used in MSW landfills as alternative daily cover [70] or final cover.

According to [149], in Canada, "British Columbia, Alberta, Ontario and Quebec have regulations requiring larger landfills to capture and control or reduce CH_4 emissions, and others include requirements for installing LFG recovery and flaring systems in operating permits. Quebec and Ontario require landfills larger than 1,500,000 m^3 of waste capacity to install systems. British Columbia requires landfills with greater than 100,000 tons of waste or greater than 10,000 tons disposed of per year to evaluate their annual CH_4 generation and install LFG systems if they exceed 1,000 tons of CH_4 per year. The lowest regulatory threshold in North America is in California, which requires landfills that generate LFG with a heat input capacity of more than 3.0 MMBtu/hr (~650 tons CH_4 generation per year) to install LFG recovery systems."

The first-order kinetic model (e.g., Landfill Gas Emissions Model (LandGEM)) is the most widely applied model to forecast landfill CH_4 generation [9]. The U.S. EPA [96] developed LandGEM, which considers the CH_4 generation potential, L_0 (m^3/t biodegradable waste), and the CH_4 generation rate associated with waste decomposition, k_{CH_4} (y^{-1}). Although LandGEM models CH_4 generation from heterogeneous wastes effectively, some input modifications could significantly enhance its accuracy. For instance, [97] concluded that LandGEM overestimated CH_4 generation. One reason could be that LandGEM only includes one type of waste, which includes inerts. Reformulating the model by involving different types of biodegradable organic wastes, such as fast decaying refuse (FDR) and slow decaying refuse (SDR) improves the model's accuracy. IPCC [13, 99] further divided the biodegradable OW into rapidly degrading waste (food waste, sewage sludge), moderately degrading waste (other (non-food) organic putrescible, garden and park waste), slowly degrading waste (paper/textile and wood/straw), and bulk waste (Table 5.1).

Besides, LandGEM does not address the interactions of different waste components interfering with the LFG generation rate. Although sulfate-reducing bacteria help to maintain pH within a reasonable range for methanogenesis (under neutral pH conditions, not acidic ones[100]), methanogens and sulfate-reducing bacteria compete for common substrates (i.e., hydrogen and acetate) to generate CH_4 and H_2S, respectively. According to [10], sulfate-reducing bacteria outcompete acetogens and methanogens for electron equivalents (e.g., hydrogen or organic acids), leading to sulfide production, which is inhibitory and causes odor and corrosion. Low concentrations of H_2S could inhibit the growth of microorganisms and suppress the CH_4 forming processes in the presence of sulphate and sulphate-reducing conditions effectively [100]. In addition to CH_4, H_2S generation is best modeled with a first-order decay equation, similar to LandGEM [70]. Therefore, a first-order kinetic model, similar to LandGEM, could evaluate H_2S generation from sulphur-containing wastes: BCD and FCD. An optimization algorithm, such as a genetic algorithm, could also be implemented to estimate H_2S generation potential, S_0 (m^3 H_2S/t sulphur), and H_2S generation rate, k_{H_2S} (y^{-1}) of BCD and FCD. [150] combined artificial neural networks with genetic algorithm to simulate the gas generation in landfills. They found that artificial neural networks is efficient in providing accurate short-term predictions. At the same time, the genetic algorithm can generate a precise model of a landfill for long-term forecasting and planning. The genetic algorithm can navigate large complex search spaces to deliver near-optimal solutions [151].

[101] used the LandGEM model to calculate landfill emissions and CH_4 generation, which uses one type of waste (i.e., MSW) for the whole landfill and the studied period. The model used a first-order decay equation in which CH_4 generation depended on the CH_4 generation rate (k_{CH_4}), CH_4 generation potential (L_0), and the mass and age of waste. They considered the range of k_{CH_4} and L_0 between 0.02 to 0.70 y^{-1} and 96 to 170 m^3 CH_4 per ton of MSW, respectively, and concluded that for 400 t MSW per day landfilled, 652,836 t CH_4 generated from 1984 to 2124, assuming the value of k_{CH_4} and L_0 equal to 0.70 y^{-1} and 170 m^3 CH_4 per ton of MSW, respectively. Based on their study, each ton of MSW generated 0.03 tons of CH_4 (42 m^3 CH_4 considering CH_4 density to be 0.7157 kg/m^3). A study by [152] estimated CH_4 generation in a landfill from 2010 to 2060, considering the value of k_{CH_4} and L_0 equal to 0.05 y^{-1} and 110 m^3 CH_4 per ton of MSW, respectively. According to their study, the least and the most CH_4

generated were 256,000 m^3 in 2010 (receiving 30,768 tons MSW or 84 t MSW per day) and 16,600,000 m^3 in 2042 (receiving 3,126,706 tons MSW or 8,566 t MSW per day). Hence, each ton of MSW generated 8.3 (2010) to 5.3 (2042) m^3 CH_4. Another study [102] applied the LandGEM model, assuming k_{CH_4} to be 0.09 y^{-1} and L_0 between 18 to 138 m^3/t MSW and achieved higher LFG generation with higher CH_4 generation potential. Their results confirmed the dependence of landfill environmental impacts of waste composition, particularly the amount of biodegradable OW. [103] compared the LandGEM model results with other models and proved this model's better ability to estimate GHG emissions from MSW landfills.

Various studies (e.g., [97, 153-155]) used LandGEM to estimate CH_4 generation from landfills, yet they did not consider waste characterization, parameter fitting of k_{CH_4} and L_0, and applying a genetic algorithm in their work. The same research gap exists for H_2S generation modeling even further as it has been rarely addressed in the literature. Waste characterization, parameter fitting of k_{H_2S} and S_0, and applying a genetic algorithm can not be found in other studies (e.g., [156, 157]).

Using a genetic algorithm, this study fits parameters to a CH_4 and H_2S generation model according to a modified first-order decay model. Model validation was done using the LFG collection data from a landfill site in the province of Quebec, Canada. The data contained thirty-nine years of measurements of OW, BCD and FCD quantities and twenty-four years of LFG amounts, and was used to evaluate the performance of first-order decay models to estimate CH_4 and H_2S generation. In the case of CH_4 generation modeling, food waste (FDR), yard waste (FDR), paper (SDR), and wood (SDR) were assumed to address OW segregation. In addition to optimizing the OW fractions, key modeling parameters of OW (k_{CH_4} and L_0) were determined independently for periods in the life of a landfill. Similarly, for H_2S generation modeling, the CD was classified into FCD and BCD, and k_{H_2S} and S_0 of BCD and FCD were determined. A range of scenarios were analyzed, including two benchmark and twelve optimizing scenarios for CH_4 and two scenarios for H_2S modeling.

Table 5.1: Waste segregation and corresponding k_{CH_4} values, biodegradation half-life in y^{-1} for OW in a landfill, based on IPCC recommended ranges (adapted from IPCC [13]).

Waste type		Climate zone							
		Boreal and temperate (MAT≤20°C)				Tropical (MAT>20°C)			
		Dry (MAP/PET<1)		Wet (MAP/PET>1)		Dry (MAP<1000 mm)		Moist and wet (MAP≥1000 mm)	
		Default	Range	Default	Range	Default	Range	Default	Range
Slowly degrading waste	Paper/textiles waste	0.04	0.03-0.05	0.06	0.05-0.07	0.045	0.04-0.06	0.07	0.06-0.085
	Wood/straw waste	0.02	0.01-0.03	0.03	0.02-0.04	0.025	0.02-0.04	0.035	0.03-0.05
Moderately degrading waste	Other (non – food) organic putrescible/ garden and park waste	0.05	0.04-0.06	0.1	0.06-0.1	0.065	0.05-0.08	0.17	0.15-0.2
Rapidly degrading waste	Food waste/sewage sludge	0.06	0.05-0.08	0.185	0.1-0.2	0.085	0.07-0.1	0.4	0.17-0.7
Bulk waste		0.05	0.04-0.06	0.09	0.08-0.1	0.065	0.05-0.08	0.17	0.15-0.2

MAT: Mean Average Temperature. MAP: Mean Average Precipitation. PET: Potential Evapotranspiration.

The novelty of this study is that it differentiates the OW and CD into four and two types to estimate CH_4 and H_2S generation, respectively, and enhance modeling accuracy. Additionally, it applies a genetic algorithm to fit various parameters, which has never been done in the literature. The methodology could be used in other landfills using their waste characterization and gas collection data.

5.2 Methodology

5.2.1 Landfilled mass and landfill gas collection trend

The studied landfill was in a wet boreal climate in the province of Quebec, Canada. The landfill had six landfilled sectors, namely 1, 2, 3, 4, 5, and 6, in which only landfilled OW, BCD, and FCD were considered (Table 5.2). Sectors 1, 2, 3, and 4 only received OW, but sectors 5 and 6 received OW and CD. Sector 5 of the landfill received BCD and FCD, and sector 6 received only BCD (Figure 5.1 (a and b) in relative values as they are the confidential data of this landfill). Although waste quantities landfilled could be found for the entire site lifetime, waste composition data were unavailable.

Figure 5.1 (c and d) shows the amount of generated CH_4 from all the sectors during thirty-nine years and H_2S from sectors 5 and 6 during eleven years in relative values as they are the confidential data of this landfill. It was assumed that CH_4 generation is associated with OW and H_2S generation with BCD and FCD. LFG flow and CH_4 concentration were measured automatically by onsite flowmeters and infrared analyzers. H_2S concentration was measured by the electrochemical analyzer. Micro 3000A, manufactured by Agilent, measured both CH_4 and H_2S. Also, 62-9/9500 flowmeter, manufactured by Thermal Instrument, was used. Data were recorded daily by the landfill operator and were available monthly. Monthly data was compiled annually for this study.

Societal changes in landfilling practices resulting from stricter legislation in Quebec (enhancements in recycling, higher raw material value, etc.) led to considering the subdivision of the landfill's lifetime into three distinct periods – Periods 1, 2, and 3 – reflecting the specific history of refuse admittance based on changes in waste characteristics. For instance, Quebec targeted the recovery ratio for recyclables, OW, and construction and demolition waste to be 70%, 60%, and 70%, respectively, and the province aims to increase bioenergy production by 50% through various methods such as bio-methanation of OW by 2030 ([113, 117]). Different optimization scenarios were posed in which the variables were time-independent (constant in periods) or time-dependent.

5.2.2 First-order decay model

The first-order kinetic equation, LandGEM [96], was applied to evaluate CH_4 generation from OW (Eq. 5.1). And a first-order kinetic equation, similar to LandGEM, was used to estimate H_2S generation from BCD and FCD (Eq. 5.2) [156].

$$Q_{CH_4} = \sum_{i=1}^{n} \sum_{j=0.1}^{1} k_{CH_4} \, L_0 \left(\frac{OW_i}{10}\right) e^{-k_{CH_4} t_{i,j}} \qquad \textbf{Eq. 5.1}$$

where Q_{CH_4} (*m*^{3}*/y*) is the annual CH_4 generation after *n* years, L_0 (*m*^{3} *CH_4/t biodegradable waste*) is CH_4 generation potential from biodegradable waste, k_{CH_4} (y^{-1}) is the CH_4 generation rate, OW_i (*t*) is the quantity of biodegradable OW landfilled in year *i*, $t_{i,j}$ is the age of the *j*th section of landfilled OW at the

i^{th} year, j = 0.1 year time increment and n is the number of years calculated (year of calculation - initial year of waste acceptance).

$$Q_{H_2S} = \sum_{i=1}^{n} \sum_{j=0.1}^{1} k_{H_2S} \, S_0 \left(\frac{CD_i}{10}\right) e^{-k_{H_2S} t_{i,j}} \qquad \textbf{Eq. 5.2}$$

where Q_{H_2S} (m^3/y) is the annual H_2S generation after n years, S_0 (m^3 H_2S/t *sulphur*) is H_2S generation potential, k_{H_2S} (y^{-1}) is the H_2S generation rate, CD_i is the quantity of CD landfilled in year i (t), $t_{i,j}$ is the age of the j^{th} section of landfilled sulphur at the i^{th} year, j = 0.1 year time increment and n is the number of years calculated (year of calculation - initial year of waste acceptance).

Table 5.2: Landfill sectors (1 to 6), landfilled wastes (OW, BCD and FCD) and landfilling years in each sector.

Sector	1	2	3	4	5	6
Waste type	OW	OW	OW	OW	OW, BCD, FCD	OW, BCD
Years	4-13	0-3, 13-14	14-22	22-27	28-34 (OW), 27-35 (BCD & FCD)	35-46 (OW & BCD)

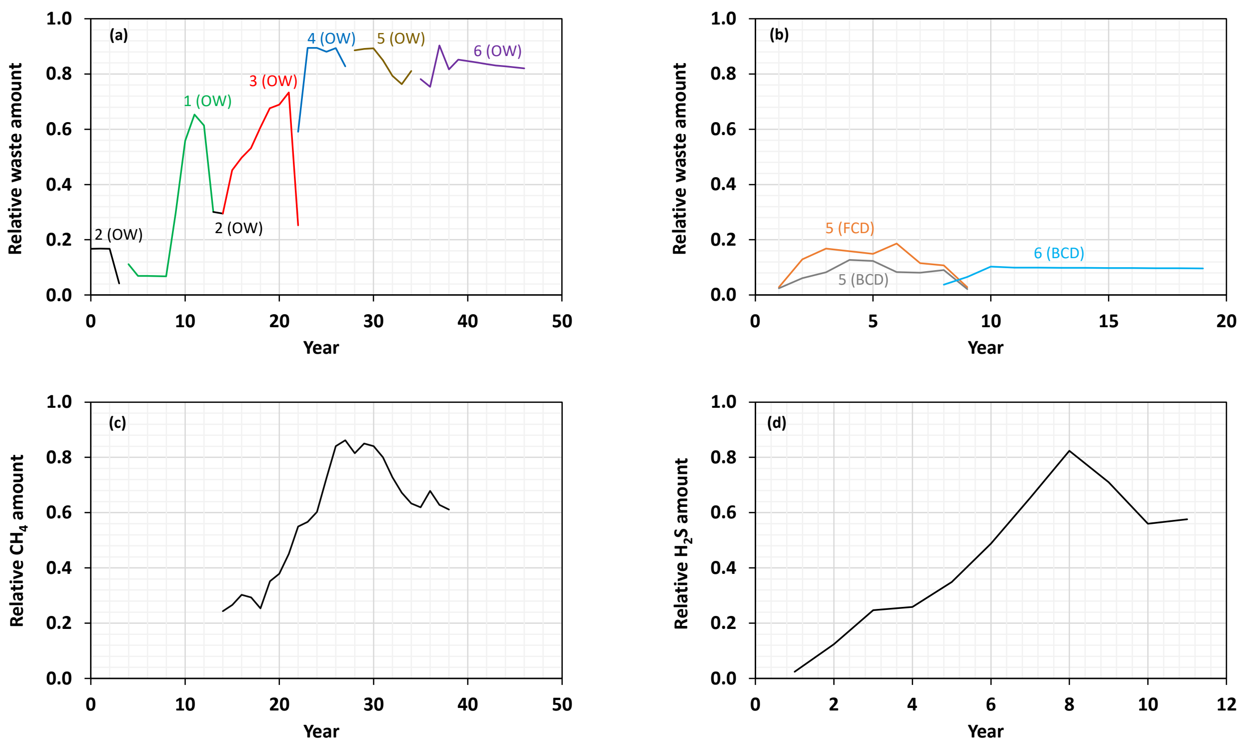


Figure 5.1: (a) Landfilled waste (OW) from sectors 1, 2, 3, 4, 5, and 6, (b) Landfilled waste (BCD and FCD) from sectors 5 and 6, (c) Measured CH_4 from all the sectors, and (d) Measured H_2S from sectors 5 and 6, all in relative values due to data confidentiality.

5.2.3 Parameter fit

A genetic algorithm optimization aims to fit the generated CH_4 or H_2S data with modeled ones. The generated data was obtained from the collected data using a collection efficiency of around 92% (collected gas divided by generated gas). Hence, it was implemented to estimate various parameters, such as L_0 (*m^3 CH_4/t biodegradable waste*) and k_{CH_4} (*y^{-1}*) of OW, and the fraction of each OW type (*1*) for CH_4 modeling, and S_0 (*m^3 H_2S/t sulphur*), and k_{H_2S} (*y^{-1}*) of BCD and FCD for H_2S modeling, based on the modeling scenarios. The objective was to minimize the residual sum of squares (RSS) of estimation between two sets of data (Eq. 5.3) [158].

$$RSS = \sum_{i=1}^{n} \left(y_{g_i} - y_{m_i}\right)^2 \qquad \textbf{Eq. 5.3}$$

where y_{g_i} is the generated value, y_{m_i} is the modeled value of CH_4 or H_2S, and *n* is the number of years in the optimization.

5.2.4 Modeling scenarios

CH_4 modeling

This study proposes distinguishing seven categories of landfill waste herein: food waste, sludge, paper, yard waste, wood, textile, and other OW (Table 5.3). However, for the parameter fits, only four categories were considered: 1) food waste, 2) yard waste, 3) paper, and 4) wood. The reason is to decrease the optimization variables and categorizing the biodegradable OW into different types: easily (e.g., food waste), slowly (e.g., paper), and hardly (e.g., wood, textiles, and leather) biodegradable wastes. k_{CH_4} is the biodegradation half-life in y^{-1} for OW in a landfill and, based on the U.S. EPA, can range between 0.02 *y^{-1}* (less than 635 mm of precipitation) and 0.04 *y^{-1}* (more than 635 mm of precipitation) [159]. IPCC reported various ranges of k_{CH_4} (0.01 *y^{-1}* to 0.70 *y^{-1}*) for different climatic conditions [13, 99]. L_0 depends on the type of waste deposited and some landfilling conditions described below. It can range vastly between 6-270 *m^3 CH_4/t* MSW [160]. [75] reported L_0 and k_{CH_4} of MSW to be 78 *m^3 CH_4/t* and 0.0427 *y^{-1}*, respectively.

Table 5.3 shows the waste segregation, k_{CH_4} and L_0 in the literature and the ones considered in this study. In the case of L_0, the minimum and maximum L_0 values were also calculated based on the mass of degradable organic carbon (DOC) following IPCC recommendations for each kind of waste (Eq. 5.4 and Eq. 5.5) [13, 99].

$$DDOC_m = W \times DOC \times DOC_f \times MCF \qquad \textbf{Eq. 5.4}$$

where $DDOC_m$ (*t*) is the mass of decomposable DOC deposited, W (*t*) is the mass of waste deposited, DOC (*t carbon/t waste*) is the degradable organic carbon in the year of deposition, DOC_f (*1*) is the fraction of DOC that can decompose, and MCF is the *CH_4* (*1*) correction factor for aerobic decomposition in the year of deposition (1 [13]).

$$L_0 = \frac{DDOC_m \times F \times 16/12}{W} \qquad \textbf{Eq. 5.5}$$

where L_0 (*t CH₄/t waste*) is the CH_4 generation potential, $DDOC_m$ (*t*) is the mass of decomposable DOC deposited, F (*1*) is the fraction of CH_4 in generated LFG, and 16/12 is the molecular weight ratio of CH_4/C. CH_4 density was assigned to 0.554×10^{-3} t/m^3.

The landfill's lifetime was subdivided into three periods reflecting the specific history of refuse admittance based on changes in waste characteristics. Accordingly, various scenarios were defined to improve the fitting of the CH_4 generation model to real generated data using the genetic algorithm. In this study, the optimization of first-order kinetic equation coefficients (k_{CH_4} and L_0) and the proportions of the different types of waste are considered to best fit the measured LFG using a genetic algorithm in MATLAB. The purpose of this study was not to compare different numerical optimization methods or CH_4 generation models. The objective was to compare different scenarios of modelization using different types of waste, different values of k_{CH_4} and L_0, different periods of landfilling and different proportions of waste. Yet, a benchmark study showed that a genetic algorithm performs better and is faster, and LandGEM is well known to be a reliable first-order decay model and was therefore considered an excellent candidate for this study.

Scenarios were divided into two series: benchmark and optimizing. Two benchmark scenarios with no optimization were considered for testing the model's superiority. These scenarios assumed one type of waste (MSW), one periodic waste fraction, and considered the entire landfill as one sector. The first benchmark scenario assumed k_{CH_4}=0.70 y^{-1} and L_0=170 $m^3\ CH_4\ t^{-1}$ *MSW* [101] and the second one assumed k_{CH_4}=0.05 y^{-1} and L_0=110 $m^3\ CH_4\ t^{-1}$ *MSW* [152].

As shown in Table 5.4 and Table 5.5, twelve optimizing scenarios were determined, from which some scenarios (4 to 6 and 10 to 12) considered different variables for each sector. In contrast, other scenarios neglected such variation and assumed the whole landfill as one sector. Scenario 1, namely Sc1_2WT_1PWF_1S, had 5 variables, including k_{CH_4} and L_0 for FDR and SDR, in addition to one periodic waste fraction for these waste types. This scenario neglected variation in time and assumed one periodic waste fraction for FDR and SDR throughout the landfill lifetime. Scenario 2, Sc2_2WT_3PWF_1S, had 7 variables as it considered variation in time and had three periodic waste fractions. Scenario 3, Sc3_2WT_CPWF_1S, assumed constant three periodic waste fractions throughout the landfill lifetime analysis and has 4 variables. The subsequent three scenarios (Sc4_2WT_1PWF_6S, Sc5_2WT_3PWF_6S, and Sc6_2WT_CPWF_6S) were the same as scenarios 1, 2, and 3, except that they considered different variables for each sector, and had 30, 42, and 20 variables, respectively. Scenarios 7 to 12 were the same as scenarios 1 to 6, except four waste types (food waste, yard waste, paper, and wood) were considered leading to 11, 17, 8, 66, 102 and 48 variables, respectively. Table 5.6 illustrates CH_4 modeling optimization variables in each scenario. This study applies to well-documented waste management landfills with accurate waste characterization data. Otherwise, the level of error given by the lack of waste characterization overshadows the effort this study is trying to produce.

Table 5.3: Lower bound and upper bound of k_{CH_4} (*y^{-1}*) and L_0 (*m^3 CH_4/t biodegradable waste*) for optimizing scenarios of CH_4 modeling.

Waste type		k_{CH_4}			L_0				
		[13]	[161]	This study	[13]	[160]	[162]	[163]	This study
FDR	**Food**	0.17-0.70	0.185	0.17-0.70	77-192	-	12-248	45-301	12-301
	Sludge	0.17-0.70	0.185	0.17-0.70	38-48	-	23-230	19-70	19-230
	Yard	0.15-0.20	0.10	0.10-0.20	173-211	63-104	104	31-136	21-211
SDR	**Paper**	0.06-0.085	0.06	0.06-0.085	247-309	57-194	66-387	67-296	57-387
	Wood[1]	0.03-0.05	0.03	0.03-0.05	53-63	15-24	-	86-130	15-241
	Textile	0.06-0.085	0.06	0.06-0.085	137-274	172-191	189-216	73-216	105-274
	Other OW	0.15-0.20	0.10	0.10-0.20	82-192	-	22-150	75-109	22-241

[1] Wood of construction and demolition waste

Table 5.4: Number, name, and description of optimizing scenarios for CH_4 modeling.

Scenario	Name	Description
1	Sc1_2WT_1PWF_1S	two waste types, one periodic waste fraction, one sector for the whole landfill
2	Sc2_2WT_3PWF_1S	two waste types, three periodic waste fractions, one sector for the whole landfill
3	Sc3_2WT_CPWF_1S	two waste types, constant three periodic waste fractions, one sector for whole landfill
4	Sc4_2WT_1PWF_6S	two waste types, one periodic waste fraction, six sectors
5	Sc5_2WT_3PWF_6S	two waste types, three periodic waste fractions, six sectors
6	Sc6_2WT_CPWF_6S	two waste types, constant three periodic waste fractions, six sectors
7	Sc7_4WT_1PWF_1S	four waste types, one periodic waste fraction, one sector for the whole landfill
8	Sc8_4WT_3PWF_1S	four waste types, three periodic waste fractions, one sector for the whole landfill
9	Sc9_4WT_CPWF_1S	four waste types, constant three periodic waste fractions, one sector for whole landfill
10	Sc10_4WT_1PWF_6S	four waste types, one periodic waste fraction, six sectors
11	Sc11_4WT_3PWF_6S	four waste types, three periodic waste fractions, six sectors
12	Sc12_4WT_CPWF_6S	four waste types, constant three periodic waste fractions, six sectors

Sc: optimizing scenario, WT: waste type, PWF: periodic waste fraction, S: sector.

Table 5.5: Lower bound and upper bound of k_{CH_4} (*y^{-1}*) and L_0 (*m^3 CH_4 t^{-1}*) and periodic waste fraction (-) for optimizing scenarios of CH_4 modeling, (a) FDR and SDR, (b) food, yard, paper, and wood.

(a)

Optimization variables	k_{CH_4}	L_0	k_{CH_4}	L_0	PWF_FDR		
Waste type/period	*FDR*	*FDR*	*SDR*	*SDR*	*1*	*2*	*3*
Default							
Sc1 & Sc 4	0.70	62.86	0.03	62.86	0.10	-	-
Sc2, Sc 3, Sc 5 & Sc 6						0.60	0.30
Lower Bound							
Sc1 & Sc 4	0.10	12.00	0.03	15.00	0.10	-	-
Sc2, Sc 3, Sc 5 & Sc 6						0.30	0.30
Upper Bound							
Sc1 & Sc 4	0.70	300.70	0.20	387.00	0.40	-	-
Sc2, Sc 3, Sc 5 & Sc 6						0.60	0.60

Sc: optimizing scenario, PWF: periodic waste fraction.

(b)

Optimization variables	k_{CH_4}	L_0	k_{CH_4}	L_0	k_{CH_4}	L_0	k_{CH_4}	L_0	PWF_food			PWF_yard			PWF_paper		
Waste type/period	Food	Food	Yard	Yard	Paper	Paper	Wood	Wood	*1*	*2*	*3*	*1*	*2*	*3*	*1*	*2*	*3*
Default																	
Sc7 & Sc 10	0.70	62.86	0.15	62.86	0.03	62.86	0.03	62.86	0.10	0.10	0.10	-	-	-	-	-	-
Sc8, Sc 9, Sc 11 & Sc 12												0.30	0.30	0.30	0.15	0.15	0.15
Lower Bound																	
Sc7 & Sc 10	0.17	12.00	0.10	21.00	0.06	57.00	0.03	15.00	0.10	0.10	0.10	-	-	-	-	-	-
Sc8, Sc 9, Sc 11 & Sc 12												0.15	0.15	0.15	0.15	0.15	0.15
Upper Bound																	
Sc7 & Sc 10	0.70	300.70	0.20	211.00	0.09	387.00	0.05	241.00	0.40	0.40	0.40	-	-	-	-	-	-
Sc8, Sc 9, Sc 11 & Sc 12									0.20	0.20	0.20	0.30	0.30	0.30	0.30	0.30	0.30

Sc: optimizing scenario, PWF: periodic waste fraction.

Table 5.6: CH_4 modeling optimization variables in each optimizing scenario.

Scenario	WT												PWF		Number of sectors		Maximum total of variables
	FDR		SDR		Food		Yard		Paper		Wood		Three	One	Six	One	
	k_{CH_4}	L_0	k_{CH_4}	L_0	k_{CH_4}	L_0	k_{CH_4}	L_0	k_{CH_4}	L_0	k_{CH_4}	L_0					
Sc1_2WT_1PWF_1S	☑	☑	☑	☑										☑(1PWF×1WT)		☑	5×1
Sc2_2WT_3PWF_1S	☑	☑	☑	☑									☑(3PWF×1WT)			☑	7×1
Sc3_2WT_CPWF_1S	☑	☑	☑	☑									Constant waste fraction			☑	4×1
Sc4_2WT_1PWF_6S	☑	☑	☑	☑										☑(1PWF×1WT)	☑		5×6
Sc5_2WT_3PWF_6S	☑	☑	☑	☑									☑(3PWF×1WT)		☑		7×6
Sc6_2WT_CPWF_6S	☑	☑	☑	☑									Constant waste fraction		☑		4×6
Sc7_4WT_1PWF_1S					☑	☑	☑	☑	☑	☑	☑	☑		☑(1PWF×3WT)		☑	11×1
Sc8_4WT_3PWF_1S					☑	☑	☑	☑	☑	☑	☑	☑	☑(3PWF×3WT)			☑	17×1
Sc9_4WT_CPWF_1S					☑	☑	☑	☑	☑	☑	☑	☑	Constant waste fraction			☑	8×1
Sc10_4WT_1PWF_6S					☑	☑	☑	☑	☑	☑	☑	☑		☑(1PWF×3WT)	☑		11×6
Sc11_4WT_3PWF_6S					☑	☑	☑	☑	☑	☑	☑	☑	☑(3PWF×3WT)		☑		17×6
Sc12_4WT_CPWF_6S					☑	☑	☑	☑	☑	☑	☑	☑	Constant waste fraction		☑		8×6

Sc: optimizing scenario, WT: waste type, PWF: periodic waste fraction.

H_2S modeling

Sulphur content in MSW is about 0.1% (corrugated boxboard: 0.14%, newspaper: 0.11%, mix paper: 0.12%, food waste: 0.25%, grass + dirt: 0.26%, plastic film: 0.07%, plastics, rubber, leather mix: 0.55%, sewage sludge digested: 0.66%, textiles: 0.20%, wood: 0.11%, glass, ceramics: 0.00%, and metals: 0.01%) [68]. However, CD contains around 1.5-9.1% sulphate [69] and 2.08% of CD, and 5.15% of earth and sand construction waste is sulphur [164]. Therefore, estimating H_2S generation in landfills containing the CD is important. H_2S is an odorous gas that negatively impacts neighbouring 'populations' health and well-being [165]. The modeling scenarios in this study considered two variables (k_{H_2S} and S_0) for BCD and FCD in each sector. Table 5.7 shows the optimization variables and their lower and upper bounds. Although the range of k_{CH_4} and L_0 values for CH_4 modeling exist in the literature, those corresponding to H_2S are relatively much less known. [70] evaluated H_2S generation from nine U.S. northeastern CD landfills and obtained $k_{H_2S} = 0.50 - 0.88\ y^{-1}$. This study identified some empirical information of the same type through trial and error. As presented in Table 5.8, the number of variables was 4 for scenario 1 and 6 for scenario 2, and both used a genetic algorithm.

5.3 Results and Discussion

5.3.1 CH_4 modeling

Figure 5.2 shows the results of two benchmark scenarios. The unrealistic results of these studies indicate the necessity of considering different waste types, various periodic waste fractions, and several landfill sectors. The first benchmark scenario had a higher k_{CH_4} (0.70 y^{-1}) and L_0 (170 m^3 CH_4 t^{-1} MSW) than the second one (k_{CH_4}=0.05 y^{-1} and L_0=110 m^3 CH_4 t^{-1} *MSW*), yet both of them led to significantly high RSS values (198,439 for the first and 49,110 for the second).

Figure 5.3 illustrates the relative measured and modeled total CH_4 generation of optimizing scenarios 1 to 12. Based on the figures, the highest RSS values were obtained for scenarios 1 (two waste types, one periodic waste fraction, one sector for the whole landfill, RSS: 7,709) and 7 (four waste types, one periodic waste fraction, one sector for the whole landfill, RSS: 7,659), and the lowest RSS values were obtained for scenarios 4 (two waste types, one periodic waste fraction, six sectors, RSS: 785) and 12 (four waste types, constant three periodic waste fraction, six sectors, RSS: 676).

Table 5.7: Lower bounds and upper bounds of H_2S modeling optimization variables for Sectors 5 and 6.

	Sector	FCD	BCD
Optimization range for k_{H_2S} (y^{-1})	5	0.10-0.90	0.10-0.90
	6	N.A.	0.10-0.90
Optimization range for S_0 (m^3 H_2S t^{-1})	5	0.04-7	0.01-10
	6	N.A.	0.01-10

Table 5.8: H_2S modeling optimization variables in each scenario.

Scenario	Sector	FCD		BCD		No. of variables
		k_{H_2S}	S_0	k_{H_2S}	S_0	
1	5	☐[1]	☐	☑[2]	☑	4
	6	N.A.[3]	N.A.	☑	☑	
2	5	☑	☑	☑	☑	6
	6	N.A.	N.A.	☑	☑	

[1] Equal to that of BCD, [2] Optimization variable, [3] Not applicable since Sector 6 did not contain FCD.

Scenarios 1 and 7 considered the whole landfill as one sector, not six sectors, for only one periodic waste fraction. Their high RSS value reveals they do not coincide with measurement and need refining. A comparison of scenarios 2 (two waste types, three periodic waste fractions, one sector for the whole landfill, RSS: 1,550) and 3 (two waste types, constant three periodic waste fractions, one sector for the whole landfill, RSS: 1,541) shows that the initial input for the waste fraction in scenario 3 was very close to reality, leading to negligible improvement by optimization. These periodic waste fractions were taken from internal reports. Scenario 4 (two waste types, one periodic waste fraction, six sectors, RSS: 785) has the lowest RSS for the two-type waste modeling series. It indicates the importance of considering sectors individually in the analysis. In scenarios 5 (two waste types, three periodic waste fractions, six sectors, RSS: 870) and 6 (two waste types, constant three periodic waste fractions, six sectors, RSS: 786), although multiple variables have been added, RSS is greater than the RSS of scenario 4. Since different sectors are filled with waste at different times, considering six sectors already indicates that we have six periods of time. Hence, assuming three additional periods is not necessary. Similar to the two-type waste modeling series, scenarios 8 (four waste types, three periodic waste fractions, one sector for the whole landfill, RSS: 1,494) and 9 (four waste types, constant three periodic waste fractions, one sector for the whole landfill, RSS: 1,801) were better than scenario 7; however, the constant periodic waste fraction didn't decrease the RSS of scenario 9. Again, similar to the two-type waste modeling series, the impact of considering individual sectors improved the modeling results for scenarios 10 (four waste types, one periodic waste fraction, six sectors, RSS: 853), 11 (four waste types, three periodic waste fractions, six sectors, RSS: 707), and 12 (four waste types, constant three periodic waste fractions, six sectors, RSS: 676). Among these scenarios, scenario 12 had the lowest RSS, indicating the importance of waste segregation and analyzing sectors individually. The slight difference in RSS of scenarios 11 and 12 indicates that considering periods seems unnecessary. Finally, it can be concluded that waste segregation improved the modeling accuracy.

Optimized k_{CH_4} (y^{-1}), L_0 ($m^3\ CH_4\ t^{-1}$), and periodic waste fractions (%) of optimizing scenarios 1 to 12 are shown in Table 5.9. k_{CH_4} values of scenario 4 (two waste types, one periodic waste fraction, six sectors), as the best scenario among the two-type waste modeling series, ranged from 0.24 to 0.70 y^{-1} for FDR and 0.03 to 0.17 y^{-1} for SDR. L_0 values of this scenario ranged from 39.52 to 190.31 $m^3\ CH_4\ t^{-1}$ for FDR and 62.27 to 167.39 $m^3\ CH_4\ t^{-1}$ for SDR. Also, the FDR fraction changed from 11% for sector 1 to 34% for sector 6. k_{CH_4} values of scenario 12 (four waste types, constant three periodic waste fractions, six sectors), as the best scenario among the four-type waste modeling series, ranged from 0.17 to 0.68 y^{-1} for food, 0.11 to 0.18 y^{-1} for yard, 0.06 to 0.08 y^{-1} for paper, and 0.03 to 0.05 y^{-1} for wood. L_0 values of this scenario ranged from 33.18 to 297.24 $m^3\ CH_4\ t^{-1}$ for food, 38.68 to 208.26 $m^3\ CH_4\ t^{-1}$ for yard, 97.82 to 368.73 $m^3\ CH_4\ t^{-1}$ for paper, and 25.53 to 201.18 $m^3\ CH_4\ t^{-1}$ for wood. MSW contains inerts, and hence, the L_0 of MSW should be lower than the L_0 of biodegradable waste considered in this study.

One of the limitations of this study is that the effect of LFG collection efficiency variation was not considered. Hence, it could be optimized within the range of 0.75-0.95 and added to the best scenarios from 1 to 12 based on their results for future studies. Moreover, the waste characterization data is required to model the gas generation. However, landfills usually lack such data and occasionally conduct waste characterization studies. In addition, the interaction of H_2S and CH_4 generation was not considered.

5.3.2 H_2S modeling

Figure 5.4 shows the relative measured and modeled total H_2S generation of scenarios 1 and 2. Both scenarios estimated the measurement values well and had an almost similar RSS value. Scenario 2 with six variables had a RSS value equal to 1,027, which was 1,049 for scenario 1. Optimized k_{H_2S} (y^{-1}) and

S_0 (m^3 H_2S t^{-1}) of these scenarios are shown in Table 5.10. Optimized k_{H_2S} of FCD and BCD is 0.10 y^{-1} for sector 5 in both scenarios and optimized k_{H_2S} of BCD is 0.89 and 0.10 y^{-1} for sector 6 in scenarios 1 and 2, respectively. The higher k_{H_2S} of BCD (0.89 y^{-1}) led to a lower S_0 (1.50 m^3 H_2S t^{-1}). [166] reported $k_{H_2S} = 0.50 - 0.88\, y^{-1}$ from nine U.S. northeastern CD landfills. Optimized S_0 of FCD and BCD is 15.98 m^3 H_2S t^{-1} for sector 5 in scenario 1, and 3.36 and 35.91 for scenario 2, respectively. In addition, the optimized S_0 of BCD is 4.11 m^3 H_2S t^{-1} for sector 6 in scenario 2. Further data could improve H_2S generation modeling.

5.4 Conclusion

Fitting parameters was done in this paper to a CH_4 and H_2S generation model by applying a genetic algorithm based on a modified first-order decay model. To predict CH_4 generation, two benchmark and twelve optimizing scenarios were considered. The benchmark scenarios did not consider any change in modeling parameters, such as waste type, periodic waste fraction or landfill sectors. These scenarios led to high residual sum of square (RSS) values. Hence, applying the LandGEM model should be done by considering different parameters to approach the actual condition of landfill gas (LFG) generation in landfills.

In addition to benchmark scenarios, twelve optimizing scenarios were considered. Scenarios 1 to 6 divided the organic waste (OW) into fast decaying refuse (FDR) and slow decaying refuse (SDR). Scenarios 7 to 12 assumed four types of OW: food waste, yard waste, paper, and wood. In all the scenarios, the OW fractions, CH_4 generation potential (L_0), and CH_4 generation rate (k_{CH_4}) were determined. In addition, some scenarios optimized the parameters mentioned for six landfill sectors, while others considered the landfill as one sector. Moreover, in some scenarios, the landfill's lifetime was subdivided into three distinct periods reflecting the specific history of refuse admittance based on changes in waste characteristics. The results showed that the differentiation of more waste types improves the modeling accuracy for CH_4. Scenarios 11 and 12 considered four waste types of six landfill sectors and had the best predictions, proving that waste characterization is a significant factor in gas prediction. Additionally, since different sectors were filled with waste at different times, assuming six sectors in the modeling already indicated six periods. Hence, considering three additional periods for the landfill's lifetime was unnecessary. Finally, all the scenarios that assumed six landfill sectors had a better parameter fit to real data.

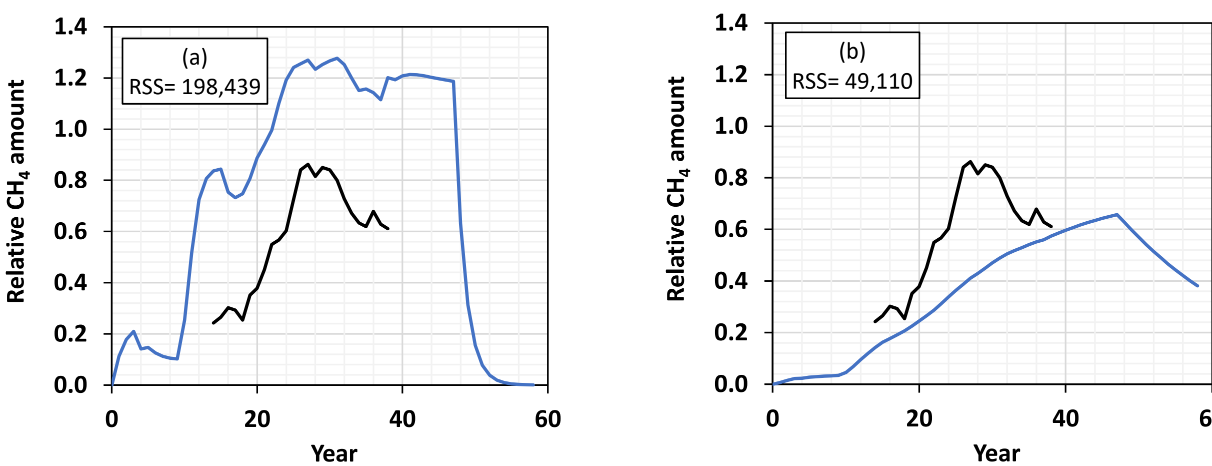


Figure 5.2: Benchmark CH_4 modeling using k_{CH_4} and L_0 of a) [101] and b) [152] in relative values due to data confidentiality (black line is measured and blue line is modeled).

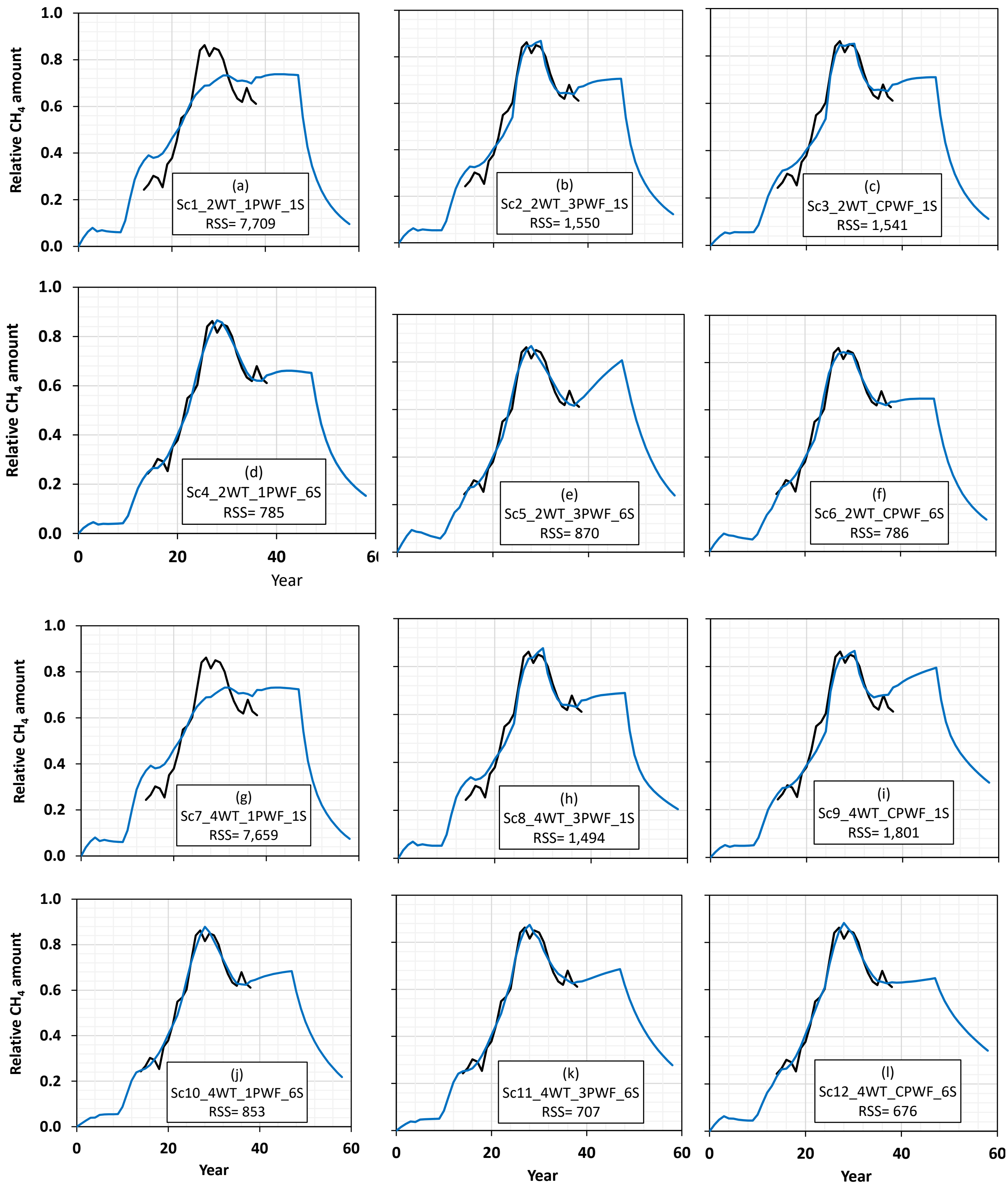


Figure 5.3: Measured (black line) and modeled (blue line) total CH_4 generation of optimizing scenarios 1 to 12 (a to l) in relative values due to data confidentiality.

Table 5.9: Optimized k_{CH_4} (y^{-1}) and L_0 (m^3 CH_4 t^{-1}) and periodic waste fraction (-) for optimizing scenarios of CH_4 modeling, (a) FDR and SDR, (b) food, yard, paper, and wood.

(a)

Optimized variables	k_{CH_4}	L_0	k_{CH_4}	L_0	PWF_1	PWF_2	PWF_3
Waste type	*FDR*	*FDR*	*SDR*	*SDR*	*FDR*	*FDR*	*FDR*
Sc1_2WT_1PWF_1S	0.70	243.25	0.15	78.23	0.15	-	-
Sc2_2WT_3PWF_1S	0.70	124.10	0.12	91.74	0.22	0.60	0.31
Sc3_2WT_CPWF_1S	0.70	108.39	0.13	98.16	0.10	0.60	0.30
Sc4_2WT_1PWF_6S	0.67	139.56	0.17	62.27	0.11	-	-
	0.70	97.85	0.03	132.55	0.28	-	-
	0.24	190.31	0.07	148.09	0.20	-	-
	0.41	39.52	0.15	167.39	0.27	-	-
	0.70	130.41	0.06	70.80	0.17	-	-
	0.49	91.52	0.11	77.91	0.34	-	-
Sc5_2WT_3PWF_6S	0.64	206.33	0.04	53.40	0.14	0.57	0.45
	0.59	80.69	0.13	197.47	0.27	0.36	0.33
	0.22	148.11	0.10	108.92	0.29	0.58	0.40
	0.50	72.81	0.15	152.99	0.13	0.35	0.31
	0.32	97.36	0.09	37.92	0.32	0.55	0.50
	0.18	129.82	0.05	185.90	0.31	0.42	0.59
Sc6_2WT_CPWF_6S	0.57	169.44	0.16	40.05	0.10	0.60	0.30
	0.58	266.65	0.10	129.90	0.10	0.60	0.30
	0.65	114.21	0.13	115.58	0.10	0.60	0.30
	0.61	50.58	0.12	222.91	0.10	0.60	0.30
	0.44	86.13	0.12	65.70	0.10	0.60	0.30
	0.39	113.80	0.10	87.18	0.10	0.60	0.30

Sc: optimizing scenario, PWF: periodic waste fraction.

(b)

Optimization variables	k_{CH_4}	L_0	k_{CH_4}	L_0	k_{CH_4}	L_0	k_{CH_4}	L_0	PWF_1			PWF_2			PWF_3		
Waste type	Food	Food	Yard	Yard	Paper	Paper	Wood	Wood	Food	Yard	Paper	Food	Yard	Paper	Food	Yard	Paper
Sc7_4WT_1PWF_1S	0.66	104.86	0.18	68.70	0.08	170.07	0.04	240.95	0.39	0.38	0.38	-	-	-	-	-	-
Sc8_4WT_3PWF_1S	0.69	267.53	0.10	21.03	0.08	75.49	0.05	123.80	0.16	0.12	0.12	0.29	0.30	0.29	0.16	0.30	0.27
Sc9_4WT_CPWF_1S	0.70	220.64	0.20	21.00	0.06	57.00	0.05	158.20	0.10	0.10	0.10	0.30	0.30	0.30	0.15	0.15	0.15
Sc10_4WT_1PWF_6S	0.62	65.49	0.14	63.90	0.07	156.88	0.04	16.49	0.39	0.20	0.35	-	-	-	-	-	-
	0.47	24.75	0.15	99.51	0.06	98.75	0.05	214.31	0.11	0.27	0.15	-	-	-	-	-	-
	0.53	98.08	0.17	161.56	0.07	288.43	0.04	35.92	0.11	0.11	0.39	-	-	-	-	-	-
	0.20	109.55	0.20	166.45	0.07	182.60	0.03	143.42	0.31	0.32	0.31	-	-	-	-	-	-
	0.67	38.25	0.18	41.89	0.07	98.40	0.05	131.30	0.32	0.31	0.37	-	-	-	-	-	-
	0.61	51.40	0.14	112.65	0.06	331.45	0.04	15.38	0.26	0.38	0.12	-	-	-	-	-	-
Sc11_4WT_3PWF_6S	0.66	161.90	0.11	153.87	0.07	346.58	0.04	17.36	0.16	0.10	0.14	0.21	0.17	0.19	0.21	0.29	0.19
	0.55	109.61	0.14	56.65	0.08	122.27	0.03	185.71	0.12	0.16	0.14	0.18	0.22	0.20	0.15	0.21	0.17
	0.36	82.76	0.12	182.47	0.08	336.13	0.05	154.87	0.14	0.13	0.15	0.24	0.19	0.17	0.21	0.21	0.18
	0.44	191.27	0.18	97.57	0.07	72.54	0.05	121.45	0.15	0.16	0.11	0.28	0.23	0.25	0.20	0.20	0.23
	0.38	146.74	0.11	103.24	0.08	92.89	0.03	82.24	0.19	0.20	0.15	0.25	0.27	0.30	0.16	0.15	0.16
	0.67	18.21	0.17	180.26	0.07	299.47	0.04	78.25	0.19	0.18	0.17	0.17	0.17	0.22	0.29	0.16	0.15
Sc12_4WT_CPWF_6S	0.68	179.35	0.12	173.05	0.06	97.82	0.04	83.87	0.10	0.10	0.10	0.30	0.30	0.30	0.15	0.15	0.15
	0.61	297.24	0.18	163.77	0.07	368.73	0.04	83.44	0.10	0.10	0.10	0.30	0.30	0.30	0.15	0.15	0.15
	0.17	172.41	0.11	208.26	0.08	321.69	0.05	201.18	0.10	0.10	0.10	0.30	0.30	0.30	0.15	0.15	0.15
	0.40	105.71	0.17	114.64	0.07	202.09	0.04	173.17	0.10	0.10	0.10	0.30	0.30	0.30	0.15	0.15	0.15
	0.64	33.18	0.12	63.22	0.07	238.09	0.03	25.53	0.10	0.10	0.10	0.30	0.30	0.30	0.15	0.15	0.15
	0.64	54.01	0.12	38.68	0.06	159.64	0.03	167.25	0.10	0.10	0.10	0.30	0.30	0.30	0.15	0.15	0.15

Sc: optimizing scenario, PWF: periodic waste fraction.

For H_2S generation modeling, H_2S generation potential (S_0), and H_2S generation rate (k_{H_2S}) of fines (FCD) and bulky materials (BCD) of the construction and demolition waste (CD) were optimized. Based on the results, both scenarios had an effective prediction with a similar RSS value. If further data could be provided to the model, more improvements could be achieved.

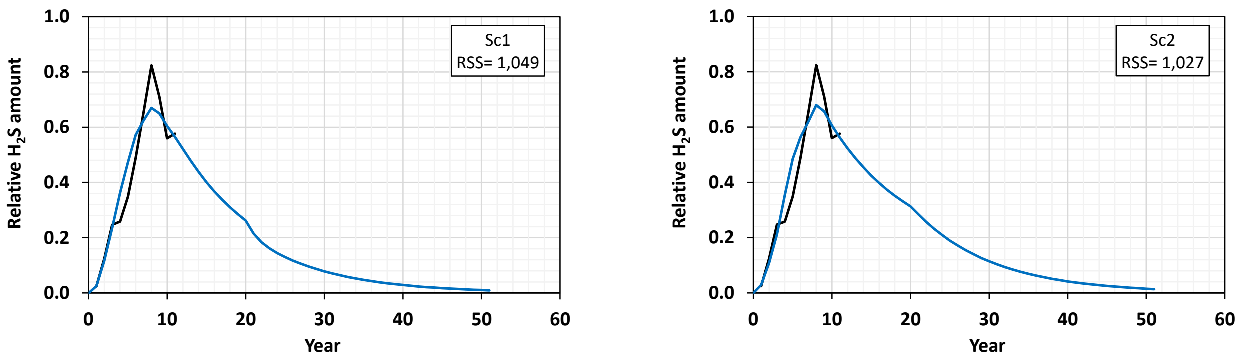


Figure 5.4: Measured (black line) and modeled (blue line) total H_2S generation of scenarios 1 and 2 in relative values due to data confidentiality.

Table 5.10: Optimized k_{H_2S} (y^{-1}) and S_0 (m^3 H_2S t^{-1}) of FCD and BCD for H_2S modeling.

Optimized variables		k_{H_2S}	S_0	k_{H_2S}	S_0
Waste type		*FCD*	*FCD*	*BCD*	*BCD*
Scenario 1	Sector 5	0.10	15.98	0.10	15.98
	Sector 6	-	-	0.89	1.50
Scenario 2	Sector 5	0.10	3.36	0.10	35.91
	Sector 6	-	-	0.10	4.11

Chapter 6. Assessment of landfill gas storage and application regarding energy management: A case study in the province of Quebec, Canada

6.1 Introduction

The worldwide production of millions of tons of solid waste per day is a severe challenge to human life, and landfilling is the most extensively used method of municipal solid waste disposal for operation easiness and cost-effectiveness [167]. In municipal solid waste landfills, a considerable portion of landfill gas (LFG) consists of methane (CH_4) (45-60% [168]). Figure 6.1 illustrates different pathways that exist for generated CH_4 in landfills. In the case of considerable CH_4 generation, LFG can be collected and converted to electricity, heat, or biomethane. A portion of CH_4 in the LFG emits to the atmosphere or oxidizes to CO_2 in the landfill cover. Oxidation process is the reaction of CH_4 with oxygen (O_2) in the landfill cover soil which results in CO_2 generation ($CH_4+2O_2 \rightarrow CO_2+2H_2O$). Landfill cover soil temperatures, soil moisture, and CH_4 soil gas concentrations are environmental factors can drastically affect the oxidation capacity [12].

[13] suggests an oxidation value of 10% for covered well-managed landfill sites to predict diffusion via the cap and escape by cracks or fissures. [14] assumed a single oxidation factor of 10% for managed anaerobic landfills from 1980 to 2004 and average oxidation factors of 14% (2005-2009), 18% (2010-2016), and 22% (2017-2022). Part of the LFG can migrate laterally from the landfill, and the rest could be stored temporarily inside the landfill. Lateral migration could account for 9-18% of the generated CH_4, where 57-79 kg h^{-1} CH_4 is generated from a landfill with an area of approximately 100,000 m^2 receiving around 2.9 million tonnes of mainly non-combustible waste and soil [15].

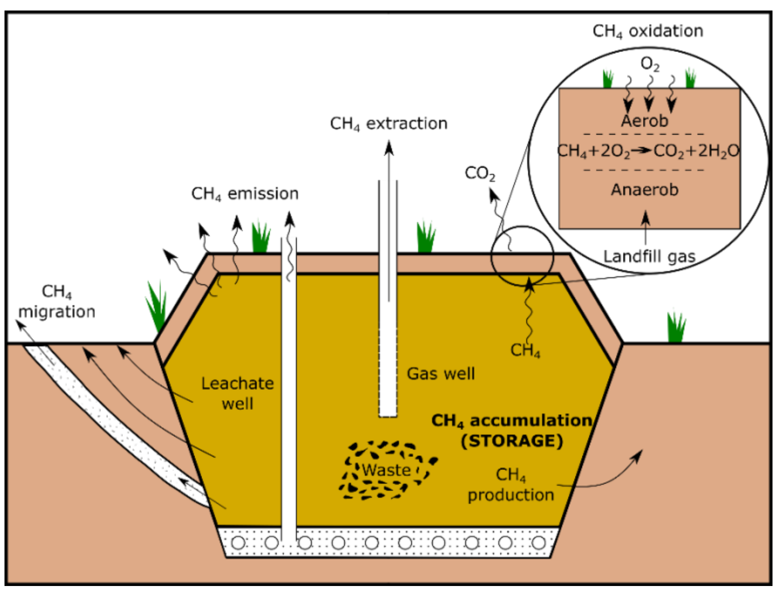


Figure 6.1: CH_4 mass balance and distribution in landfills (adapted from [15]).

The gas storage capacity of landfills depends on several parameters, including size, waste compaction and density, settlement, waste porosity, water content, leachate levels, etc. It can be calculated through a CH_4 mass balance (Eq. 6.1, all units in mass time^{-1}) [15, 19, 159]. Accordingly, the generated CH_4 equals the sum of emitted, oxidized, recovered, migrated and stored CH_4.

$$CH_4\ generation = CH_4\ emitted + CH_4\ oxidized + CH_4\ recovered\ (flared\ or\ valorized) + CH_4\ migrated + \Delta CH_4\ storage \qquad \textbf{Eq. 6.1}$$

[72] reported that, in theory, ΔCH_4 storage could be determined by shutting down the gas collection system for a determined period of time (e.g., one week) and monitoring the CH_4 concentration and LFG flow entering the gas engine before and after restarting the gas collection system. The quantity of additional CH_4 collected after shutdown compared to the prior shutdown approximates ΔCH_4 storage. But, shutting down the gas collection system could pose a risk to the health of citizens and is not recommended [72]. However, shutdowns due to equipment maintenance or seasonal shutdowns of unregulated closed landfills, for example during winter, when operating a gas collection system is challenging, provide an opportunity to study the CH_4 storage capacity inside the landfill.

As mentioned before, shutting down the LFG collection system could be unsafe and is forbidden for regulated active landfills. So, the most difficult term of the landfill CH_4 mass balance is the change in CH_4 storage inside the landfill, which was studied by [19] and [15]. There are two types of ΔCH_4 storage inside the landfill: short-term (hours or days) and long-term (years to decades). In the case of short-term storage, [15] explained that fluctuations in atmospheric pressure or LFG extraction rates lead to short-term CH_4 storage changes. They used ideal gas law assuming a constant gas temperature to calculate the changes in storage volume of the landfill as a function of atmospheric pressure changes [15]. Hence, this method explains that CH_4 storage can change as a function of pressure and temperature, yet temperature inside a landfill can be considered rather isothermic. Waste temperatures rise for months to years until reaching steady elevated conditions compared to ambient ground temperatures [169].

In the case of long-term storage, waste degradation and settlement cause changes in void space and the amount of material available for degradation and gas production lead to changes in CH_4 storage over the long term [15]. To further explain, if there is less LFG generation and the generated LFG is efficiently collected, there will be less LFG stored. In this regard, change in CH_4 storage can be calculated from a change in the CH_4 concentration in the waste, total landfill volume, and the gas-filled porosity of the landfill (Eq. 6.2) [19].

$$\Delta CH_4 = (\Delta\rho) V \varphi_{air} \qquad \textbf{Eq. 6.2}$$

where ΔCH_4 (*kg*) is the change in CH_4 storage, $\Delta\rho$ (*kg/m*^{3}) is the CH_4 concentration changes within the waste, V (*m*^{3}) is the volume of the landfill, and φ_{air} (*1*) is the gas-filled porosity.

It is worth mentioning that the fluctuations in waste humidity influence the gas-filled porosity since LFG emits from liquid and gas phases via the pore spaces in the waste and landfill cover. Once the liquid saturation of the porous medium grows (e.g., because of precipitation, snow melting, etc.), the gas permeability reduces and the liquid permeability rises due to the porosity and permeability heterogeneity [170].

Another study [171] explains that "the rate of change in atmospheric pressure (dP/dt) can cause variability in CH_4 emissions varying several orders of magnitude. Their study assumed that the advective transport mechanism, caused by the vertical pressure difference, is the driving force of LFG from the refuse into the atmosphere through cover soil. If $dP/dt > 0$, this vertical pressure difference will be reduced, and the surface LFG emission will be inhibited. Under these conditions, the constantly generated LFG will be accumulated in the refuse depending on the landfill's gas storage capacity. Otherwise, when $dP/dt < 0$, LFG will be emitted at the surface."

Even though landfills are the most common waste management method worldwide, occupying a considerable space and disturbing other land use up to several kilometres is one of their main drawbacks. Integrating renewable energy generation systems such as photovoltaic power plants on landfill sites can compensate for the space requirement and decrease the costs associated with the grid's electric energy [17]. Moreover, it enhances power usage and flexibility, especially during high-power demand. Several studies combined landfills with photovoltaic power plants [17, 18], but further investigation is required to consider the landfill as a low-cost short-term gas storage system to provide flexible renewable energy.

Based on extensive gas collection data, this study aims to introduce an approach to use the collected stored CH_4 of a closed municipal solid waste landfill in the province of Quebec, Canada, as an energy resource. Accordingly, this study analyzes stored CH_4 that can be recovered by the gas collection system using gas collection data. The LFG collection system of the mentioned landfill experienced mechanical problems, leading to the system's shutdown for some periods. Hence, the periods of the LFG collection system's shutdown and restart were studied. During these periods, CH_4 accumulation, the LFG collection system's vacuum and atmospheric pressure were analyzed. To the authors' best knowledge, this is the first study investigating the collected stored CH_4 of a municipal solid waste landfill using the gas collection data for modulating energy production from LFG for applications such as photovoltaic power plants according to the energy demand.

6.2 Methodology

6.2.1 Landfill gas collection data

The studied landfill is located in a cold and temperate area of Quebec province. It operated for over twenty years and received around 180,000 m^3 of solid waste annually. The waste comprised incinerator ashes and lime sludge (50%), domestic waste (17%), dry materials (13%), cement dust (13%) and pulp and paper residues (7%). The landfill was modern, well designed and maintained, and thus, was properly covered. So, the gas collection efficiency of the landfill, when the collection system was working, was assumed to be more than 90%.

Figure 6.2 shows the schematic of the LFG flow when the gas collection system is on or off. When the LFG collection system is active (Figure 6.2 (a)), the applied vacuum (around -2.0 kPa) extracts the LFG from wells and pipes and it is influenced by the atmospheric pressure. When the LFG collection system is shut down (Figure 6.2 (b)), the LFG accumulates inside the landfill until it reaches the landfill gas storage capacity. At this point the LFG surplus will migrate from the landfill. Meanwhile, some emissions happen before the gas storage capacity is reached.

Three types of LFG collection data were considered as measured at the pumping station: CH_4 flow (m^3/h), LFG flow (m^3/h), and CH_4 concentration (%v/v). LFG flows and CH_4 concentrations were measured every 15 minutes by Sierra Instruments' SteelMass 640S flowmeter (accuracy ±1% of reading plus 0.5% of full scale) and Edinburgh's Guardian Plus CH_4 analyzer (accuracy ±1%). Both instruments were located after the LFG pumps on the main collection conduit. However, only CH_4 flow was studied due to high fluctuations in CH_4 concentration. Nonetheless, LFG flow and CH_4 concentration were used as markers to determine the points that CH_4 flow accumulated and stabilized after restarting the LFG collection system. LFG flow, CH_4 flow, and CH_4 concentration were measured in the range of 0-127 m^3/h, 0-58 m^3/h, and 0-72%, respectively.

Figure 6.3 shows the schematic of the CH_4 flow to determine the collected stored CH_4 of the landfill. Collected stored CH_4 is differentiated form CH_4 collection capacity as the latter represents the total amount of CH_4 that can be accumulated in the landfill, whereas the former represents the stored CH_4 that is effectively collected after the restart of the LFG collection system. Five marker points have been noted in Figure 6.3: A, B, C, D, and E. Point A is the restart point of the LFG collection system. The quantity of the CH_4 flow at this point is zero or close to zero. Point B is when the CH_4 flow is similar to actual values. The trend of the collected stored CH_4 starts at point B. Point C is the peak of the collected stored CH_4 trend, and point D is the end of this trend. The surface under the curve between these points represents some of the accumulated CH_4, called the collected stored CH_4 (S) in this study. Since the emissions and landfill dynamics during this period are unknown, and the gas collection efficiency is not 100% when the gas collection system restarts, this surface is not the total stored CH_4 previously explained in theory but a part of it. The CH_4 flow accumulates between points B and D depending on the landfill gas storage capacity and gradually stabilizes between points D and E. Hence, point E ends the stabilized period of CH_4 flow for the studied shutdown-restart episode. Based on these explanations, the collected stored CH_4 of the landfill was calculated according to Eq. 6.3.

$$S = 24T_{BD} \sum F_{BD} - F_{DE\_Avg} \qquad \textbf{Eq. 6.3}$$

where S (*m³*) is the collected stored CH_4, T_{BD} (*days*) is the collected stored CH_4 duration from point B to D in Figure 6.3:, F_{BD} (m^3/h) is the CH_4 flow from point B to D in Figure 6.3: recorded every 15 minutes, F_{DE_Avg} (m^3/h) is the average CH_4 flow from point D to E in Figure 6.3:, and 24 is the conversion number indicating 24 hours per day.

6.2.2 Atmospheric pressure data

Hourly atmospheric pressure data were taken from a weather station approximately 37 km from the landfill site. The data was published by National Centers for Environmental Information (https://www.ncei.noaa.gov/) and collected for this study.

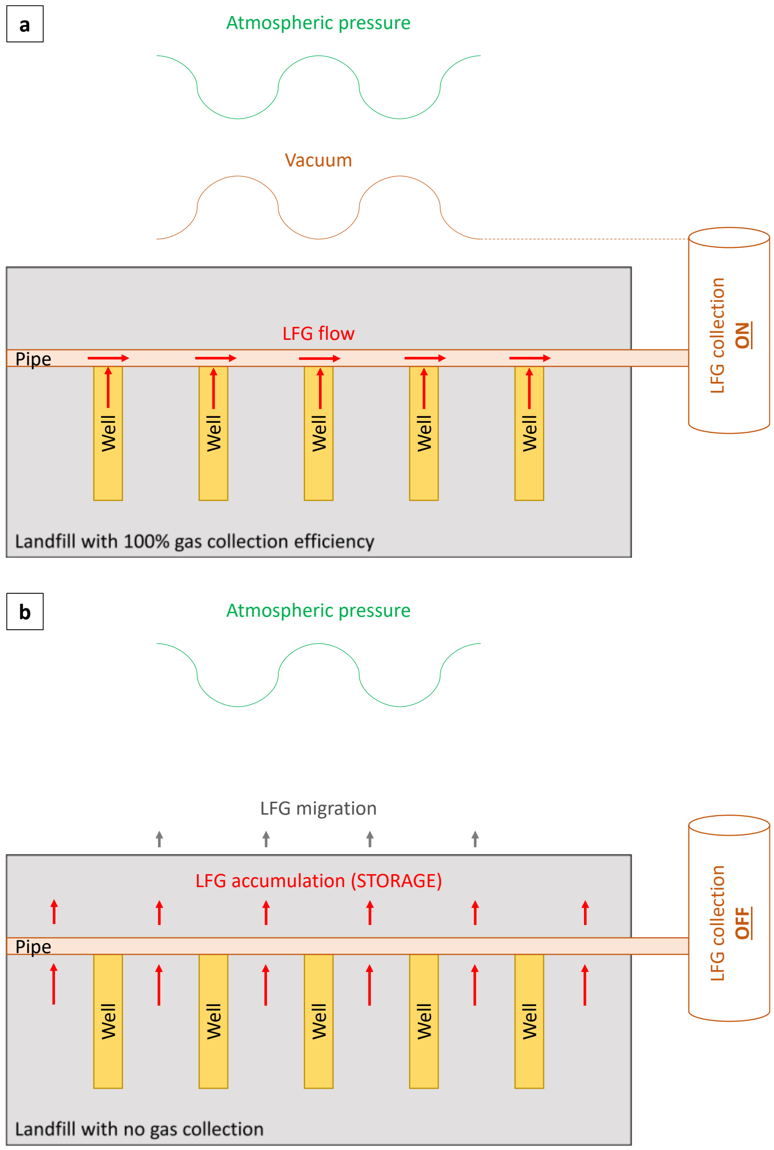


Figure 6.2: Schematic of the LFG flow when the gas collection system is (a) on or (b) off.

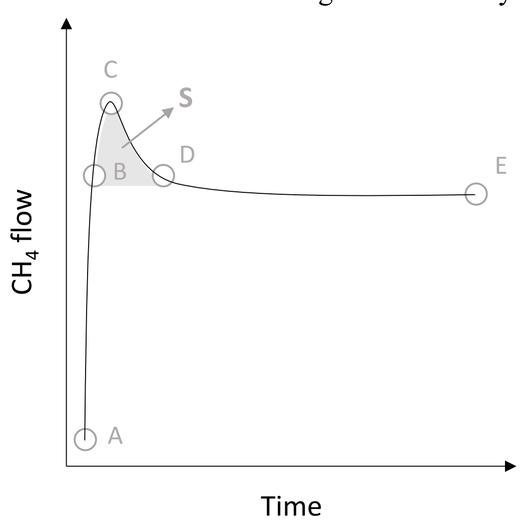


Figure 6.3: Schematic of the CH_4 flow to determine the highlighted CH_4 storage capacity of the landfill.

6.3 Results and Discussion

6.3.1 Gas collection data set

Table 6.1 shows the gas collection data set of the points indicated in Figure 6.3, including the durations, collected stored CH_4, collected CH_4, average CH_4 flow, average vacuum and average atmospheric pressure. All the data, except for durations and atmospheric pressure, are in relative values due to data confidentiality. Figure 6.4 is a schematic presentation of Table 6.1 and illustrates some of CH_4 flow and vacuum in relative values due to data confidentiality and atmospheric pressure gradient (difference of the atmospheric pressure per hour). As can be seen in this figure, CH_4 flows have similar trends shown previously in Figure 6.3. There is a peak period in all the CH_4 flows followed by a stable period. The vacuums and atmospheric pressure gradients in this figure have opposite trends.

In Table 6.1, the date represents three seasons and three years: winter (W), summer (S), or fall (F) of the year 1, 2, or 3. For instance, W1 is the winter of year 1, and so on. T_A (day) is the shutdown duration up to A, changing from 0.60 to 166.55 days. T_{AB} (day) is the start-up duration from A to B. It shows how fast the CH_4 flow can reach a value considered in calculating the collected stored CH_4. Based on the collection data, T_{AB} can be as low as zero or as high as 0.06 days (1.5 hours). Yet, it is around 0.02 days on average (0.6 hours). T_{BC} (day) is the duration of collected stored CH_4 up to the peak from point B to C. As shown in Table 6.1, 75% of the time, it takes less than 0.08 days (2 hours) for the CH_4 flow to reach the peak. However, 75% of the time, 0.39 days (9.2 hours) is required for the CH_4 flow to stabilize (T_{CD}). Accordingly, T_{BD} (day) is the duration of the collected stored CH_4 from point B to D. 50% of the time, it ranges between 0.17 to 0.41 days (4.0 to 9.8 hours) with a minimum and maximum of 0.04 and 0.55 days (1.0 and 13.3 hours), respectively, if we ignore three outliers (0.79, 1.05, and 1.30 days equal to 19.0, 25.3, and 31.3 hours). These outliers might be explained by an abnormal response time to obtain the stored CH_4 peak and come back to normal collected CH_4 flow. Looking through the collected stored CH_4 (S) and collected CH_4 from B to D during T_{BD} (C_{BD}) indicates that the average $\frac{S}{C_{BD}}$ is 10.5% ranging mainly from 4.1% to 14.5% with a minimum and maximum of 1.8% and 25.5%, respectively. The average CH_4 flow from B to D (F_{BD_Avg}) and D to E (F_{DE_Avg}) shows an increase of 11.0% in one hour. Also, the average atmospheric pressure from B to D (P_{BD_Avg}) and D to E (P_{DE_Avg}) change from 92.3 to 94.3 kPa and 92.0 to 94.3 kPa, respectively.

6.3.2 Correlation of the data

Minitab 21[2] was used to do a statistical analysis of the data. Accordingly, the Pearson correlation coefficients of the data (Table 6.2) and linear regression equations of correlated parameters underlined in Table 6.2 (Table 6.3) were defined. As can be seen in Table 6.2, some parameters are highly correlated, underlined and written in red. The other Pearson correlation coefficients are not considerable. For instance, a higher shutdown period, T_A, was expected to lead to a higher S. However, the low Pearson correlation coefficient (0.30) does not approve of this assumption. Hence, further analysis was done to find a correlation between T_A and S. In this regard, four categories of T_A were selected: T_A < 2 days, T_A < 3 days, T_A < 4 days, and T_A < 5 days, and their relevant S were analyzed. Yet, the results didn't conclude with a high correlation between the short-term T_A and S.

It is here discussed briefly how the findings of this study could be utilized to optimize both sources of energy from landfill and photovoltaic power plants over 24 hours. The goal is to modulate the LFG

[2] (https://www.minitab.com/en-us/ accessed 7 March 2023)

collection to use its energy when needed. Hence, the optimal time for stopping biogas pumping to avoid losing biogas in emissions, oxidation and lateral migration can be estimated considering the following assumptions.

1. As explained previously, when the LFG collection system is shut down (Figure 6.2 (b)), the LFG accumulates inside the landfill until it reaches the landfill gas storage capacity, and the rest will migrate from the landfill.
2. The rate of LFG generation is constant over a short period (a few hours or days).

S (m^3) accumulates according to the gas generation rate (m^3/h) (Eq. 6.3) and the average $\frac{S}{C_{BD}}$ was explained previously to be 10.5%. Therefore, for a storage percentage that is 10.5%, an accumulation time (T_A) of 10.5% is required (i.e., 2.5 hours). This finding implies that all the experiments were long enough to reach maximum collected storage as T_A changed from 0.60 to 166.55 days. Consequently, it seems normal that a correlation between S and T_A has not been observed. Considering this approach, variations in the value of S can be justified due to factors such as fluctuations in atmospheric pressure, variations in waste moisture content, waste degradation and settlement, and temperature.

According to Hydro-Quebec, daily peaks in electricity consumption are on weekday mornings, 6-9 a.m., and evenings, 4-8 p.m. [172]. If the shutdown time is 2.5 hours and since it takes 0.6 hours to start recovering the stored methane, therefore the shutdown should be done 3.1 hours before Hydro-Quebec's consumption peak. In this scenario, the total shutdown time should not exceed 2.5 hours because after this period of time there is a risk of emission increase.

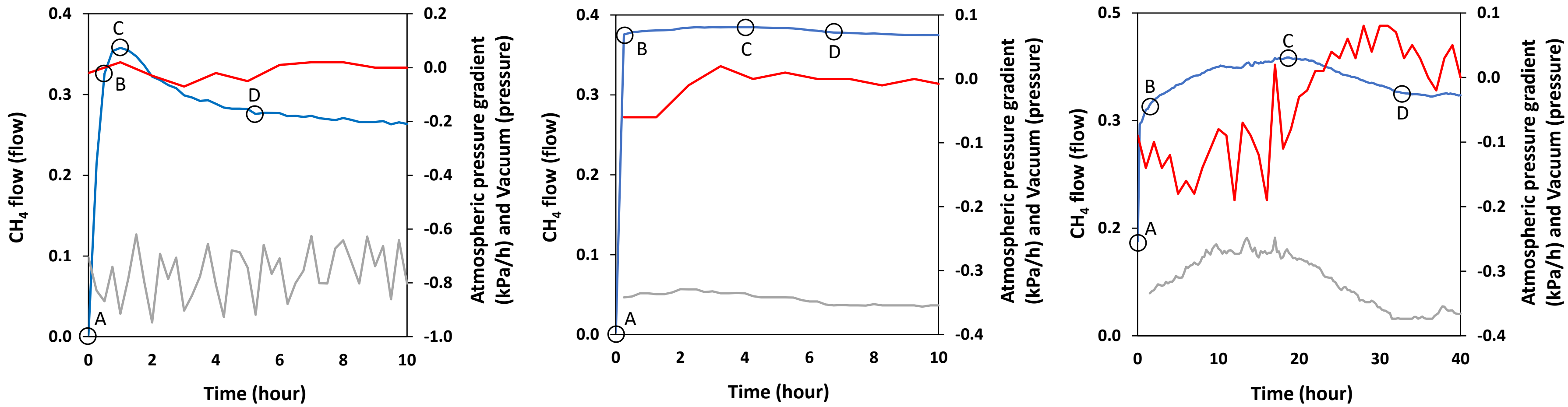
Moreover, T_{CD} is highly correlated to T_{BD} and C_{BD}. Yet, there is a lower correlation between T_{BC} and T_{CD}. Hence, it can be concluded that the collected stored gas trend starts with a sharp increase in CH_4 flow in a shorter time (T_{BC}) and a gradual decrease in CH_4 flow in a longer time (T_{CD}), making T_{CD} more correlated to T_{BD} rather than T_{BC}. Additionally, the collected gas and the collection duration are highly correlated for the gas storage period (T_{BD} and C_{BD}). An interesting finding of the good correlation of T_{AB}, T_{BC}, T_{CD}, and T_{BD} is that there is proportionality from the beginning. In other words, when the gas collection system starts working, the geometry of the curve (BCD triangle in Fig. 3) is always very similar.

Furthermore, a similarly high correlation exists between T_{BD} and C_{BD} with S. Hence, S can be estimated based on these parameters. In addition, the highest correlation (0.97) was found between average CH_4 flows (F_{BD_Avg} and F_{DE_Avg}).

Table 6.1: Durations, collected stored CH_4, collected CH_4, average CH_4 flow, average vacuum and average atmospheric pressure of points indicated in Figure 6.3.

No.	Date	T_A (day)	T_{AB} (day)	T_{BC} (day)	T_{CD} (day)	T_{BD} (day)	S (volume)	C_{BD} (volume)	F_{BD_Avg} (flow)	V_{BD_Avg} (pressure)	P_{BD_Avg} (kPa)	T_{DE} (day)	C_{DE} (volume)	F_{DE_Avg} (flow)	V_{DE_Avg} (pressure)	P_{DE_Avg} (kPa)
1	W1	11.71	0.02	0.02	0.18	0.20	6.59	29.50	0.31	-0.78	93.99	1.25	875.79	0.24	-0.77	93.96
2	W1	4.55	0.01	0.00	0.04	0.04	0.09	2.09	0.42	-0.39	92.60	0.49	225.34	0.40	-0.42	92.75
3	S1	84.06	0.04	0.04	0.20	0.24	1.22	50.02	0.36	-0.39	92.40	1.04	892.93	0.35	-0.38	92.05
4	S1	2.63	0.03	0.08	0.25	0.33	1.76	66.72	0.25	-0.69	93.66	0.65	240.27	0.25	-0.69	93.22
5	S1	3.65	0.01	0.02	0.39	0.41	10.58	99.74	0.26	-0.52	93.34	1.57	1311.97	0.23	-0.53	93.53
6	S1	0.90	0.01	0.02	0.34	0.36	4.25	113.50	0.36	-0.91	93.96	1.61	2096.43	0.35	-0.90	93.94
7	S1	0.82	0.01	0.02	0.39	0.41	11.70	142.36	0.37	-0.81	93.04	1.13	986.08	0.34	-0.84	93.30
8	S1	2.79	0.00	0.02	0.13	0.15	2.33	23.59	0.45	-0.79	93.42	1.25	1468.89	0.40	-0.85	93.86
9	S1	0.69	0.02	0.04	0.19	0.23	3.73	51.33	0.41	no data	93.84	0.71	442.03	0.38	no data	94.17
10	S1	0.60	0.01	0.01	0.17	0.18	1.26	30.84	0.40	no data	93.29	0.79	565.91	0.39	no data	93.52
11	F1	80.20	0.01	0.16	0.11	0.27	1.22	67.09	0.38	-0.34	93.49	0.20	35.68	0.38	-0.35	93.46
12	F1	2.55	0.06	0.72	0.58	1.30	177.78	1438.58	0.37	-0.30	92.27	1.05	824.67	0.32	-0.38	92.46
13	W2	147.00	0.05	0.05	0.74	0.79	65.49	828.29	0.57	-0.27	92.99	2.18	5721.57	0.52	-0.30	92.92
14	S2	29.45	0.05	0.09	0.96	1.05	151.46	1934.48	0.75	-0.41	93.60	0.59	572.23	0.69	-0.41	93.60
15	S2	0.74	0.04	0.04	0.24	0.28	25.91	101.62	0.54	-0.33	93.51	1.67	3325.18	0.52	-0.37	93.45
16	S2	2.17	0.05	0.04	0.26	0.30	30.92	122.81	0.56	-0.50	93.51	1.57	2424.05	0.42	-0.54	93.46
17	S2	2.63	0.01	0.00	0.14	0.14	1.05	16.56	0.36	-0.47	94.28	0.07	4.77	0.34	-0.49	94.27
18	S3	166.55	0.02	0.16	0.40	0.55	106.30	485.91	0.68	-0.65	93.65	2.41	7108.30	0.53	-0.70	93.58
19	S3	3.97	0.01	0.01	0.16	0.17	3.06	29.46	0.43	-0.89	93.89	0.84	644.70	0.39	-0.97	93.84
20	S3	33.85	0.01	0.02	0.02	0.04	0.56	3.69	0.74	-0.59	93.56	0.88	1116.94	0.63	-0.64	93.67

Date, collected stored CH_4, collected CH_4, average CH_4 flow, and average vacuum are in relative values due to data confidentiality.



No. 1 W1 | No. 11 F1 | No. 12 F1

Figure 6.4: Some of CH_4 flow (blue) and vacuum (gray) in relative values due to data confidentiality, and atmospheric pressure gradient (red).

Table 6.2: Pearson correlation of the data.

	T_A	T_{AB}	T_{BC}	T_{CD}	T_{BD}	S	C_{BD}	F_{BD_Avg}
T_{AB}	0.22							
T_{BC}	0.07	0.52						
T_{CD}	0.30	0.65	0.37					
T_{BD}	0.25	0.72	0.74	0.90				
S	0.30	0.69	0.74	0.80	0.93			
C_{BD}	0.22	0.68	0.59	0.89	0.93	0.93		
F_{BD_Avg}	0.42	0.28	-0.04	0.33	0.22	0.44	0.43	
F_{DE_Avg}	0.39	0.27	-0.07	0.36	0.22	0.41	0.46	0.97

Values written in red and underlined are highly correlated.

Since this kind of study is believed to initiate a research topic regarding landfill CH_4 storage, further investigation through coupling photovoltaic power plants, effects of different parameters on landfill gas storage capacity, and so on, could be considered some of the topics to be worked on in future studies. In order to predict the quantity of collected stored methane, the collection equipment should be regulated similarly before and after the shutdown. In other words, collected CH_4 flow before the shutdown and during the stable period (F_{DE_Avg}) are similar. This assumption seems realistic since the time between the shutdown and the stabilisation of flow after restart (F_{DE_Avg}) is as short as few hours (like the suggested 2.5 hours shutdown in this study). To verify this idea, a comparison between CH_4 flows before shutdown and after the restart can be done. This could be easily verified but it was not done in this study because shutdowns happened due to operational problems. Therefore, operational data did not allow this verification.

Additionally, an important consideration in future studies is to conduct tests to ensure no environmental impacts occur during the shutdown period. These tests could include measuring the pressure increase in the closed collection wells until a plateau pressure is reached or performing surface emission measurements to quantify the time required to observe the rise in emissions. Therefore, the maximum shutdown time without generating an increase in emissions can be validated. Also, a predictive model could propose the maximum shutdown duration before significant environmental impacts appear.

6.4 Conclusion

Landfill gas storage can become an interesting option to control and manage energy output from landfill sites.

This study assessed the landfill gas accumulation and storage inside a closed landfill in the province of Quebec, Canada. Gas collection data was used to analyze the methane flow and durations when the gas collection system was shutdown or restarted. According to this methodology, it is possible to predict the collected stored methane, start-up duration and duration of the whole collected stored methane from the flow before shutdown. Although this study is for a single landfill case study, the methodology can be applied to other landfills.

Table 6.3: Regression equations of correlated parameters underlined in Table 6.2.

$$S = 3.74 + 0.09443\, C_{BD} \qquad \textbf{(Eq. 6.4)}$$

$$F_{BD\_Avg} = -0.0183 + 1.1586\, F_{DE\_Avg} \qquad \textbf{(Eq. 6.5)}$$

$$T_{AB} = 0.00902 + 0.04153\, T_{BD} \qquad \textbf{(Eq. 6.6)}$$

Shutdown periods were as low as 0.60 days and as high as 166.55 days. It could take up to 0.06 days (1.5 hours) for the pumping system to start up the gas collection and reach the flow before shutdown (T_{AB}). The recovery duration of the collected stored methane ranged from 0.17 to 0.41 days (4.0 to 9.8 hours) with a minimum and maximum of 0.04 and 0.55 days (1.0 and 13.3 hours), respectively, ignoring three outliers (0.79, 1.05, and 1.30 days equal to 19.0, 25.3, and 31.3 hours). The average ratio of the collected stored methane compared to total methane collection during the storage period was 10.5%, ranging mainly from 4.1% to 14.5% with a minimum and maximum of 1.8% and 25.5%, respectively. The ratio of the average methane flow in the storage period and in the stabilization period was 11.0%, showing the equivalent of an increase in stored energy of 11%. Additionally, the results indicated that the dynamics of the flow variations after restarting the gas pumping (as shown in the shape of the BCD triangle in Fig. 3) are always very similar. The summary of the results is as follows:

- The collected stored methane is accumulated in an average of 2.5 hours.
- If the shutdown time is 2.5 hours and since it takes 0.6 hours to start recovering the stored methane, therefore the shutdown should be done 3.1 hours before Hydro-Quebec's consumption peak.
- Collected stored methane represents 10.5% of methane flow.

A predictive model of the quantity of the collected stored methane can be applicable only when operational parameters, such as landfill gas pumping speed, are similar before and after shutdowns. When shutdowns are planned to store methane within a short period, then operational data is likely to be similar.

Chapter 7. Conclusion

In conclusion, this Ph.D. thesis has explored various aspects of waste management and energy recovery in different cities, providing valuable insights and recommendations for improving current practices.

The first conclusion highlights the significant potential of energy recovery from food waste and yard waste, particularly in New York City and Montreal. The comparison of waste flow between the two cities revealed that a large portion of waste is currently being landfilled, indicating a need for alternative management methods. Estimating the biogas generation potential using the Buswell equation demonstrated that a substantial amount of biogas could be generated from food waste and yard waste in both cities. This underscores the importance of diverting these waste streams from landfills and utilizing them for energy production.

The second conclusion focuses on analyzing waste flow in Montreal, explicitly examining different waste management scenarios. A life cycle assessment study determined that scenarios involving anaerobic digestion and composting for food waste and yard waste showed promising results regarding energy savings and reduced greenhouse gas emissions. Furthermore, an optimization algorithm was applied to optimize the waste flow in Montreal, leading to a waste management system that prioritizes anaerobic digestion, composting, recycling, and landfilling. This integrated approach minimizes energy consumption and carbon dioxide emissions and promotes renewable energy production and job creation. However, challenges such as waste separation, policy support, and public awareness must be addressed for successful implementation.

The third conclusion focuses on modeling landfill gas generation, specifically methane and hydrogen sulfide. By applying a genetic algorithm to optimize modeling parameters, the study showed that considering different waste types and landfill sectors improves the accuracy of landfill gas predictions. Waste characterization was identified as a crucial factor in gas prediction, and scenarios involving all the landfill sectors provided the best parameter fit. The study also indicated that further data could enhance the modeling accuracy for hydrogen sulfide generation.

Lastly, the fourth conclusion centers on landfill gas storage and its potential as an energy management strategy. The case study of a closed landfill in Quebec demonstrated the feasibility of predicting the collected stored methane, start-up duration, and recovery duration. Shutdown periods varied, and it was observed that the collected stored methane represented a significant portion of the landfill gas flow. Additionally, the study highlighted the consistent dynamics of flow variations after restarting the gas pumping system. While proposing a similarity between the collection flow before shutdown and the stable flow, further tests are needed to validate this proposition and assess potential environmental impacts.

This Ph.D. thesis has provided comprehensive insights into waste management, energy recovery, and landfill gas dynamics. The findings highlight the significance of diverting organic waste from landfills, optimizing waste management strategies, and exploring innovative approaches for energy generation. Addressing the identified challenges and implementing the proposed recommendations can contribute to more efficient and sustainable municipal solid waste management systems globally.

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