Doctoral Dissertation

Environmental Pollutions of Landfilled Municipal Solid Waste and Its Energy Recovery Potential: A Case Study of Phnom Penh Municipality

（埋立固形廃棄物の環境汚染とエネルギー回収の可能性： プノンペン市の事例）

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Summary of Dissertation

Municipal solid waste (MSW) generation in Cambodia has significantly increased in recent decades. The absence of intermediate waste treatment facilities and limited source segregation practices have created substantial pressure on MSW landfills throughout the country. Like many other developing nations, the organic fraction accounts for more than 50%, ultimately ending up in landfills, mixing with other waste compositions. Under anaerobic conditions in landfills, biodegradable waste decomposes, producing leachate and landfill gas (LFG), particularly methane (CH_4). Leachate is a liquid commonly containing various pollutants, including dissolved organics, inorganic macro compounds, heavy metals, and xenobiotic organic compounds. If not adequately managed, leachate has the potential to contaminate groundwater and surface water, accumulate in the soil, and substantially enter the food chain, posing risks to human health. Additionally, CH_4 is a potent greenhouse gas (GHG) that contributes to global warming and climate change.

This study aimed to assess environmental pollution resulting from MSW landfilling and determine a mitigation strategy by optimizing resource recovery in Phnom Penh municipality. Firstly, the study investigated the influence of landfill leachate on the surrounding environment of the Dangkao landfill site in Phnom Penh and its vicinity. The focus was on determining the concentration of heavy metals in surface water, groundwater, soil, plants, and fish using an Inductive Coupled Plasma Optical Emission Spectrophotometer. Secondly, the study estimates the generation of CH_4 by employing the Landfill Gas Emission Model (LandGEM) and IPCC first-order decay (FOD) model. Based on CH_4 estimation results, the overall GHG emissions from different landfill management options were quantified. Thirdly, various MSW management scenarios were developed to minimize landfilling and maximize recycling and resource recovery. The study evaluated the GHG emissions and reductions associated with the proposed scenarios, following the life cycle assessment framework and IPCC 2006 guidelines. Finally, the study analyzed the economic feasibility of waste-to-energy (WTE) technologies, including incineration, anaerobic digestion (AD), and LFG recovery. This analysis took into account the energy recovery potential, economic viability, and emissions reduction.

The findings indicated that Cd, Cr, Ni, and Pb concentrations in surface water exceeded the permissible limits. Additionally, all heavy metals were found higher in the downstream canal, suggesting a significant accumulation of these pollutants in leachate that is discharged into the canal. Furthermore, Cd, Cr, and Ni were also found to be above the standard limits of the World

Health Organization (WHO). The groundwater located in the landfill exhibited higher levels of all heavy metals compared to other sampling sites. This can be attributed to an improper liner, which allows these metals to seep through the landfill and pass through the soil, ultimately contaminating the groundwater. Regarding heavy metals in the soil, only Cd and Ni were found to exceed the allowable limits. The soil samples collected from low-lying inundation areas showed a higher accumulation of heavy metals than other sampling sites. This suggests that the heavy metals are likely transported through water and accumulate in the soil during flooding or high-water levels. Similarly, in plant samples, only Cd concentrations exceeded the allowable limits of the WHO, suggesting that the plants have absorbed and accumulated high levels of Cd. Excessive Cd, Pb, and Zn levels were found in fish samples, exceeding the allowable limits. These findings indicate that the migration of leachate, coupled with the low-lying topography, contributed to the accumulation of heavy metals in water and soil, which were then transferred to plants and fish. The elevated levels of heavy metals in groundwater, plants, and fish signify a potential health risk for individuals who regularly consume these contaminated sources. The study also observed that the accumulation of heavy metals was mostly higher during the dry season, possibly due to lower water volume, reduced dilution, and higher water temperatures.

The estimated CH_4 generation from the Dangkao landfill showed an increasing trend, rising from 1.54 M kg/year in 2010 to 36.50 M kg/year in 2022, according to the LandGEM model. The IPCC FOD model estimated a relatively higher CH_4 generation, ranging from 2.17 M kg/year in 2010 to 42.83 M kg/year in 2022. Considering that 75% of CH_4 generation is collected for electricity production, the energy potential was estimated at 51 GWh/year based on the LandGEM model and 61 GWh/year based on the IPCC FOD model. Four landfill management scenarios were developed to address the environmental pollution arising from CH_4 emissions. Scenario 1 represents the current landfill management practice without leachate treatment and LFG collection systems, resulting in an average of 397 and 496 M $kgCO_2$-eq/year of GHGs based on the LandGEM and IPCC FOD models, respectively. Scenario 2 involves an improved landfill management practice incorporating a leachate treatment system, emitting an average of 409 M $kgCO_2$-eq/year based on the LandGEM model and 509 M $kgCO_2$-eq/year according to the IPCC FOD model. The increase in emissions scenario 2 primarily stems from the additional emissions arising from the leachate treatment process. Scenario 3 introduces an engineered landfill equipped with a leachate treatment and flaring system, leading to a reduction in GHG emissions of at least 55%. Additionally, scenario 4 presents an upgraded system that utilizes LFG recovery for electricity production, mitigating at least 83% of GHG emissions. Despite being the most favorable option due to its substantial GHG

reduction and electricity generation capabilities, scenario 4 still contributes to environmental pollution through uncollected CH_4 and leachate leakage. Hence, implementing a waste landfill reduction strategy is crucial to minimize the environmental impacts of the landfill.

To mitigate environmental pollution resulting from heavy metal contamination and landfill CH_4 emissions, five MSW management scenarios were developed to minimize landfilling. The study considered the direct GHG emissions from waste transportation, open burning, composting, recycling, AD, incineration, and landfilling. Additionally, the avoided emissions from recycling and electricity generation from incineration and AD plants were also quantified. The results indicate that scenario 5 achieved the most significant net GHG emissions savings. In this scenario, food waste and recyclables were separated at a rate of 75%, resulting in GHG emissions savings of approximately –1.59 M $kgCO_2$-eq/day. This saving was achieved through composting 472 tMSW/day of food waste, recycling 867 tMSW/day of mixed recyclables, AD of 943 tMSW/day of digestible food waste, and incineration of 1,617 tMSW/day of commingled waste. On the other hand, the worst-case scenario represents the current MSW management method, which generates the highest GHG emissions of 3.89 M $kgCO_2$-eq/day. This is primarily due to the open burning of uncontrolled waste (200 tMSW/day) and landfilling (3,530 tMSW/day). Based on the analysis, it is highly recommended to implement an integrated MSW management system that includes source separation for recycling and resource recovery purposes.

The economic feasibility of incineration, AD, and LFG recovery technologies was assessed for the period from 2023 to 2042, considering the levelized cost of electricity (LCOE), payback period (PBP), and net present value (NPV). The results indicated the following ranking: incineration > AD > LFG recovery. Incineration technology produced the highest energy output, ranging from 793 to 1,626 GWh/year, while AD and LFG recovery technologies yielded 163 to 333 GWh/year and 115 to 272 GWh/year, respectively. The economic analysis showed an average LCOE of 0.07 USD/kWh for LFG recovery, 0.053 USD/kWh for incineration, and 0.093 USD/kWh for AD. Incineration and LFG recovery were found to be economically feasible, with positive NPVs and the potential for profitability within 8.36 years for incineration and 7.13 years for LFG recovery. However, AD technology exhibited a negative NPV and would require more than 20 years to generate a return on investment. Despite its economic drawbacks, AD technology demonstrated the most promising environmental performance, with potential savings of approximately –139.74 M $kgCO_2$-eq/year. Incineration, although profitable, concerns have been raised due to emissions and bottom ash management. To address these concerns, compliance with air emission standards

and proper bottom ash management are essential to mitigate potential health risks. The government has recently made efforts to encourage investment in WTE technologies in Cambodia. However, regulations and incentive policies, such as investment subsidies, tax exemptions, carbon credits, etc., should be implemented to make WTE projects more attractive for commercial schemes.

In conclusion, the management of MSW is a pressing global issue. The detrimental effects of heavy metals contamination and the generation of CH_4 can result in local and global pollution, as well as adverse impacts on human health. To address the complexities of MSW management, a comprehensive strategy is needed combining multiple treatment technologies, focusing on enhancing recycling and resource recovery, particularly WTE solutions. Raising public awareness to participate in source separation and providing subsidies for installing the necessary infrastructure and facilities for waste collection and processing are crucial steps in promoting sustainable waste management practices. Government support and partnerships between the public and private sectors can play a vital role in implementing these initiatives and improving the overall waste management system.

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Abbreviations

AD	Anaerobic digestion
CAPEX	Capital expenditure
CF	Capacity factor
CH_4	Methane
CO_2	Carbon dioxide
DOC	Degradable organic carbon
DOC_f	Fraction of degradable organic carbon
DS	Dry season
EF	Emission factor
ERP	Energy recovery potential
FIT	Fit-in tariff
GHG	Greenhouse gas
GWP	Global warming potential
ICE	Internal combustion engine
IPCC	Intergovernmental Panel on Climate Change
IRR	Internal rate of return
LandGEM	Landfill Gas Emissions Model
LCC	Life cycle costing
LCOE	Levelized cost of electricity
LFG	Landfill gas
LHV	Lower heating value
MCF	Methane correction factor
MSW	Municipal solid waste
NMOC	Non-Methane Organic Compound
NPV	Net present value
N_2O	Nitrous oxide
OF	Oxidation factor
O&M	Operation and maintenance
OPEX	Operation expenditure
PBP	Payback period
SRF	Solid refuse fuels
TLCC	Total life cycle cost
WS	Wet season
WTE	Waste-to-energy
3Rs	Reduce, reuse, and recycle

CHAPTER 1

INTRODUCTION

1.1. Background

Municipal solid waste (MSW) is a growing global challenge, particularly in developing and least developed countries that often face financial constraints, limited technological resources, and inadequate policy frameworks to effectively management waste (Batista et al. 2021; Ferronato et al. 2017; Khan et al. 2022; Modak et al. 2017). Projections indicate that global MSW generation will rise significantly in the coming decades. According to Kaza et al. (2018), it is expected to increase from 2.01 billion tons in 2016 to 3.40 billion tons in 2050. Notably, low- and middle-income countries are projected to experience the highest growth rate, accounting for at least 40% of the increase. Among the regions contributing the most to this trend, East Asia and Pacific countries accounted for 23%, followed by Europe and Central Asia (20%) and South Asia (17%) (Kaza et al. 2018). The increase in waste generation is attributed to several factors, such as population growth, rapid urbanization and industrialization, and the improvement in lifestyle (UNEP 2017).

The composition of MSW varies across countries with different income groups. Low- and middle-income countries tend to have a higher proportion of organic waste, averaging between 46 to 53%, compared to high-income countries, which have an average of 34% (UNEP 2015). In contrast, high-income countries typically have a higher proportion of recyclable materials, such as paper, plastics, metals, and glass, in their waste streams, as shown in Figure 1. The composition of waste can have significant implications for environmental and health risks associated with waste management. For example, the presence of hazardous or toxic materials in the waste stream can pose risks to workers and the environment if not properly handled and disposed of. Additionally, the composition of waste can also have implications for the circular economy and the 3Rs (reduce, reuse, recycle). For instance, if the waste stream contains a high percentage of materials that can be recycled, such as paper, plastics, and metals, it may be more feasible and cost-effective to invest in recycling infrastructure and programs. On the other hand, if the waste stream is dominated by organic waste, composting and anaerobic digestion may be more appropriate waste management options.

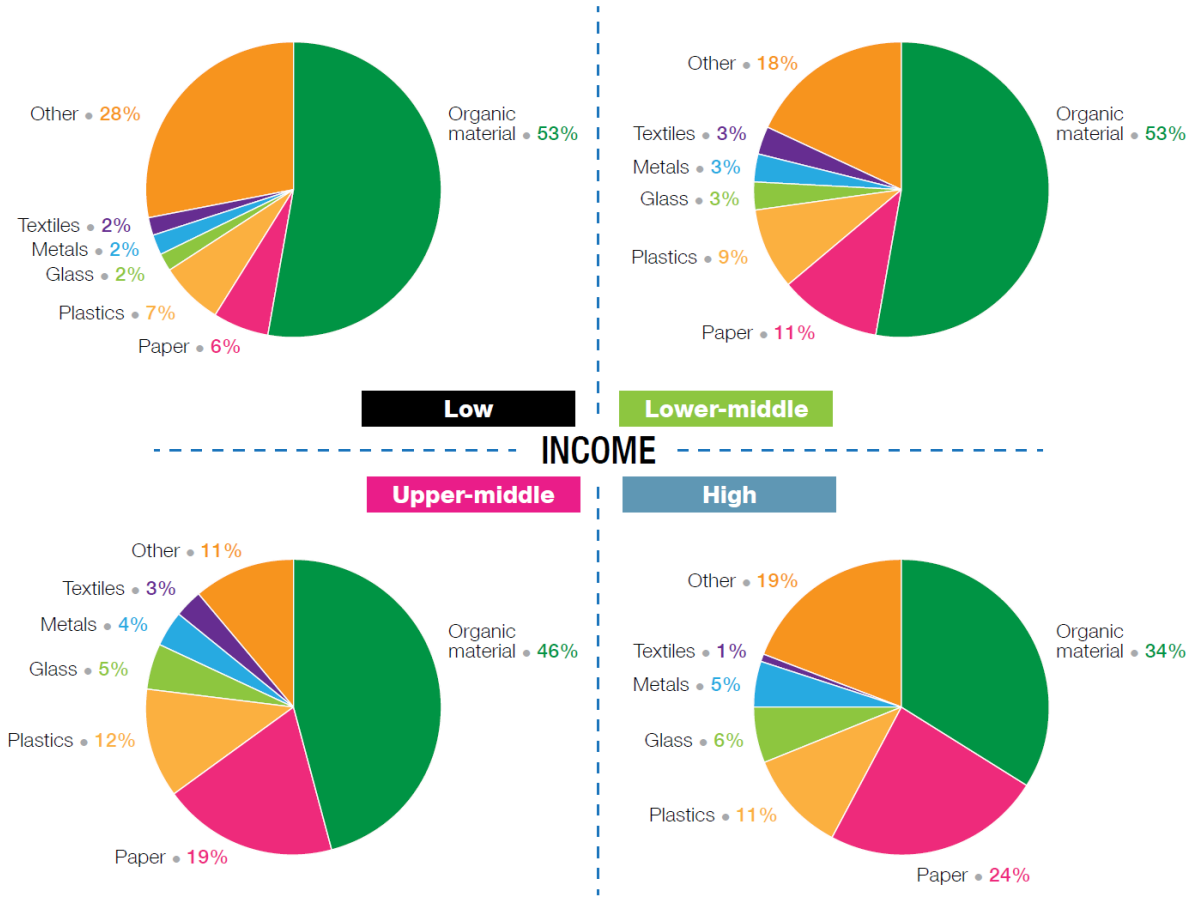


Figure 1. MSW composition grouped by country income levels (UNEP 2015)

With an average per capita generation of 1.14 kg/capita/day (UNEP 2017), ASEAN countries are expected to generate approximately 270 million tons of MSW (M tMSW) in 2020. MSW composition in all ASEAN countries is predominantly organic (greater than 50%), except for Singapore, where organic waste makes up only 10.5% of the total MSW (UNEP 2017).

The management of MSW is a growing concern due to its detrimental effects on the environment and human health. In Asian countries, the moisture content in MSW is typically high, primarily due to the high proportion of food waste, which makes the separation and processing of food waste difficult (Ishigaki et al. 2011). Improper management of MSW can result in the release of pollutants into the air, water, and soil. For example, burning waste in open dumps can emit toxic gases and particulate matter that are harmful to human health and contribute to climate change. MSW can attract vermin and create breeding grounds for disease-carrying insects. The production and disposal of MSW can deplete natural resources, leading to economic and environmental consequences. These consequences include increased costs for waste management and reduced availability of resources for future generations. The management of MSW also has implications

for social equity. Waste is often disproportionately generated by low-income communities and communities of color. These communities may also be more likely to reside in close proximity to waste management facilities, resulting in negative health impacts and a diminished quality of life. Addressing the issues associated with MSW requires a comprehensive approach that encompasses waste reduction, reuse, recycling, recovery, and safe disposal, as depicted in Figure 2.

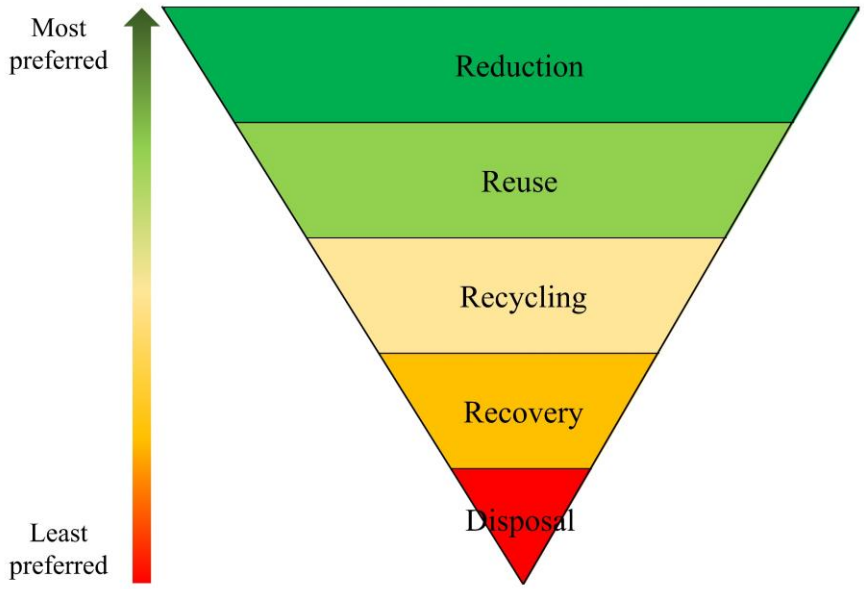


Figure 2. Waste management hierarchy

1.2. GHG emissions from MSW management

Greenhouse gas (GHG) emissions can be generated throughout the entire life cycle of a product, from resource extraction, production, consumption, and final disposal (Figure 3). When it comes to waste, GHG emissions can occur at various stages of waste management system, including collection and transportation, treatment, and final disposal (Chen and Lo 2016; Kristanto and Koven 2020). It is worth noting that emissions from MSW management system account for approximately 5% of global GHG emissions (IPCC 2006). The main GHGs emitted from MSW management include carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). During the collection and transportation of MSW, GHG emissions occur due to the consumption of fossil fuels to power vehicles. Similarly, GHG emissions can also arise during the treatment of MSW, which includes recycling, incineration, anaerobic digestion (AD) or composting. Recycling plays a vital role in waste management as it reduces the amount of waste sent for disposal and allows for the reintegration of materials into the economy. However, it is important to note that GHG emissions associated with the recycling process can arise from the CO_2 generated by the electricity used to

power material recovery facilities. Despites this, the use of recycled materials in the manufacturing of new products can result in significant energy savings (Turner, Williams, and Kemp 2015). For instance, according to Turner et al. (2015), using recycled aluminum to produce new aluminum requires 95% less energy compared to using virgin materials. This demonstrates the substantial energy efficiency gained through the utilization of recycled materials. As the costs and environmental impact of virgin materials continue to escalate, the relative value of secondary materials is expected to rise as well (Hoornweg and Bhada-Tata 2012).

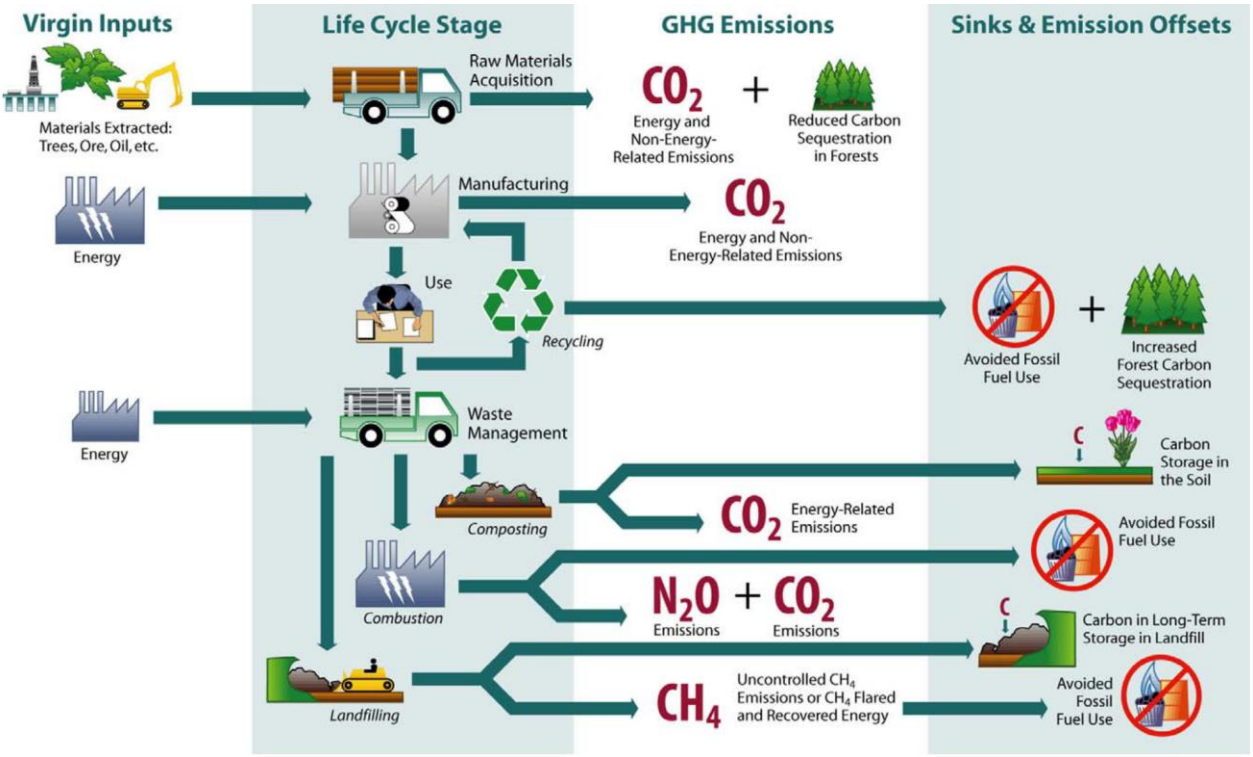


Figure 3. Greenhouse gas sources and sinks associated with waste life cycle (US EPA 2006)

Biological treatment methods such as composting and anaerobic digestion (AD) are employed for the management of organic waste. Composting is a practical approach that reduces the volume of waste going to landfills by converting biodegradable waste into fertilizer. However, it is important to note that compositing organic waste under aerobic conditions can result in production of GHGs that contribute to global warming (Hoklis and Sharp 2014; Pikoń and Gaska 2010; Seng et al. 2013). Additionally, improper composting practices can lead to water contamination (Gaeta-Bernardi and Parente 2016). Studies have shown that composting source-separated organics, as opposed to processing mixed MSW with front-end or back-end separation, significantly reduces contamination in the final compost (Lundie and Peters 2005; Tayyeba, Olsson, and Brandt 2011). Moreover, utilizing compost as a substitute for chemical fertilizer in agricultural activities has the

potential to increase carbon storage in the soil and reduce GHG emissions associated with waste decomposition and chemical fertilizer production. Composting is generally considered less complex, more forgiving, and less expensive than AD. However, organic waste, especially food waste, can serve as feedstock for AD plants to produce CH_4 and digestate as by-products. The generated CH_4 can be flared or used for heat and electricity generation. The electricity generated from AD can replace high-emission energy sources, thereby significantly reducing GHG emissions (IPCC 2006). Furthermore, the produced digestate can be utilized as a natural fertilizer, replacing chemical fertilizers and contributing to the reduction of GHG emissions associated with fertilizer production processes (Sang-Arun, Heng, and Al. 2011).

Incineration is a waste treatment process in which waste materials are subjected to high temperatures and burned through oxidation, breaking them down into their chemical components. Typically, temperatures ranging from 750 to 1000 °C are used, resulting in the generation of heat and energy (Tozlu, Özahi, and Abuşoɨlu 2016). This technology can significantly reduce the mass and volume of waste, by up to 70% and 90%, respectively (Ghosh et al. 2020). Achieving these substantial volume reductions is often possible when the waste streams contain significant amounts of packaging materials, paper, cardboard, plastics, and horticultural waste (Chen 2018). One limitation of incineration is that it primarily produces CO_2 and bottom ash as by-products. However, it is considered more favorable than direct landfilling when pollution control requirements and costs are sufficiently addressed. This is because incineration enables the recovery of energy from the waste before its final disposal (Chen and Liu 2021; Xin et al. 2020). It is important to note that incineration without energy recovery is generally not preferred due to its high costs and potential for pollution. In contrast, open burning of waste is strongly discouraged as it results in severe air pollution caused by low-temperature combustion (Kristanto and Koven 2020).

1.3. Environmental pollution from landfill

As shown in Figure 4, landfilling continues to be a commonly used waste management option in both developed and developing countries due to its simplicity and cost-effectiveness (Christensen, Manfredi, and Knox 2010; Kumar et al. 2004; Kumar and Sharma 2014a). When MSW is deposited in landfills, anaerobic conditions can develop, resulting in the production of CH_4, a potent GHG that is 25 times more effective at trapping heat in the atmosphere than CO_2 (IPCC 2006). Inadequate landfill management practices has resulted in adverse environmental impacts (Niskanen et al. 2013), including GHG emissions (Scheutz and Kjeldsen 2019), the spread of

disease (Ferronato and Torretta 2019), and the contamination of water, soil, and plants with heavy metals (Vongdala et al. 2019).

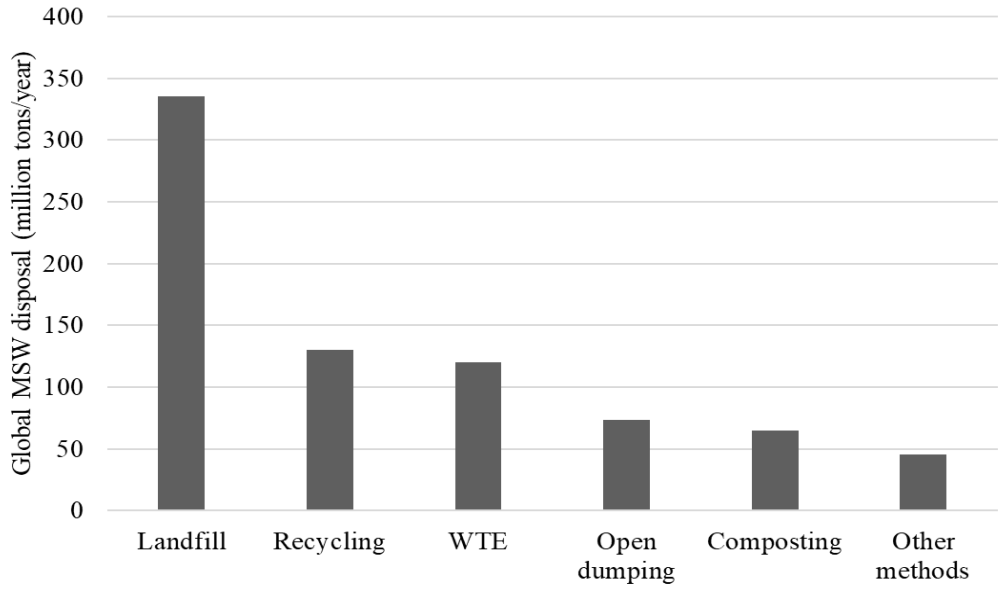


Figure 4. Total MSW disposal worldwide Hoornweg and Bhada-Tata (2012)

Landfill gas (LFG) is a natural by-product that is produced as a result of the anaerobic decomposition of biodegradable wastes buried in the landfill site over time (Ishigaki et al. 2011). LFG mainly consists of methane (CH_4) and carbon dioxide (CO_2), with trace elements of non-methane organic compounds (NMOCs) (Amini, Reinhart, and Niskanen 2013; Machado et al. 2009). The generation of LFG can vary based on the physical composition of MSW, permeability and moisture, temperatures, mixture of waste, and landfill management, such as surface soil cover, leachate recirculation, and liquid additions (Lizik et al. 2013; Spokas, Bogner, and Corcoran 2021). The high proportion of biodegradable waste, coupled with the high moisture content in MSW, can accelerate the rate of degradation and result in the rapid production of LFG once the waste is deposited in the landfill (Ishigaki et al. 2011). Tropical landfills experiencing high levels of precipitation may exhibit a higher rate of CH_4 generation compared to dry landfills. As a result, Asian countries, where organic waste is predominant, waste separation is low, and precipitation is high, tend to have higher LFG emissions (Ishigaki et al. 2011).

The fugitive CH_4 emissions from MSW landfill sites have a high global warming potential (GWP), accounting for approximately 5% of global anthropogenic GHGs (Bogner et al. 2007). CO_2 is the second-largest gas emitted from landfills but is not considered as a GHG due to its biogenic origin (IPCC 2006). To mitigate GHG emissions from landfills, LFG collection systems can be

implemented to capture and utilize the gas. LFG can be collected for flaring or combusted for energy generation through the use of internal combustion engines, gas turbines, or microturbines (LMOP 2021). However, even with an effective LFG collection system in place, a certain amount of LFG may still be released into the atmosphere (Barlaz, Chanton, and Green 2009). This can occur through CH_4 oxidation when the landfill surface is covered with soil, leakage from the collection system, or through the leachate collection system (Amini et al. 2013). Landfills equipped with an LFG collection system can reduce their pollutant effects by at least 75% (IPCC 2006).

Leachate is a liquid by-product that forms when water percolates through the waste in an MSW landfill. MSW landfills typically receive a combination of household waste, market waste, commercial waste, and hazardous waste such as batteries, paints, vehicle maintenance products and other residuals (Slack, Gronow, and Voulvoulis 2005). As a result, leachate contains a mixture of organic matter, inorganic pollutants, and hazardous substances that are present in the landfilled waste (Aziz, Umar, and Yusoff 2010; Slack et al. 2005). Implementing advanced landfill management techniques can help minimize environmental pollution, accelerate decomposition rates, enhance LFG generation, improve leachate quality, and reduce expenditures on leachate treatment (Bareither et al. 2010).

1.4. Overview of MSW management in Cambodia

Solid waste in Cambodia is commonly classified into three categories: household waste, commercial waste, and industrial and hazardous wastes, which includes medical waste (Akenji et al. 2019; Sethy, Sothun, and Wildblood 2014). However, there is no consensus on this classification, and the availability of up-to-date data in Cambodia is relatively limited (Kham and Heilmann 2015). MSW in Cambodia encompasses waste generated from households, markets, restaurants, shops, hotels, offices, street sweepings, and miscellaneous sources (JICA 2005; Seng et al. 2010). With rapid urbanization and industrialization, more people are migrating to urban cities to search for better job opportunities, higher education, and improved access to healthcare services. This has resulted in an unpredictable MSW generation pattern as urban areas experience increased population density and lifestyle changes.

1.4.1. MSW generation

Cambodia, like many other countries, is facing significant challenges in managing MSW. The rapid urbanization and industrial development in the country have led to a migration of people from rural to urban areas. However, the lack of intermediate waste treatment facilities and limited source segregation practices have put immense pressure on MSW landfills across the country. In 2021,

approximately 2.94 M tMSW were collected and disposed of in 164 landfills in Cambodia (Dek et al. 2022). Unfortunately, most of these landfills are unsanitary and do not have essential infrastructure such as soil cover, leachate treatment systems, and LFG control measures. As a result, waste is often simply dumped without proper management practices. In an attempt to reduce the quantity of waste being landfilled and extend the lifespan of the landfills, burning waste at the landfills is a common practice.

The amount of MSW generated has significantly increased over the past few decades and varies from region to region. However, obtaining comprehensive and consistent national data on MSW in Cambodia has been challenging, except for the data available for Phnom Penh municipality where reliable data is obtained through the installation of a weighting bridge at the landfill site (Seng et al. 2010). Despite this, aggregated data from various sources indicates a linear increase in MSW generation throughout the country. For instance, the data shows an increase from 2.50 M tMSW in 1990 to 4.24 M tMSW in 2016 (NCSD/MoE 2020). Akenji et al. (2019) reported a MSW generation of 4.09 million tons with a generation rate of 0.73 kg/capita/day in 2015. According to the Ministry of Environment, approximately 4.78 M tMSW were estimated to be generated in 2020, with a per capita generation of 0.78 kg/day, as shown in Table 1. The global average per capita waste generation is 0.74 kg/day (Kaza et al. 2018).

Table 1. Estimation of MSW generation based on population for 2008-2020

Year	Population (million) [a]	GDP [a]			MSW generation (M tMSW)		Per capita (kg/day)
		Per Capita (USD)	Annual (million USD)	Annual growth (%)	[b]	[c]	
2008	13.88	746	10,352	6.69	3.74	3.71	0.73
2009	14.09	738	10,402	0.09	3.78	3.79	0.74
2010	14.31	786	11,242	5.96	3.85	3.85	0.74
2011	14.54	882	12,830	7.07	3.91	3.92	0.74
2012	14.78	951	14,054	7.31	3.99	3.99	0.74
2013	15.03	1,013	15,228	7.36	4.09	4.06	0.74
2014	15.28	1,093	16,703	7.14	4.16	4.14	0.74
2015	15.52	1,163	18,050	7.12	4.18	4.41	0.78
2016	15.77	1,270	20,017	6.94	4.24	4.49	0.78
2017	16.01	1,385	22,177	6.84	-	4.58	0.78
2018	16.25	1,512	24,572	7.47	-	4.67	0.79
2019	16.49	1,643	27,089	7.05	-	4.69	0.78
2020	16.72	1,513	25,291	-3.14	-	4.78	0.78

Source: [a] World Bank (2021), [b] NCSD/MoE (2020)

1.4.2. MSW characteristics and composition

The composition of MSW in Cambodia is predominantly food and organic waste, accounting for approximately 55% of the total waste, as shown in Table 2. Often, this organic waste is disposed of without proper separation from other waste fractions such as plastic, glass, textile, and paper, at the sources. These practices can not only pollute the environment but also negatively affect human health.

Table 2. Comparison of waste composition among provinces and municipality

Locations	Waste composition (%)							
	Food waste	Paper	Plastic	Metals	Textile	Glass	Wood and dry matter	Other
Country [a]	55	3	10	7	-	8	-	17
Phnom Penh [b]	49	7	21	1	8	1	7	6
Battambang [c]	71	2	10	3	2	4	6	2
Siem Reap [c]	54	6	11	1	3	3	11	11
Kampong Cham [c]	60	5	12	1	1	2	3	16
Kampong Chhnang [d]	80	2	3	8	1	1	-	-
Pursat [e]	50-65	2-4	10-15	2-6	2-4	4-6	1-2	10-15
Kampong Thom [f]	61	5.3	13.5	0.6	3.7	2.6	3	4

Source: [a] Hoornweg and Bhada-Tata (2012), [b] Seng, Fujiwara, and Seng (2018), [c] Sang-Arun et al. (2011), [d] Sethy (2017), [e] ADB (2019), [f] ADB (2018)

1.4.3. MSW treatment and disposal

The current management of MSW in Cambodia heavily relies on landfilling, as depicted in Figure 5. Recycling and composting, on the other hand, represent the smallest proportion in MSW treatment primarily due to the lack of source segregation. Source segregation, involving sorting out waste at the point of generation, is a fundamental method of waste separation that can contribute to various benefits, including reducing waste volume, recovering valuable resources, minimizing landfill size, and lowering costs associated with collection, transportation, and treatment (Sarkodie and Owusu 2021). In 2008, the government of Cambodia introduced the 3Rs (reduce, reuse, and recycle) approach through the National Strategy on 3Rs for Waste Management. However, the effectiveness of this approach seems to be limited due to the absence of regulations

and supporting mechanisms. The success of implementing the 3Rs largely depends on active participation and awareness of waste generators (Valkenburg et al. 2008).

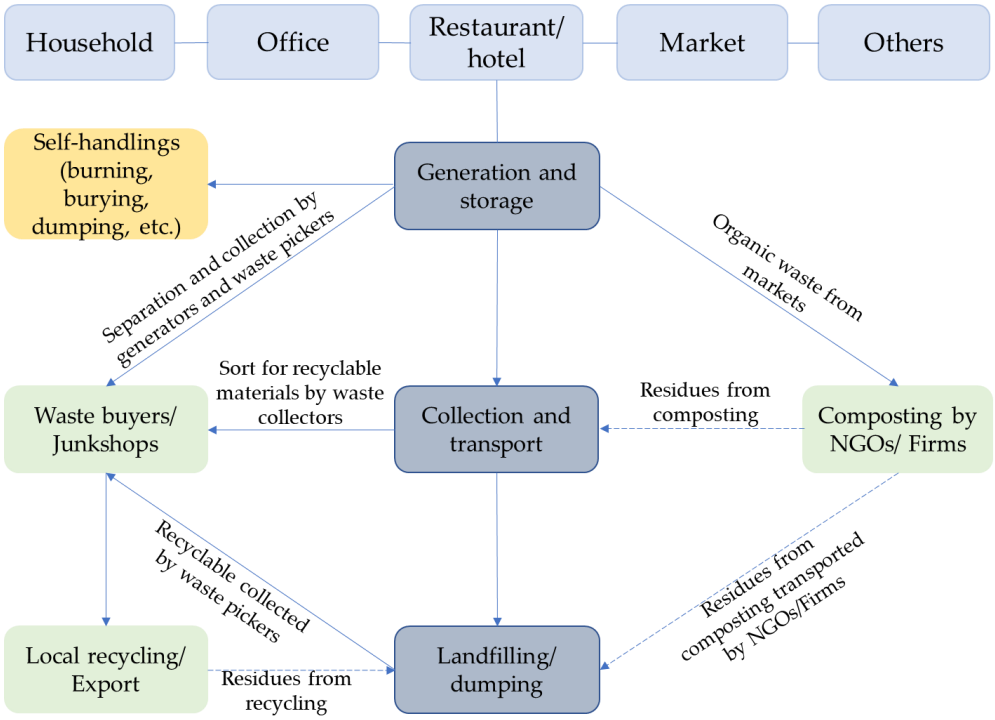


Figure 5. Diagram of MSW management stream in Cambodia

1.4.4. Open burning

Open burning of waste is a common practice in many rural areas where waste collection services are unavailable. Residents who lack access to waste collection services often resort to burning, burying, or illegally disposing of their waste on vacant land and water bodies (Rathana 2009). According to Kham and Heilmann (2015), open burning of uncollected waste accounts for a significant proportion of waste mismanagement (57%), followed by burying (11%), dumping on vacant lands (9%), disposal into water (5%), and other methods (9%). Open burning poses a substantial threat to environmental pollution. When waste materials such as plastics, rubber, and other materials are burned, they release harmful chemicals into the air, including particulate matter, carbon monoxide, nitrogen oxides, sulfur dioxide, dioxins, and furans. Furthermore, burning waste releases CO_2 and other GHGs, contributing to the warming of the planet and exacerbating climate change.

1.4.5. Recycling

Waste recycling in Cambodia is generally limited, and there is a reliance on the informal sector for recycling activities (Sothun 2012). The collection of recyclable waste occurs at various stages of

the MSW management process. At the source of generation, some waste generators sort out recyclable materials such as aluminum cans and glass bottles before temporarily storing them in designated bins or containers. However, this practice is limited, resulting in a significant amount of waste being disposed of without proper source separation. Waste pickers play a crucial role in recovering recyclable resources from waste at household bins, temporary storage containers, and landfill sites. During waste collection, workers often search for any remaining recyclable materials before transporting them to landfills. Recyclable materials recovered by waste generators, waste pickers, or waste collection workers are typically sold to waste buyers (known as Edjai) or directly to junkshops. Waste buyers travel around the cities with pushcarts, collecting and purchasing recyclable materials. It is estimated that around 3,000 waste pickers in Cambodia collect a significant amount of recyclable waste, which is then sold to waste buyers and junkshops. These materials are often export to other countries (EuroCham Cambodia 2019). This informal sector, including waste pickers, plays a crucial role in reducing the amount of waste sent to landfills.

1.4.6. Composting

Organic waste constitutes more than 50% of the total generated MSW; however, its potential value is largely untapped. Instead, organic waste is often mixed with other types of waste and sent to landfills. By converting organic waste into compost, up to 20% of the MSW can be utilized (Seng et al. 2018). Increasing composting activities can have a significant impact on reducing environmental pollution caused by GHG emissions and help address waste management issues (Seng et al. 2013). Unfortunately, there has been limited effort to compost food waste due to several factors. These include low public awareness of organic fertilizer, insufficient landfill space for composting facilities, a lack of waste segregation, and limited technology and human resources for composting (Seng et al. 2013). Furthermore, the preference for chemical fertilizers over compost by the people contributes to an unstable compost market (Rathana 2009).

1.4.7. Landfill and disposal

The primary method of waste disposal in Cambodia is landfilling, and unfortunately, most of the landfills in the country are unsanitary open dumpsites, lacking essential components such as soil cover, leachate treatment, and gas control systems. This inadequate management and disposal of MSW have had negative impacts on the environment, human health, and have contributed to global warming through GHG emissions (Abdel-Shafy and Mansour 2018). To address these challenges and ensure the effective management of landfill sites, the government of Cambodia established a state-owned company called the Enterprise for Managing Transfer Stations and Landfills for Solid

Waste (EML) in late 2020. This company is tasked with constructing and developing transfer stations, landfills, and other treatment infrastructures such as resource recovery facilities, recycling facilities, and WTE incineration plants across the country.

Currently, large-scale technology for the treatment of MSW is not available in Cambodia. In urban cities and districts with limited access to waste collection services, small-scale incinerators without energy recovery have been installed. As of 2021, there were 54 incinerators installed throughout the country, with burning capacities ranging from 1 to 8 tons per hour. However, MSW in Cambodia is typically mixed without pre-processing and is dominantly composed of organic materials with high moisture content and low net calorific value, which require auxiliary fuel for combustion. Incomplete burning waste in the incinerators can result in the production of harmful substances such as carbon monoxide, dioxin, and other harmful substances.

To promote the development of WTE, the government has encouraged the private sector to explore business opportunities focused on energy recovery from waste. However, the high cost associated with traditional WTE incinerators, coupled with low waste collection rates and scattered waste disposal practices, pose challenges for implementing WTE projects. Among the cities, Phnom Penh has been identified as a promising city due to its high volume of waste generation. A study conducted by the Global Green Growth Institute indicated that a WTE plant in Phnom Penh could generate up to 10 MW of energy (GGGI 2020). The study also suggested a Feed-in-Tariff (FIT) rate of 0.10 USD/kWh, along with a gate fee of 18 USD/tMSW. However, the current landfill gate fee in Phnom Penh is only 0.7 USD/tMSW, and the FIT for renewable energy for biomass-fired plants in the country is from 0.095 to 0.120 USD/kWh (Sokrethya et al. 2023).

1.5. Policy and strategy in MSWM in Cambodia

Numerous regulations, policies, and guidelines have been implemented and adopted to improve MSW management in the country, as depicted in Figure 6. According to the Law on Natural Resource Management and Environmental Protection (1996), the Ministry of Environment is tasked with developing regulations and guidelines and overseeing waste management, including hazardous, industrial, and medical wastes (RGC 1996). In 1999, sub-decree No. 36 on Solid Waste Management entered into force. Under this sub-decree, solid waste is categorized into garbage, solid waste, and hazardous waste (RGC 1999). The responsibility for collecting, transporting, storing, recycling, reducing, and disposing of garbage and hazardous waste lies with the municipality and city authorities. Importing all types of waste from other countries is strictly prohibited in accordance with the regulations.

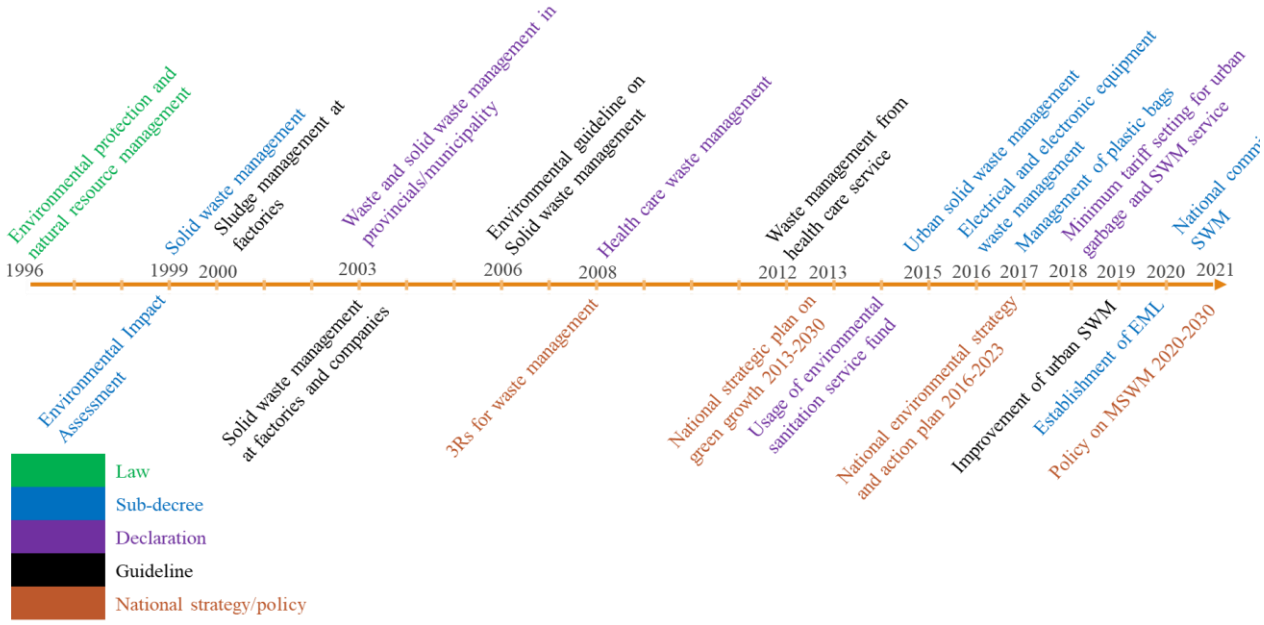


Figure 6. MSW management related legislation, policies, and guidelines

Sub-decree No. 168 on the Management of Plastic Bags was enacted in 2017 to promote the use of biodegradable plastics and the public participation in reducing plastic bag consumption through reusing plastic bags or adopting eco-friendly alternatives (RGC 2017). This sub-decree specifically regulates the importation, production, distribution, and use of plastic bags. Only plastic bags that meet the minimum criteria of 0.03 mm thickness and a base width larger than 25 cm are allowed to be imported and produced locally. Supermarkets and shopping centers are required to charge customers 0.10 USD/bag.

1.6. Problem statements

Over the past decade, there has been a significant increase in the quantity of MSW in Cambodia. The management of this waste has become a major concern, especially in Phnom Penh municipality, which contributes approximately 25% of the total MSW generated. The lack of treatment infrastructure, technological capabilities, and sanitary landfills has resulted in environmental pollution and adverse effects on human health. Landfills are the primary method of waste disposal, and improper management practices such as insufficient soil cover, leachate treatment, and gas control systems contribute to global warming through the release of landfill methane and contamination of surrounding soil and groundwater via leachate penetration. Open burning of waste, often employed to reduce waste volume, releases harmful pollutants into the air, including carbon monoxide, particulate matter, and toxic chemicals. Additionally, the lack of recycling and composting facilities leads to the loss of valuable resources and the unnecessary

disposal of organic waste in landfills. Despite the critical nature of this issue, there is a lack of comprehensive research on the current extent of environmental pollution caused by MSW management in the country. Therefore, there is an urgent need to investigate the scope of environmental pollution arising from MSW management and develop effective strategies to minimize the environmental burden associated with MSW.

1.7. Research objectives

This study aims to assess environmental pollution resulting from the MSW landfilling and determine a mitigation strategy by optimizing resource recovery. To achieve this goal, the specific objectives of this study are to:

1) Assess the environmental pollution from the landfill site, considering heavy metals contamination and GHG emissions.
2) Quantify the GHG emissions from different MSW management strategies with the potential to reduce environmental burdens of landfill.
3) Evaluate the feasibility of WTE technologies and their contribution to GHG emissions reduction.

By addressing these objectives, the following research questions have been taken into account:

1) Is there heavy metal leakage through leachate that is affecting the surrounding environment of the landfill?
2) How do current MSW landfill management practices and landfill gas emissions contribute to the potential for global warming?
3) What strategy can be implemented to minimize the negative environmental impacts of landfilled MSW and optimize resource recovery?
4) Which energy recovery technology is the most suitable for Phnom Penh in terms of energy generation, environmental pollution reduction, and economic viability?

1.8. Conceptual Framework

Landfills play an important role in MSW management and are considered sources of pollution. When MSW is deposited in a landfill, it begins to decompose and produces LFG, which contains mostly CH_4 and CO_2. LFG is a potent greenhouse gas that contributes significantly to climate change when released into the atmosphere. CH_4, in particular, is a potent GHG that contributes significantly to climate change when released into the atmosphere because it is estimated to be 25

times more effective at trapping heat than CO_2 over a 100-year period. In addition, the decomposition of the MSW produces leachate, containing pollutants such as heavy metals, organic compounds, and pathogens. Leachate can contaminate groundwater and nearby surface water sources if not properly managed, potentially contaminating drinking water sources and harming aquatic life. Landfills can also contaminate soil with hazardous chemicals and heavy metals, which can impact soil quality and the health of plants growing in the area (Figure 7).

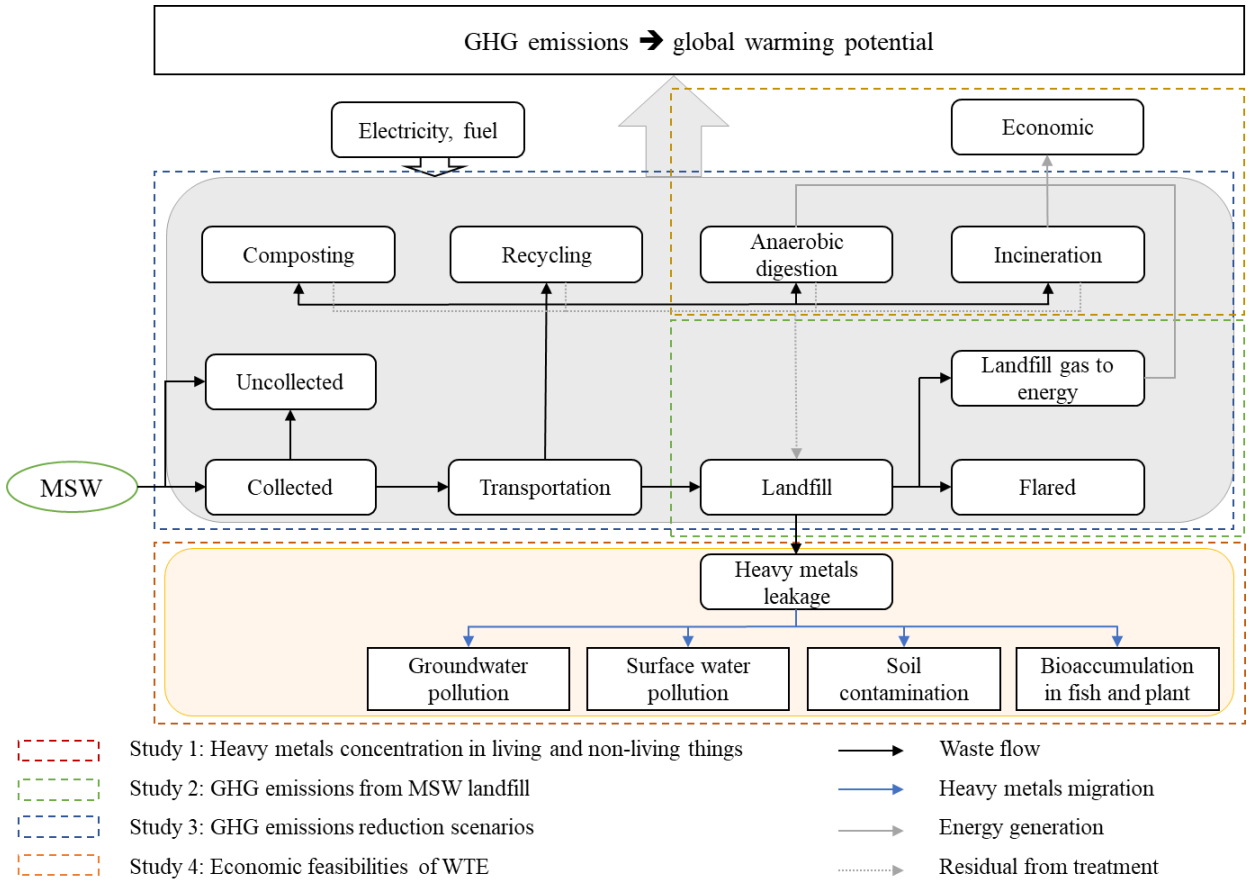


Figure 7. Problem associated with MSW and study framework

Numerous strategies have been formulated to address the environmental pollution emanating from MSW landfills. These encompass the implementation of effective waste management practices, including waste generation reduction, recycling, biological and thermal treatments, as well as the establishment of well-designed, operated, and maintained landfill facilities. Notably, energy recovery from waste through means such as biogas and thermal power generation plays a vital role in promoting sustainable waste management practices and facilitating the transition towards a circular economy. These technological approaches yield significant advantages. Primarily, they diminish the dependence on landfilling as a disposal method for MSW, thereby mitigating the environmental pollution associated with landfills. Furthermore, these technologies facilitate the

conversion of waste into valuable resources including heat and electricity, thereby contributing to the generation of clean and renewable energy. Consequently, there is a reduction in the reliance on finite fossil fuels, and an emphasis on harnessing waste as a potential energy source. Therefore, the incorporation of WTE technologies assumes utmost significance in fostering sustainable waste management practices, advancing energy generation, promoting resource recovery, and safeguarding the environment.

1.9. Research scope and approach

Phnom Penh, the capital city of Cambodia, has been selected as the focus of this study, with a specific emphasis on landfilled MSW. Other waste types and uncollected waste are excluded from consideration within this study. In pursuit of the aforementioned research objectives, four approaches were carried out. Firstly, laboratory tests were conducted on samples of surface water, groundwater, soil, plants, and fish collected from the vicinity of the landfill site to investigate potential heavy accumulation influenced by landfill leachate. Secondly, mathematical quantification models were employed to assess and quantify GHG emissions arising from landfill management practices and LFG emissions. The purpose of this analysis was to gain insights into the landfill site's contribution to global warming. Thirdly, in light of the environmental impacts of the landfill site, scenarios for MSW management were evaluated, with the aim of utilizing resources from waste and mitigating environmental pollution. Lastly, the economic feasibility of WTE technologies was rigorously assessed to investigate the potential for energy recovery, environmental protection, and economic benefits within the context of the study.

1.10. Structure of the dissertation

This doctoral dissertation is structured into six chapters, each dedicated to different aspects of the research topic. Chapter 1 serves as an introductory section, providing an overview of the research and emphasizing its significance. In Chapter 2, an in-depth investigation is conducted into the accumulation of heavy metals in water, soil, and living organisms in the vicinity of the landfill. Chapter 3 focuses on quantifying GHG emissions resulting from MSW landfilling. Chapter 4 critically evaluates the potential impacts of various alternatives for MSW management. Chapter 5 delves into an exploration of the energy, economic, and environmental benefits associated with WTE technologies. Lastly, Chapter 6 comprises the general discussion, conclusion, and recommendations drawn from the research findings.

CHAPTER 2

INVESTIGATION OF HEAVY METALS ACCUMULATION IN WATER, SOIL, AND LIVING ORGANISM IN THE LANDFILL VICINITY

2.1. Introduction

Landfill is the primary method for MSW management in Phnom Penh municipality, Cambodia. The landfill receives various types of MSW, including waste generated from households, markets, commercial areas, street sweeping, slaughterhouses, and occasionally industrial and construction and demolition wastes. MSW often contains both non-hazardous and hazardous waste, such as battery waste, painting waste, and electrical and electronic equipment waste (E-waste). Due to limitations in waste separation at the source and the absence of material recovery facilities, a significant amount of comingled waste is disposed of at the landfill. Under anaerobic conditions in the landfill, leachate is generated as biodegradable waste breaks down and decomposes. Leachate infiltration is the main mechanism through which contaminants migrate from landfills into the surroundings. Leachate generation and migration occur during landfill operation and may persist even after the landfill is closed due to the natural decomposition of waste. Precipitation percolates through the landfill, carrying dissolved substances that pass through the soil and eventually reach groundwater or water bodies if the landfill was not adequately equipped with a leachate collection system (Iravanian and Ravari 2020; Makuleke and Ngole-Jeme 2020; Sanga, Fabian, and Kimbokota 2022; Vaverková 2019). The penetration of leachate into the surrounding environment is also associated with soil textile, while the ability of pollutants to disperse is a function of soil mineralogy, organic matter content, cation exchange capacity (CEC), and pH (Makuleke and Ngole-Jeme 2020; Oyediran, Olalusi, and Jimoh 2020). Clay soil is commonly used as a barrier and liner material in constructed landfills to prevent leachate from seeping into the soil and groundwater (Oyediran et al. 2020).

The characteristics of leachate discharged from landfills are influenced by a multitude of factors, encompassing the quantity of waste, waste composition, moisture content of waste, precipitation, climate conditions, waste compaction, geological characteristics, and landfill age (Abiriga et al. 2021; Vaverková 2019). Typically, landfill leachate contains a diverse array of pollutants, including dissolved organics such as chemical oxygen demand (COD), total organic carbon (TOC), and biochemical oxygen demand (BOD), inorganic macro components like calcium, magnesium, sodium, potassium, ammonium, iron, chloride, sulfate, and hydrogen carbonate), heavy metals,

and xenobiotic organic compounds like benzene, toluene, ethylbenzene, xylene, tetrachloroethylene, and trichloroethylene (Kjeldsen et al. 2002; Vaverková 2019; Vodyanitskii 2016). Of particular concern are heavy metals and xenobiotic compounds, as they have garnered widespread global attention due to their harmful impacts on both human health and the environment (Makuleke and Ngole-Jeme 2020).

Heavy metals present in leachate are recognized as the most serious pollutants owing to their high concentrations, which have the potential to contaminate soil, groundwater, and surface water upon release into the natural environment (Abdel Gawad 2018). Once heavy metals infiltrate soil or water, they can be absorbed by plants and aquatic organisms, accumulating in the food chain (Xu et al. 2017). This accumulation poses a significant risk to human health due to the inherent toxicity of heavy metals (Kassim et al. 2022). Moreover, heavy metals can persist in the environment for extended periods, continuing to contaminate water and soil even after the landfill has ceased operation. Among the most problematic metals found in leachate and the surrounding environment of the landfill are Cadmium (Cd), Chromium (Cr), Copper (Cu), Lead (Pb), Nickel (Ni), and Zinc (Zn) (Jaishankar et al. 2014; Vongdala et al. 2019). Certain heavy metals, such as Cd, Cr, and Pb, can exhibit high toxicity to water, even at low concentrations, significantly impacting the aquatic ecosystem, the food chain, and human health (Chu et al. 2019). On the other hand, Cu and Zn are essential nutrients required in small amounts for human, animal, and plant health. However, excessive intake or exposure to Cu and Zn can lead to toxicity (Rweyemamu et al. 2020).

Previous investigation have extensively documented the concentrations of heavy metals in soil, surface water, groundwater, and plants in the landfill site and its surrounding area (Hredoy et al. 2022; Hussein et al. 2021; Vongdala et al. 2019). Vongdala et al. (2019) observed elevated levels of Pb and Cr in groundwater during both wet and dry seasons, surpassing the permissible standards set by ANES and WHO. Additionally, Cd and Cu levels in the soil samples exceeded the Dutch Pollutant Standards. Their study also reported substantial accumulation of Cd, Cr, Pb, and Zn in the edible parts of *Ipomoea aquatica,* ranging from 5 to 86 times higher than the WHO standards. Similarly, Hredoy et al. (2022) reported elevated levels of heavy metals in soil, surface water, groundwater, and plant samples collected from the vicinity of the landfill. The concentration of Cd, Cr, and Pb in plant samples amounted to 0.4, 2.26, and 8 mg/kg, respectively, exceeding the allowable limits. Moreover, the researchers identified that samples collected closer proximity to the landfill exhibited significantly higher concentrations of heavy metals. A separate investigation conducted by Hussein et al. (2021) also revealed elevated levels of Cd, Cr, Cu, Pb, and Zn in soil

samples when compared to background values. The contamination of water sources with heavy metals from leachate poses a considerable threat to both the water body and the aquatic organisms inhabiting it (Agarwal, Kumar, and Behari 2007). Fish, in particular, play a crucial role in indicating the health of aquatic ecosystems and evaluating the potential risk of heavy metal pollution for human consumption. In this context, Hossain et al. (2018) examined the accumulation of heavy metals in fish samples collected from a leachate-treated pond and found concentrations of Cd and Ni exceeding the WHO-recommended limit.

Exposure to heavy metals leaching from landfills presents a substantial threat to both the environment integrity and public health. The Dangkao landfill, situated in a low-lying area, is prone to frequent inundation, especially during the wet season. The absence of a leachate treatment system increases the risk of toxic substances seeping out from the landfill. This contamination can spread throughout the soil surface, potentially impacting the surrounding ecosystem. A study conducted by Xaypanya et al. (2018) investigated heavy metal contamination in leachates, revealing higher concentrations of Cd, Cr, and Pb exceeding permissible standards, with the exception of Zn. However, their study did not account for the potential migration of these heavy metals to the surrounding environment. Therefore, it is crucial to assess the concentration of heavy metals in the nearby ecosystem, including surface water, groundwater, soil, plants, and fish, given their toxic effects and potential transfer into the human body through the food chain. This study represents the first empirical investigation of its kind in Cambodia and holds the potential to inform future developments in MSW landfills within the country.

2.2. Materials and Methods

2.2.1. Study area and description

Phnom Penh municipality is the most densely populated city of Cambodia, occupying a land area of 679 km^2 and housing a population of 2.28 million people as of 2019 (National Institute of Statistics 2020). The city experiences an average annual precipitation of 1,550 mm, with the lowest amount occurring in February (9 mm), and the highest in September (255 mm). Located in the southern part of Phnom Penh at coordinates 11º28’29” N and 104º53’11” E, the Dangkao landfill is one of the largest waste disposal sites in Cambodia (Figure *8*). It serves as the destination for MSW disposal from 14 districts (khans) within the municipality. Initially surrounded mostly by agricultural lands, the areas surrounding the landfill has witnessed the development of residential areas and factories due to urbanization. The landfill commenced operations in August 2009 and initially received approximately 1,200 tMSW/day. Over time, this amount has increased to 3,530

tMSW/day in 2022. The landfill is divided into two areas, Area A-B and C-D, with pit depths of 10 m and 30 m below the ground surface, respectively (Xaypanya et al. 2018). Areas A-B have been closed and covered with soil since February 2016, while Area C-D was opened after the closing of Areas A-B to accommodate additional waste disposal. It was estimated that the landfill produces 356,800 m^3 of leachate (Xaypanya et al. 2018), which is stored in a lagoon for recirculation.

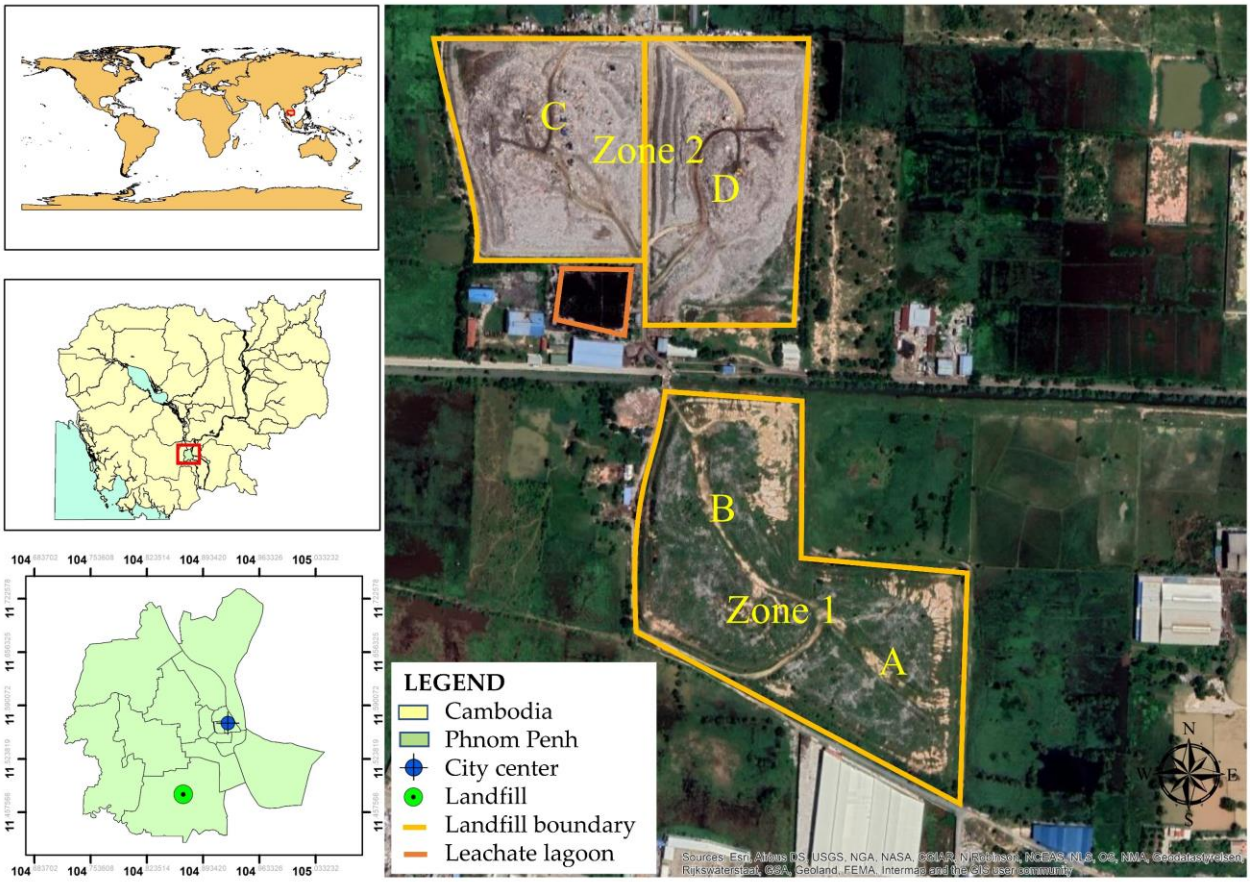


Figure 8. Location of Dangkao landfill

2.2.2. Samples collection

Samples of surface water, groundwater, soil, plants, and fish were randomly collected from the landfill site and its adjacent vicinity during two different periods: the dry season (DS) in February 2022 and the wet season (WS) spanning from June to July 2022. The sampling locations are highlighted in Figure 9. Prior to sample collection, all equipment and plastic containers used were cleaned with acid and rinsed with deionized water to ensure the reliability of the results. A total of 5 samples of surface water, 3 samples of groundwater, 5 samples of soil, 3 types of plant species, and 4 different fish species were collected in each season.

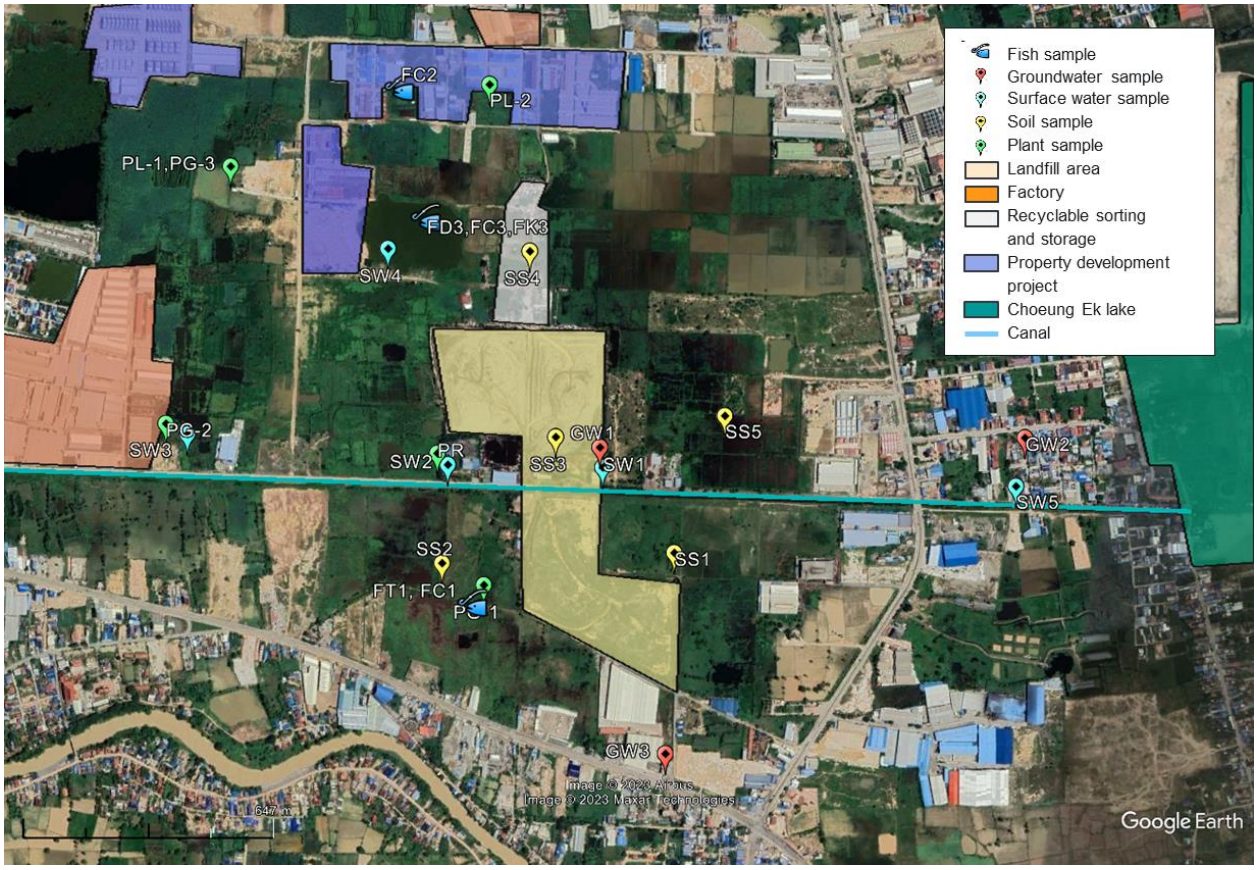


Figure 9. Sampling locations

2.2.3. Sample preparation

Surface water samples were collected in triplicate from the canal and surface water ponds. Samples were collected at a depth of 1-1.5 meters from each source and combined into a single composite sample for each sampling site. Groundwater samples were obtained from one borehole within the landfill site and 2 boreholes in nearby villages used by locals for gardening, bathing, washing, and sometimes for cooking. All samples were stored in 1-L polyethylene bottles and labeled SW for surface water and GW for groundwater, with consecutive order numbers assigned for each location. GPS coordinates were recorded for each sampling point. On-site measurement for pH, temperature, Electricity Conductivity (EC), and Dissolved Oxygen (DO) was conducted using the HORIBA (U-50 Multiparameter Water Quality Meter, Kyoto, Japan). The samples were acid-preserved with HNO_3 to reach a pH <2 before being transported in an ice-cool box to the laboratory, where they were refrigerated at 4 °C (Sanga et al. 2022). Table 3 provides a summary of the samples and the location of each sampling site.

Table 3. Sampling information and its locations

Site	Latitude	Longitude	Distance from landfill (m)	Description of samples
Surface water				
SW-1	11°28'59.02"N	104°53'17.29"E	30	Surface water in the middle stream of canal
SW-2	11°28'58.99"N	104°53'15.28"E	100	Surface water in the upstream of canal
SW-3	11°29'2.82"N	104°52'39.93"E	750	Surface water in the landfill vicinity and close to garment factories
SW-4	11°29'17.44"N	104°52'58.97"E	300	Surface water in the landfill vicinity and close to property development project
SW-5	11°28'57.31"N	104°53'52.78"E	1100	Surface water at the downstream of canal
Groundwater				
GW-1	11°29'0.71"N	104°53'17.06"E	0	Groundwater collected from borehole used by landfill office for cleaning and bathing
GW-2	11°29'1.38"N	104°53'53.57"E	1100	Groundwater collected from borehole in the nearby village used for watering plants
GW-3	11°28'34.96"N	104°53'22.58"E	300	Groundwater collected from borehole in the nearby village for daily consumption
Soil				
SS-1	11°28'51.83"N	104°53'23.40"E	30	Soil on the edge of landfill boundary
SS-2	11°28'51.04"N	104°53'3.50"E	200	Soil in inundation area close to the landfill site
SS-3	11°29'1.59"N	104°53'13.27"E	0	Soil near the landfill office
SS-4	11°29'17.19"N	104°53'11.11"E	150	Soil in the vacant land close to the landfill site
SS-5	11°29'3.32"N	104°53'27.79"E	330	Soil in the paddy field nearby the landfill site
Plant				
PG-1	11°28'49.19"N	104°53'7.07"E	70	*Ipomoea aquatica* in an adjacent lake nearby the landfill site
PG-2	11°29'2.82"N	104°52'39.93"E	740	*Ipomoea aquatica* grows in the landfill vicinity and close to garment factories
PL-1	11°29'24.37"N	104°52'45.53"E	640	*Nelumbo nucifera* grows outside the landfill site
PL-2	11°29'2.82"N	104°52'39.93"E	740	*Nelumbo nucifera* planted outside the landfill site and close to property development project
PR-2	11°29'0.25"N	104°53'3.13"E	110	*Oryza sativa* grows nearby the landfill site
PG-3	11°29'24.37"N	104°52'45.53"E	640	*Ipomoea aquatica* grows outside the landfill site
Fish				
FT1	11°28'49.09"N	104°53'8.12"E	70	*Channa striata* in an adjacent lake nearby the landfill site
FC1	11°28'49.09"N	104°53'8.12"E	70	*Trichopodus trichopterus* in an adjacent lake nearby the landfill
FC2	11°29'17.44"N	104°52'58.97"E	550	*Trichopodus trichopterus* in the pond nearby the property development project
FD3	11°29'17.44"N	104°52'58.97"E	200	*Oxyeleotris marmorata* in the pond nearby the landfill and property development project
FK3	11°29'17.44"N	104°52'58.97"E	200	*Trichopsis vittata* in the pond nearby the landfill and property development project
FC3	11°29'17.44"N	104°52'58.97"E	200	*Trichopodus trichopterus* in the pond nearby the landfill

Following Vongdala et al. (2019), soil samples were collected from the surface layer (0-0.25 m depth) in an area of 50 m^2 at each location in and around the landfill. Five random sub-samples were taken from each site to create a composite sample weighing 0.5 kg, which was then placed in sterilized plastic bags. The soil samples were air dried at room temperature and then ground to a fine powder using mortar and pestle. The powdered soil samples were filtered using a 0.2 mm sieve filter and stored in the dark at 4 °C for analysis.

Plant samples, including *Ipomoea aquatica* (3 locations)*, Oryza sativa* (1 location)*, and Nelumbo nucifera* (2 locations)*,* were collected in triplicate from the surrounding areas of the landfill and placed into sealed polyethylene bags. A total of 36 samples were collected in two seasons. The plant samples were washed with tap water and then rinsed with distilled water for one minute to remove soil particles. They were then separated into roots, stems, and leaves and dried at 40 °C for 96 hours in an oven until a consistent weight was maintained. The dried samples were ground into powder using a mortar and pestle, placed in zippered polyethylene bags, and stored in a dark environment for further analysis.

Channa striata
(Striped snakehead)

Trichopodus trichopterus
(Three spot gourami)

Oxyeleotris marmorata
(Marble goby)

Trichopsis vittate
(Croaking gourami)

Figure 10. Photo of fish samples

Four fish samples (*Channa striata, Trichopodus trichopterus, Oxyeleotris marmorata, and Trichopsis vittate)* were collected from three ponds located at distances ranging from 70 and 550 meters away from the landfill site in both the DS and WS, resulting in a total of 64 fish samples. The fish were caught using fishing nets and immediately placed in air-sealed plastic bags for transportation in an ice-cool box to the laboratory. In the laboratory, the fish samples were washed several times with deionized water. For *Channa striata* and *Oxyeleotris marmorata*, the edible parts (flesh and skin) were cut into small pieces, while *Trichopsis vittata* and *Trichopodus trichopterus* were dried in their entirety due to their small size. To obtain the appropriate amount for analysis, 7 samples of *Trichopsis vittata* were collected from each location and combined as a composite one sample. All samples were dried at 70 °C for 22 hours until a constant weight was reached and then ground into a powder.

2.2.4. Heavy Metals Analysis

The surface water and groundwater samples were filtered using Ø 90 mm filter papers (Qualitative Paper Filter Advantec, no.1) to obtain a 50-mL solution. To this solution, 0.5 mL of 65% HNO_3 was added, and it was heated on a hot plate for 2 hours at 80–90 °C without boiling (Vongdala et al. 2019). After cooling to room temperature, the samples were filtered with 0.2-µm syringe filters. For the soil samples, 5 grams of soil were mixed with 20 mL of 7 mol/L HNO_3, stirred for an hour, and autoclaved at 120 °C for 30 minutes. After cooling, the mixture was filtered with filter papers. The filtrate was then diluted with deionized water to obtain a 50 mL solution and filtered again through a 0.2-µm syringe filter.

In the case of plant samples, 0.5 grams of each powdered sample were digested with 8 mL of conventional aqua regia (65% HNO_3:30% HCl) and left overnight at room temperature (Rashid et al. 2016). The mixture was then heated on a hot plate at 100 °C for 4 hours. The digested samples were transferred to a 50-mL volumetric flask using filter paper and diluted with deionized water. The resulting solutions were filtered again through a 0.2-µm syringe filter. For each fish sample, 0.5 grams were placed into capped vials, and 2.5 mL of 65% HNO_3 was added. The vials were left at room temperature for 12 hours to allow the reaction to complete before heating on a hot plate for 4 hours at 100 °C. The digested mixture was then filtered and diluted with deionized water in the same way as soil and plant samples. All sample solutions were stored in a refrigerator at 4 °C for analysis.

In the determination of heavy metals, the present study utilized an inductively coupled plasma optical emission spectrometer (ICP-OES, SPS 3000) manufactured by Hitachi, Japan. The ICP

multi-element standard solution IV from Sigma-Aldrich, Germany was used, and calibration curves were developed for analyzing various heavy metals (Figure 11).

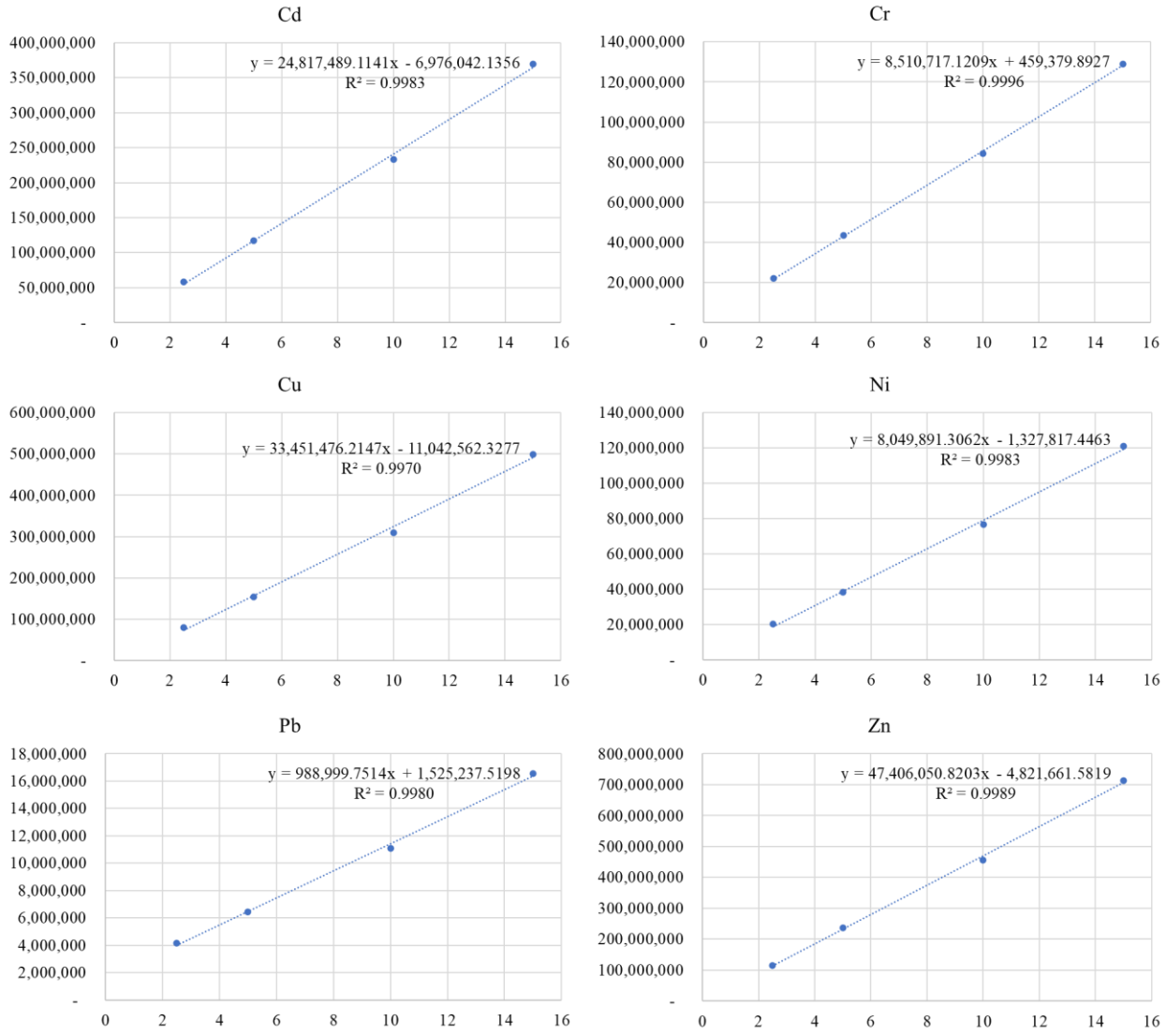


Figure 11. Standard curve for final concentration calculation

2.2.5. Data Analysis

The study employed the Statistical Package for Social Science (SPSS) version 23 to obtain descriptive statistics, such as the mean and standard deviation, for heavy metal concentrations in the samples. ANOVA with Tukey's test of significance was then used to compare the means of heavy metal concentrations between the wet and dry seasons. The statistical analyses were conducted with a 95% confidence interval, and significant differences were reported at a probability value of 0.05.

2.3. Results and discussion

2.3.1. Concentration of heavy metals in surface water and groundwater

In the WS, the pH of surface water ranged from 6.96 to 7.55, indicating normal conditions (Table 4). However, during the DS, notable increases in pH levels were observed, specifically 8.76, 10.00, and 8.02 for SW3, SW4, and SW5, respectively. These variations can be attributed to factors such as rainwater infiltration and dilution effects. Moreover, pH values can be influenced by the influx of contaminants from both natural sources and human activities, including percolation of leachates from leachate pond, land uses (Naveen, Sumalatha, and Malik 2018), and the industrial sector. The presence of high alkalinity values may be attributed to the disposal of ash and slag resulting from waste combustion in the landfill, as well as combustion of wood, agricultural resides, and peat, which can potentially lead to groundwater contamination (Naveen et al. 2018).

Table 4. Result of onsite measurement for surface water

Sample	pH		Temperature (°C)		EC (mS/cm)		DO (mg/L)	
	DS	WS	DS	WS	DS	WS	DS	WS
Surface water								
SW 1	7.48	7.36	30.95	30.48	9.51	0.90	2.86	8.94
SW 2	6.59	7.46	29.71	31.90	0.38	0.57	2.21	0.28
SW 3	8.76	7.03	35.67	30.80	0.98	0.30	1.96	5.71
SW 4	10.00	6.96	33.84	32.49	0.43	0.24	2.15	5.61
SW 5	8.02	7.55	32.99	32.36	9.60	1.10	2.43	6.07
Groundwater								
GW 1	6.77	6.77	30.56	30.33	1.36	1.36	4.34	8.77
GW 2	7.85	7.85	31.93	31.27	2.39	2.33	1.98	11.50
GW 3	7.25	7.75	31.37	31.80	0.96	0.81	2.14	11.29

The values presented in mean; DS: dry season; WS: wet season.

The pH, electrical conductivity (EC), and dissolved oxygen (DO) levels of the groundwater samples during both seasons were found to be within the range specified by the drinking water standards of Cambodia and the World Health Organization (WHO).

The concentrations of heavy metals in surface water samples collected from the landfill site and its surrounding area were analyzed. The measured concentrations of different heavy metals in surface water were found to fall within specific ranges: Cd (0.38–0.83mg/L), Cr (0.04–0.25 mg/L), Cu (0.49–0.64 mg/L), Ni (0.38–0.68 mg/L), Pb (ND–0.94 mg/L), and Zn (0.16–0.33 mg/L), as

presented in Table 5. Comparing the results with a leachate study by Xaypanya et al. 2018, this study revealed lower concentrations of Cr and Zn, while the Cd concentration was observed to be four times higher. Typically, the average concentration of Cd in surface water and groundwater is less than 1 µg/L (Faroon et al. 2012). Cd, known for its high bioaccumulation potential, is recognized as one of the most harmful and toxic heavy metals for aquatic life (Garai et al. 2021). Additionally, other heavy metals such as Cr, Ni, and Pb were also found to exceed the permissible standards. Another study conducted in natural water near the landfill also recorded higher level of Ni (Wieczorek et al. 2021). The elevated levels of heavy metals in surface water can likely be attributed to inadequate landfill management practices and the presence of toxic leachate from the unlinered leachate pond (Parvin and Tareq 2021). These contaminants have the potential to be transported during the WS, infiltrating both surface water and groundwater (Baziene, Tetsman, and Albrektiene 2020).

Table 5. Comparison of heavy metals concentration in surface water and groundwater (mg/L)

Author	Cd	Cr	Cu	Ni	Pb	Zn
Surface water (n=10)						
This study	0.38-0.83	0.04-0.25	0.49-0.64	0.38-0.68	ND-0.94	0.16-0.33
Xaypanya et al. (2018)	0.09-0.21	0.62-1.10	-	-	0.29-0.63	0.88-1.44
Vongdala et al. (2019)	ND	0.05-0.19	0.01-0.05	0.01-0.04	0.02-0.17	ND
Abu-Daabes, Qdais, and Alsyouri (2013)	0.40-1.23	1.27-6.92	0.5-2.9	0.24-2.88	-	-
Boateng, Opoku, and Akoto (2019)	0.49-1.08	0.70-1.92	1.31-1.98	-	1.52-3.57	1.72-6.09
Azim et al. (2011)	0.00-0.01	0.50-1.03	0.07-0.20	0.23-0.40	0.00-0.03	0.12-0.27
Sub-decree on water pollution control	0.05	0.05	1.00	0.20	0.10	2.00
Groundwater (n=6)						
This study	0.36-0.51	0.03-0.15	0.48-0.64	0.37-0.53	ND-0.75	0.15-0.23
Akinbile (2012)	-	0.25	ND	-	1.11-1.20	3.30-5.40
Alam et al. (2020)	ND	-	0.00-0.02	-	0.10-0.17	0.00-0.02
Azim et al. (2011)	0.001	0.05	0.01-1.00	0.01-0.02	0.05	0.40-5.00
Boateng et al. (2019)	0.00-0.03	0.00-0.08	0.01-0.25	-	0.01-0.09	0.06-0.61
Vongdala et al. (2019)	ND	0.02-0.08	ND-0.01	ND-0.01	ND-0.06	ND
Hredoy et al. (2022)	0.002	0.005	-	0.05	0.003	-
CDWQS	0.003	0.05	1.00	0.02	0.01	3.00

WHO standard	0.003	0.05	2.00	0.07	0.01	3.00

DS: dry season; WS: wet season; SD: standard deviation; ND: not detectable; the value expressed in mg/L; CDWQS: Cambodian Drinking Water Quality Standards; WHO: World Health Organization; values with bold indicated exceed the standard level.

Moreover, the concentrations of heavy metals in surface water may vary over time, influenced by factors such as the age of the landfill and variations in waste composition (Abiriga et al. 2021; Hosseini Beinabaj et al. 2023; Vaverková 2019). The concentrations of Cd, Cr, Cu, Pb and Zn in the surface water are comparable with findings in other studies, as shown in Table 5.

During the WS, lower concentrations of heavy metals were observed in SW2, SW3, and SW4, as depicted in Figure 12. This phenomenon may be attributed to the additional surface runoff that surface water bodies near the landfill site receive during this season, contributing to dilution effects and resulting in decreased concentrations of heavy metals (Naveen et al. 2018). Conversely, some heavy metals were found to be at higher levels during the DS in SW1 and SW5. This could be associated with the elevated content of certain heavy metals in MSW disposed of during this period (Vongdala et al. 2019). Furthermore, SW1 is situated in the middle stream of the canal, in close proximity to the leachate storage pond and the discharge from the landfill. Consequently, heavy metals present in leachate and the landfill site might be washed out by rainfall (Hossain et al. 2018) into the canal and subsequently flowed downstream to SW5. All the sites exhibited Pb concentrations exceeding the permissible discharge standards as stipulated in the sub-decree on water pollution control of Cambodia, except for SW2, where only traces of this element were detected during the DS. It is noteworthy that the vicinity of the Dangkao landfill is prone to flooding and inundation during the WS, which increases the risk of pollution in both surface water and groundwater due to the leachate pond and the landfill site (Parvin and Tareq 2021).

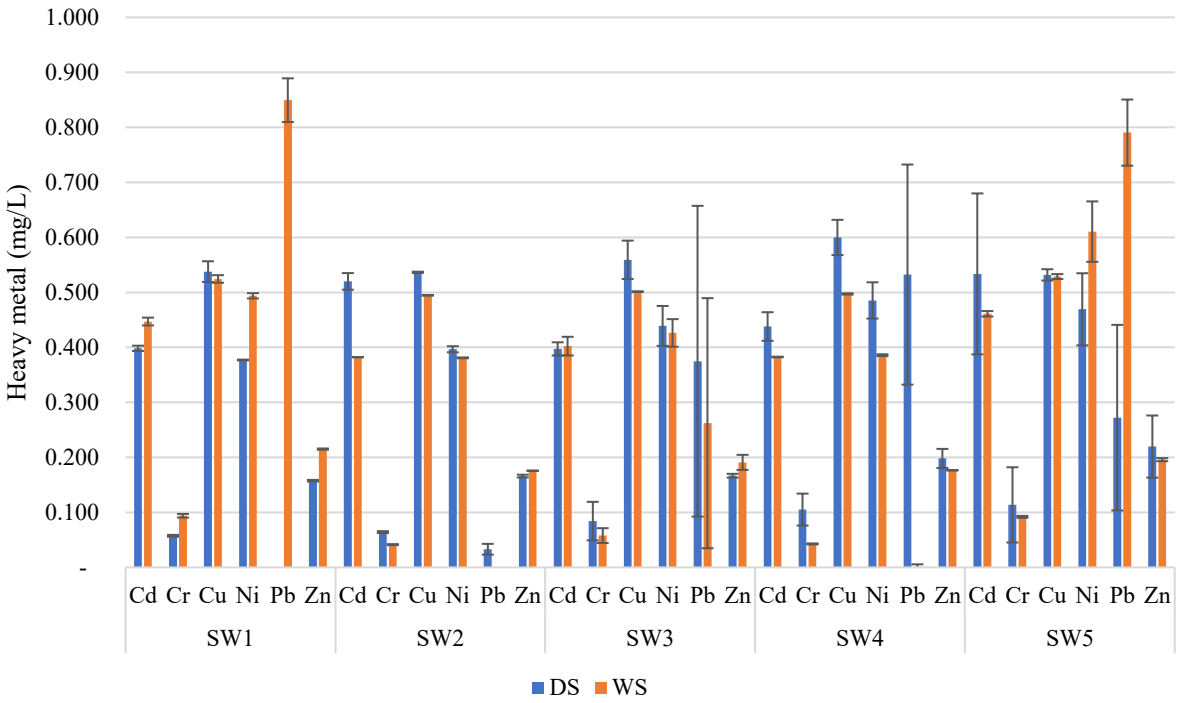


Figure 12. Heavy metal concentrations in surface water in the DS and WS

SW1, SW2, and SW5: surface water in the sampling sites located in the middle stream, upstream, and downstream of canal nearby the landfill; SW3 and SW 4: surface water outside the landfill used for agricultural purpose; DS: dry season; WS: wet season; bar values are mean ± Standard error; different letters are significantly different within the same sample ($p < 0.05$).

Figure 13 illustrates that the highest concentrations of heavy metals were detected in SW5, followed by SW1. The elevated concentration of heavy metals in SW5 can be attributed to its downstream location in a canal, which receives leachate from the landfill and discharges wastewater from residential and industrial areas. This mixture may accumulate before ultimately being discharged into Choeung Ek lake. It is important to note that Choeung Ek lake functions as a drainage system, receiving untreated sewage water and industrial wastewater before being discharged into the river. Previous studies have indicated that the inlet points to Choeung Ek lake exhibit elevated levels of toxic elements compared to the outlet point and other surface water (Holm, Marcussen, and Dalsgaard 2010).

Therefore, the high concentrations of heavy metals may have some influence from Choeung Ek lake, particularly during the WS when the lake is at its fullest and water recedes into the canal. Furthermore, SW1, located in the middle stream, exhibited the second-highest concentrations of Cd, Cr, Cu, Ni, and Pb followed by SW2 in the upper stream. This result confirms the potential leakage of contaminants from the landfill to the surrounding environment.

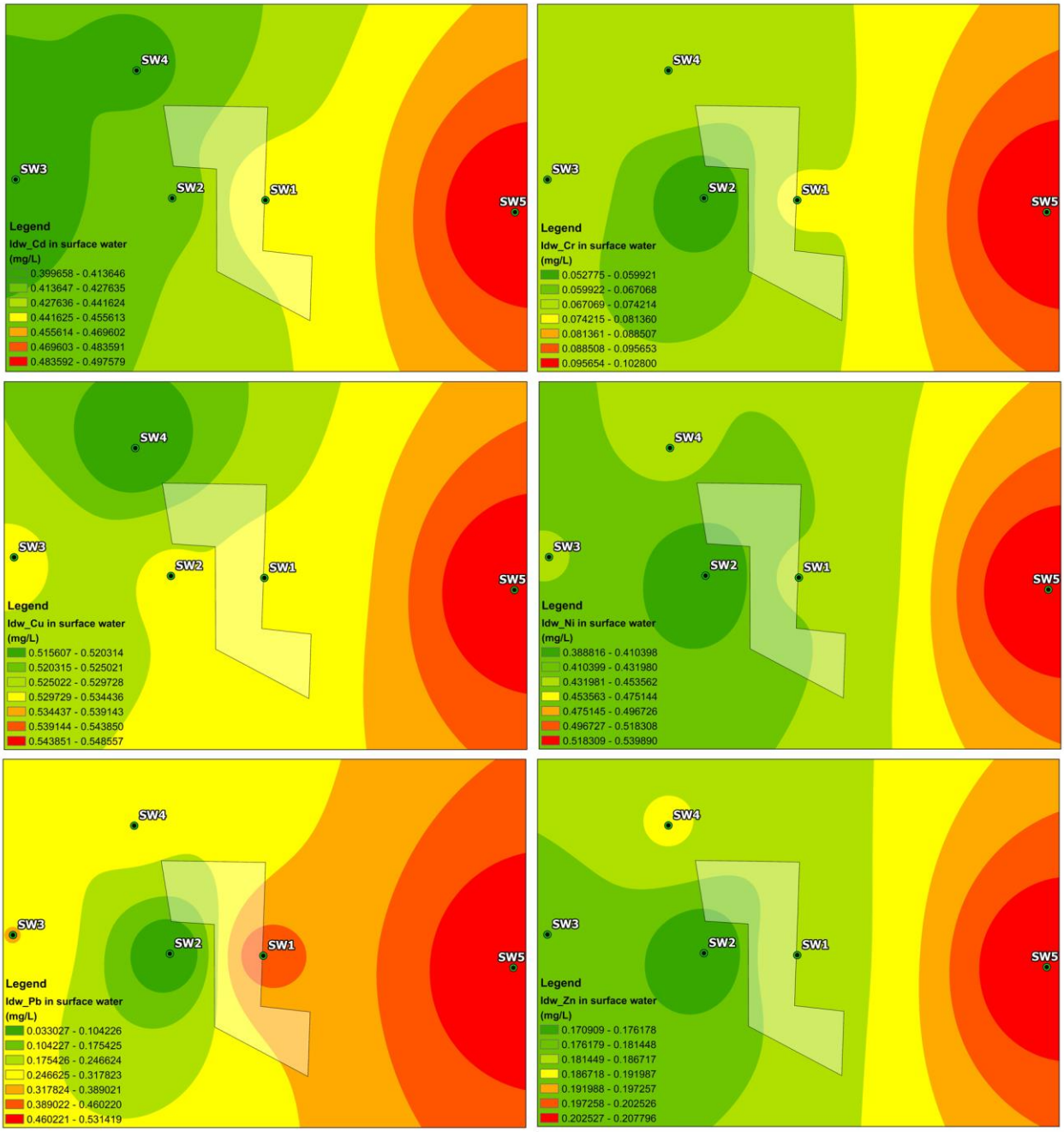


Figure 13. Heavy metal spatial dispersion map in surface water

Cd, Cr, Cu, Ni, and Zn were detected in all groundwater sampling sites, as shown in Figure 14. During the DS, heavy metal concentrations were higher compared to the WS, except for Zn in GW2 and GW3. The lower concentration during the WS can be explained by dilution effects caused by rainwater, which is consistent with a study conducted in Lao PDR (Vongdala et al. 2019). Meanwhile, the high Zn concentration in GW2 and GW3 during the WS may be associated with the higher concentration observed in SW1, SW2, and SW3 during the same period. Cu exhibited the highest concentrations among the detected heavy metals in groundwater but remained below the maximum limit set by the WHO. However, Cd and Ni concentrations in all groundwater

samples exceeded the WHO drinking water quality standard in both seasons. During the DS, Cr concentrations were slightly higher than the standard level. Notably, Pb concentrations were not detected in groundwater samples, although traces of Pb were found in surface water during the DS. This phenomenon may be associated with the low content of Pb in the MSW, which might be reduced through soil infiltration and plant absorption.

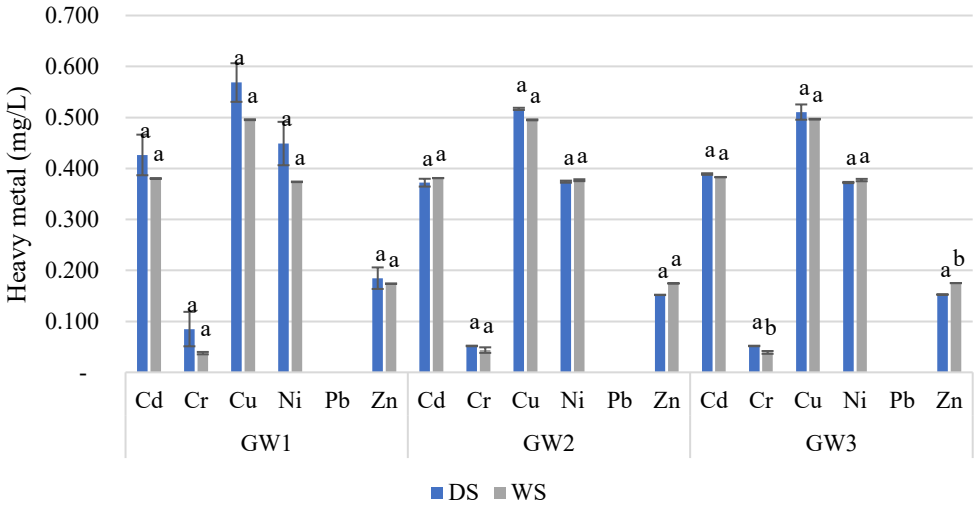


Figure 14. Heavy metal concentrations in groundwater

GW1: groundwater in the landfill; GW2: groundwater outside the landfill used for watering plant; GW3: groundwater outside the landfill used for cleaning, washing, and cooking; DS: dry season; WS: wet season; bar values are mean ± Standard error; different letters are significantly different within the same sample ($p < 0.05$).

Figure 15 illustrates the spatial distribution of heavy metals in groundwater. GW1, situated within the landfill site, demonstrated the highest concentration of heavy metals compared to groundwater samples collected from nearby villages. This can be attributed to the substantial depth of the landfill in Area C-D, approximately 30 m deep, which increases the risk of groundwater contamination through leachate infiltration (Xaypanya et al. 2018). A study conducted by Hredoy et al. (2022) also reported that locations closest to the landfill site exhibited the highest concentration of heavy metals. Although GW2 and GW3 showed lower concentrations, certain parameters, particularly Cd and Ni, exceeded the allowable limits of the WHO. This may potentially have adverse effects on human health when the water is utilized for drinking or cooking.

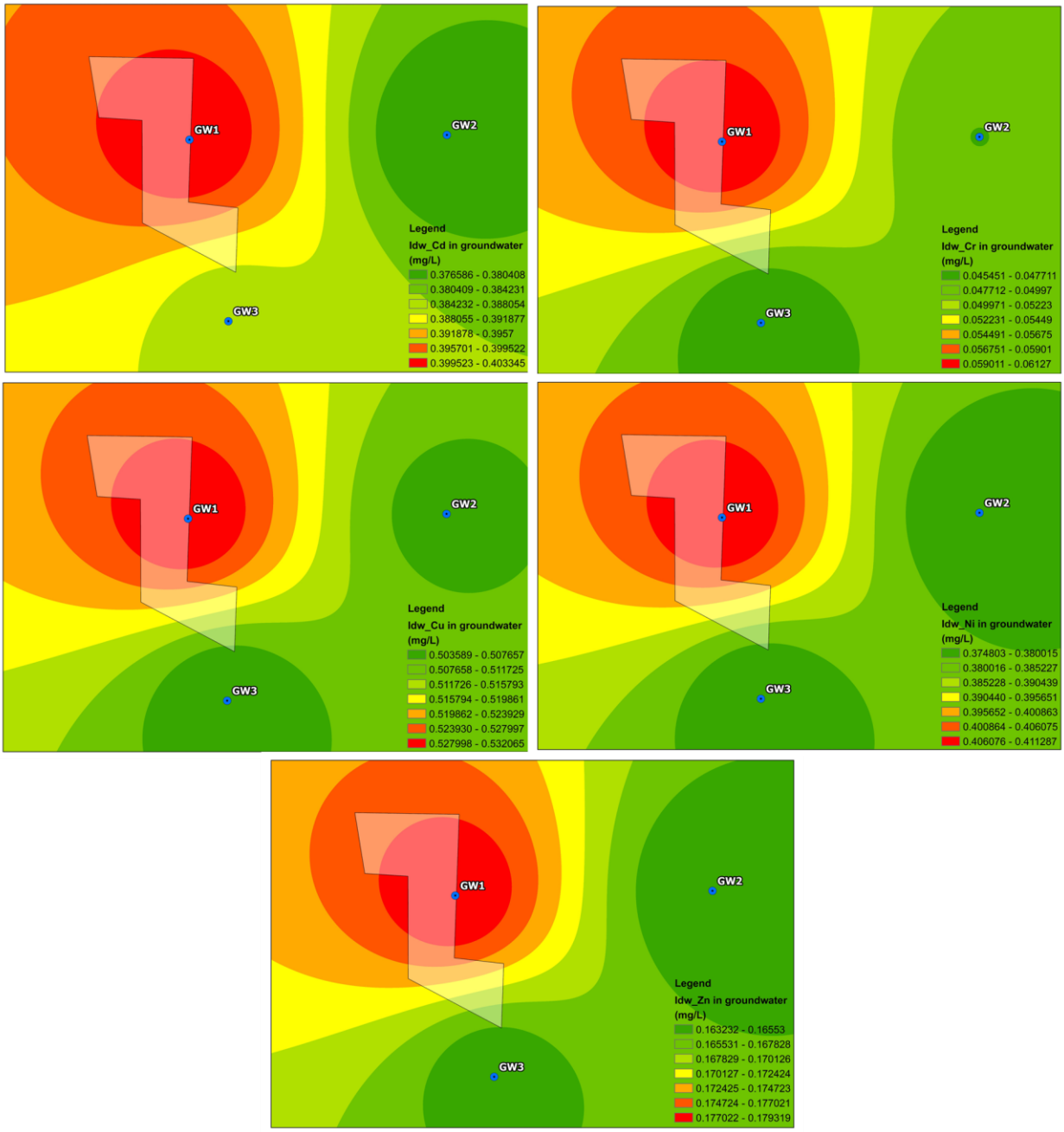


Figure 15. Heavy metal spatial dispersion map in groundwater

2.3.2. Concentration of Heavy Metals in Soil

Soil samples were meticulously analyzed to evaluate the environmental impact of the landfill on its surrounding area. The sampling sites were positioned within a 300-meter distance from the landfill. Most of the heavy metal concentrations during both seasons were found to be below the permissible standards for the disposal of toxic and hazardous substances set by the Ministry of Environment (MoE) in Cambodia, with the exception of Cd and Ni, as shown in Figure 16. Remarkably, the concentration of Cd was observed to be 51 times higher than the permissible standards of pollutants to be released into the environment. It also exceeded the intervention values

specified in the Dutch Soil Protection Act, signaling an environmental risk that necessitates remediation action (Bird et al. 2003). In comparison, a study by Xaypanya et al. (2018) reported lower levels of Cd (10–18 mg/kg) in soil in the vicinity of the Dangkao landfill. However, their study revealed higher levels of Cr and Zn in comparison to the present study. Cd has been recognized for its have high potential to accumulate in the environment, particularly in soil and sediments for several decades (Jaishankar et al. 2014). The elevated levels of Cd and other heavy metals in the vicinity of the landfill vicinity can be absorbed by vegetation, and when consumed, can have adverse effects on human and animal health (Arao et al. 2010; Vongdala et al. 2019). The concentrations of Cr, Cu, Pb, and Zn in soil are currently below the permissible limits. However, it is important to consider the potential for long-term accumulation of these metals, which could result in elevated levels in the future (Olafisoye, Adefioye, and Osibote 2013).

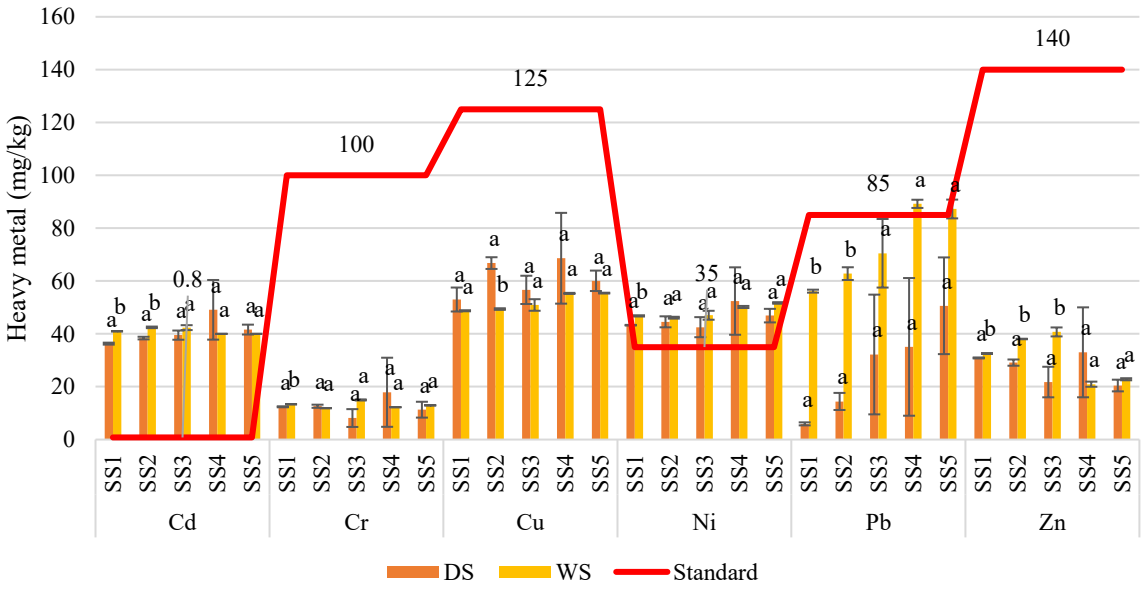


Figure 16. Heavy metal concentrations in soil.

SS1: soil sample in the rice field near the landfill; SS2: soil sample outside the landfill; SS3: soil sample in the landfill; SS4: soil sample outside the landfill; SS5: soil sample outside the landfill; DS: dry season; WS: wet season; bar values are mean ± Standard error; different letters are significantly different within the same sample ($p < 0.05$). Intervention Values (IV) of the Dutch Soil Protection Action: Cd (12); Cr (380); Cu (190); Ni (210); Pb (530); and Zn (720). The Intervention Values indicate that the functionality of soil for human, animal, and plant life is at risk of being severely compromised. Excessive concentrations beyond these intervention values can lead to significant contamination (Jayakumar et al. 2021).

Cd, known for its high toxicity, raises significant concerns regarding its release from sediment and its subsequent transfer to the food production system (Holm et al. 2010). In this study, the

concentration of Cd in the soil sample was found to be the highest compared to other studies, as indicated in Table 6. The elevated level of Cd may be attributed to landfill fires and firefighting activities (Oyediran et al. 2020), as these incidents are commonly observed in the Dangkao landfill. Additionally, Cd is commonly used in batteries and rechargeable Ni-Cd batteries, which are often discarded mixed with household waste (Filho and Miguel 2017; Järup 2003). Most of Cd-containing materials are not recycled and instead end up in landfills (Järup 2003). On the other hand, the elevated levels of heavy metals, particularly Pb, in soil samples during the WS can be attributed to the leaching of metal pollutants from the leachate storage pond and landfill into the nearby soils. The absence of a leachate treatment system in the Dangkao landfill results in uncontrolled leachate leakage to the nearby lowlands (Parvin and Tareq 2021), which often leads in surface water contamination and soil pollution in the surrounding water bodies and agricultural fields (Xaypanya et al. 2018). According to Jayakumar et al. (2021), there is a potential for soil contamination by heavy metals such as Cd and Pb, which can be present as impurities in fertilizers used for agricultural purposes. Additionally, Tóth et al. (2016) indicated that the use of fungicides, phosphate fertilization, and inorganic fertilizers can contribute to variations in the concentration of Cd, Cr, Ni, Pb, and Zn in soil. The accumulation of heavy metals in soil can also be influenced by other factors such as soil type, soil porosity, and type of grass cover, as noted by Baziene et al. (2020).

Table 6. Comparison of heavy metal concentrations in soil (mg/kg)

Author	Cd	Cr	Cu	Ni	Pb	Zn
This study	41.08	12.79	56.50	47.16	50.42	29.04
Xaypanya et al. (2018)	10-18	38-71	-	-	50-55	64-83
Vongdala et al. (2019)	3.73-3.76	39.67-48.08	54.06-66.82	19.43-19.94	67.99-80.17	52.48-77.46
Alam et al. (2020)	0.15	-	1.63	-	14.22	3.75
Hredoy et al. (2022)	0.11	47.73	-	28.98	16	-
MoE standards	0.80	100	125	35	85	140
Dutch Standards	0.80	100	36	35	85	140

Figure 17 indicates that soil heavy metal concentrations are more likely to be higher near the closed cell of the landfill (Area A-B) compared to the open cell (Area C and D), specifically for Cd, Cr, Cu, and Ni. This observation is consistent with findings from a previous study conducted by Xaypanya et al. (2018). It is noteworthy that SS2 is situated in a lowland area and experiences frequent inundation throughout the year, resulting in high exposure to leachate leakage and

accumulation. The accumulation of heavy metals in the topsoil generally higher than that in deeper soil layers (Afolagboye, Ojo, and Talabi 2020; Oketola and Akpotu 2015; Shittu et al. 2018). The elevated concentration of Pb in SS5 and Zn in SS4 may be associated with leachate migration, as well as agricultural activities and the leakage of contaminants from places where recyclables are sorted and temporarily stored, which can contain Zn-containing materials. The disposal of batteries, fluorescent lamps, food waste, and burning tires at the site can contribute to the elevated level of Zn (Adeolu et al. 2011).

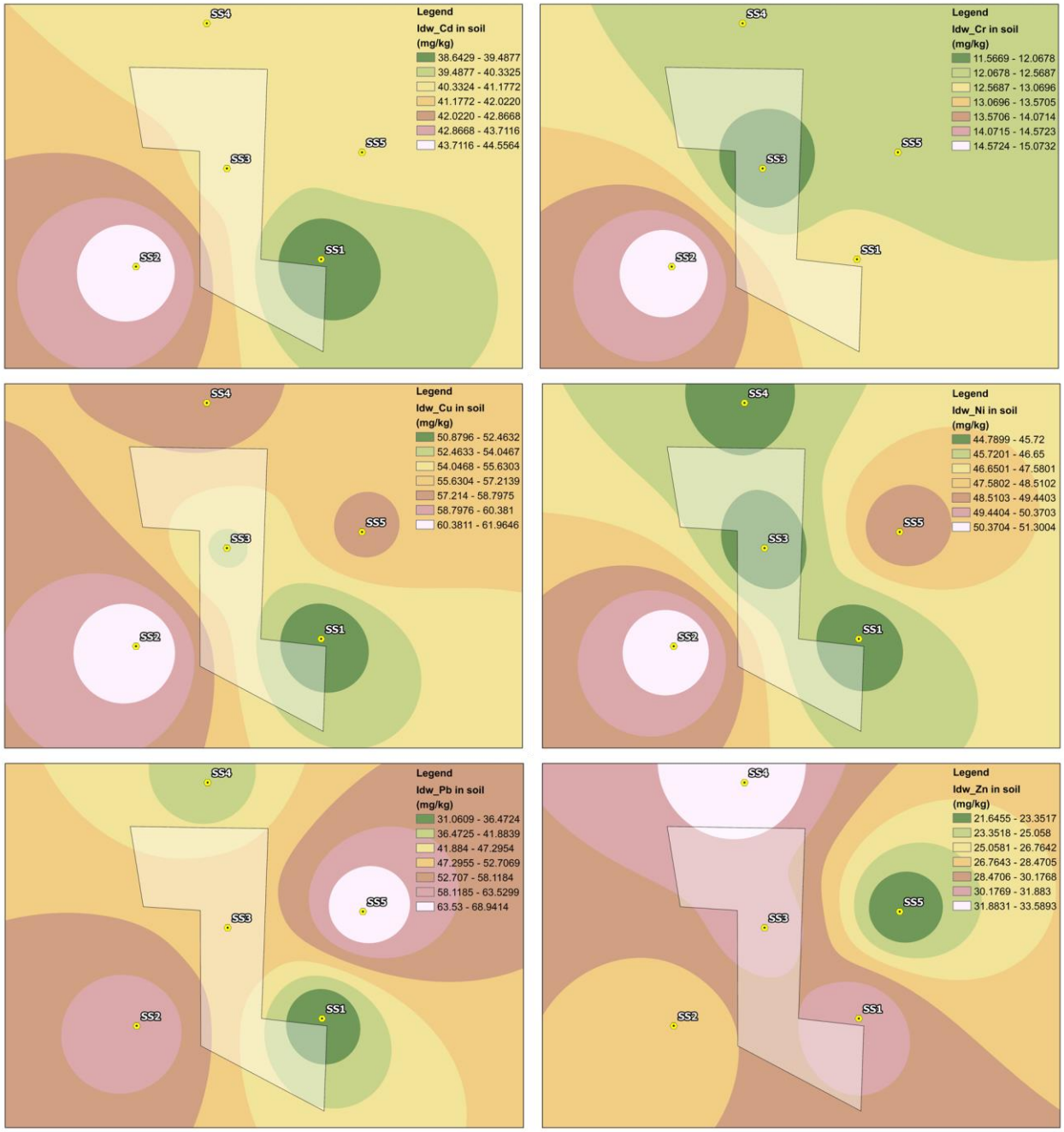


Figure 17. Heavy metal spatial dispersion map in soil

2.3.3. Concentration of Heavy Metals in Plant

Plant samples were collected from the vicinity of the landfill to observe the transfer of heavy metals in the food chain. The average concentrations of heavy metals in plant samples were as follows: Cd: 0.40 mg/kg, Cr: 0.08 mg/kg, Cu: 0.57 mg/kg, Ni: 0.45 mg/kg, Pb: 0.42 mg/kg, and Zn: 0.18 mg/kg, as shown in Table 7. All heavy metal concentrations, except for Cd, were found to be below the permissible limits of the WHO in plant samples. Over time, plants gradually take up Cd from the soil, leading to its accumulation within the plants and subsequent concentration along the food chain, ultimately reaching the human body (Jaishankar et al. 2014).

Table 7. Comparison of heavy metal concentrations in plants (mg/kg)

Author	Cd	Cr	Cu	Ni	Pb	Zn
This study	0.40	0.08	0.57	0.45	0.42	0.18
Vongdala et al. (2019)	0.09-8.24	7.07-164	7.55-219	0.08-71.33	0.25-47.30	0.33-55.03
Alam et al. (2020)	0.06	-	19.59	-	0.54	45.85
Hredoy et al. (2022)	0.40	2.26	-	4.76	8.00	-
Marcussen, Dalsgaard, and Holm (2009)	0.00-0.02	-	0.93-2.95	0.11-0.41	0.06-0.21	3.39-9.08
WHO standards	0.02	1.30	10.00	10.00	2.00	0.60

The high concentration of Cd poses a health risk, particularly for *Ipomoea aquatica* and *Oryza sativa*, as these species are commonly consumed by humans. A study conducted by Hu and Seyfferth (2021) in various paddy fields in Cambodia reported lower Cd concentrations, specifically 0.091 mg/kg in grain and 0.25 mg/kg in stem of *Oryza sativa*. Furthermore, the Cd level in *Ipomoea aquatica* was found to be five times higher than the maximum level observed in edible parts of *Ipomoea aquatica* grown in Choeung Ek lake (Holm et al. 2010). According to Satarug et al. (2010), Cd is known to have a high rate of transfer from soil to plant, leading to elevated levels in fruit and vegetables. The bioaccumulation index of Cd can be as high as 10, and its concentration in plants is typically directly proportional to its concentration in the soil (Kumari and Mishra 2020). Meanwhile, the concentrations of Zn were found to be lowest in the plant samples. A study by Alam et al. (2020) reported a much higher concentration of Zn (60.40 mg/kg) in *Oryza sativa*.

In the plant samples, the concentrations of Cr and Pb varied among roots, stems, and leaves. For *Oryza sativa*, the lowest concentrations of these metals were observed in the stems, while the highest concentrations were found in the roots (Figure *18*). In *Nelumbo nucifera*, the roots from site 2 exhibited the highest concentration of Cr and Pb, while the stems from site 3 had the highest concentration of these metals. This indicates variations in bioaccumulation across different parts of the plants.

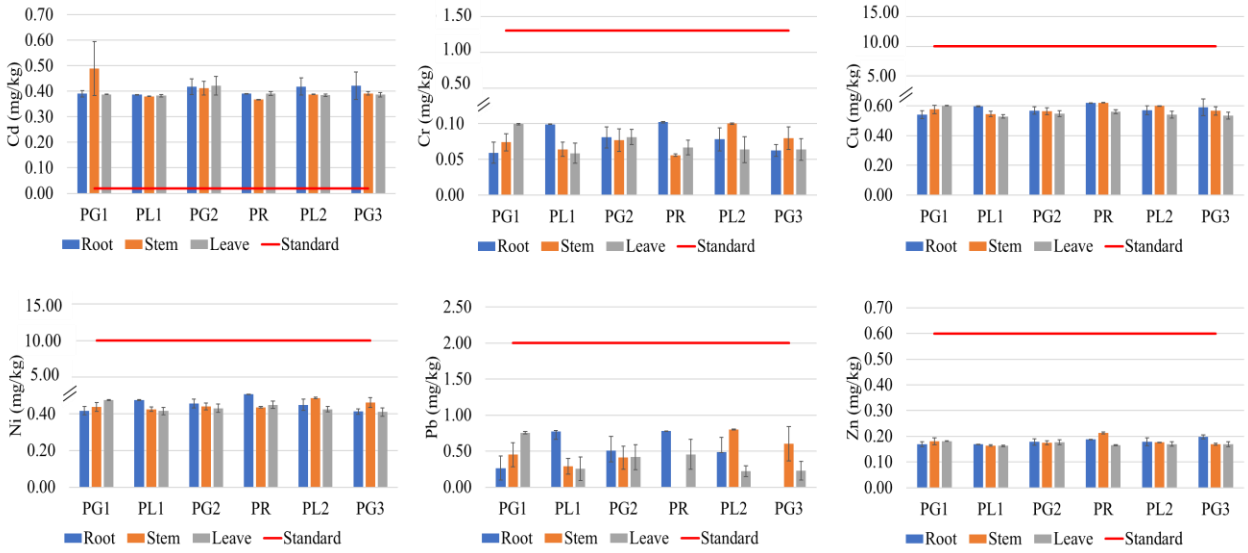


Figure 18. Concentration of heavy metals in plants during the dry season

PG1, PG2, PG3: *Ipomoea aquatica* in sites 1, 2 and 3; PL1, PL2: *Nelumbo nucifera* in site 1 and 2; PR: *Oryza sativa*; bar values are mean ± Standard error; Standard refers to the permissible limit of heavy metal concentration in plant of WHO.

Ipomoea aquatica (PG1 and PG2) and *Oryza sativa* (PR) grown in submerged areas near the landfill exhibited higher concentrations of Cd and Cr. These finding suggests that the observed Cd levels in *Ipomoea aquatica* and *Oryza sativa* in this study are likely influenced by the accumulation of these metals in the soil of these plants' growing areas (Figure 19). In contrast, *Nelumbo nucifera* (PL2) demonstrated high concentrations of Cu, Ni, Pb, and Zn. The increased levels of these metals in PL2 can be attributed to the discharge of household waste, industrial wastewater, and construction waste, which often contain paint, battery, tin, and electronic waste (Holm et al. 2010). It is important to note that PL2 is situated in close proximity to property development projects, an iron factory, and recyclables sorting and storage areas.

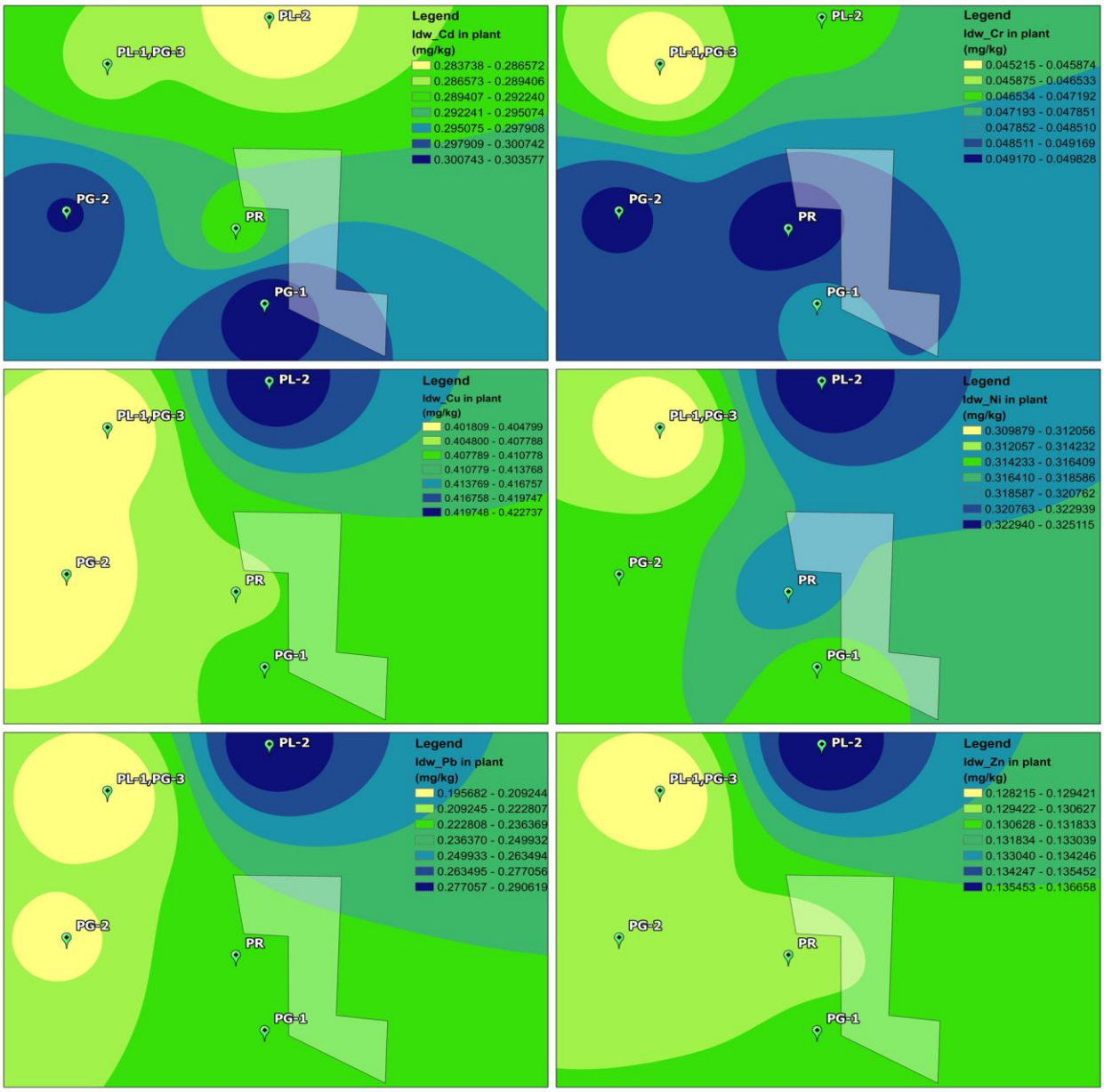


Figure 19. Heavy metal spatial dispersion map in plants

2.3.4. Concentration of Heavy Metals in Fish

Four fish species, namely Striped snakehead (*Channa striata*), Three spot gourami (*Trichopodus trichopterus*), Marble goby (*Oxyeleotris marmorata*), and Croaking gourami (*Trichopsis vittate*), were collected from three different sites near the landfill during both seasons. These fish species are commonly found in freshwater and are frequently consumed, providing significant economic value to the local population. As depicted in Figure 20, the accumulation of heavy metals varied considerably among the different species and locations. This variation can be attributed to their distinct feeding habits and bioaccumulation factor (Anim-Gyampo Maxwell, Kumi Michael 2013).

In normal metabolism, fish may uptake heavy metals from water, food, or sediment (Perera et al. 2015).

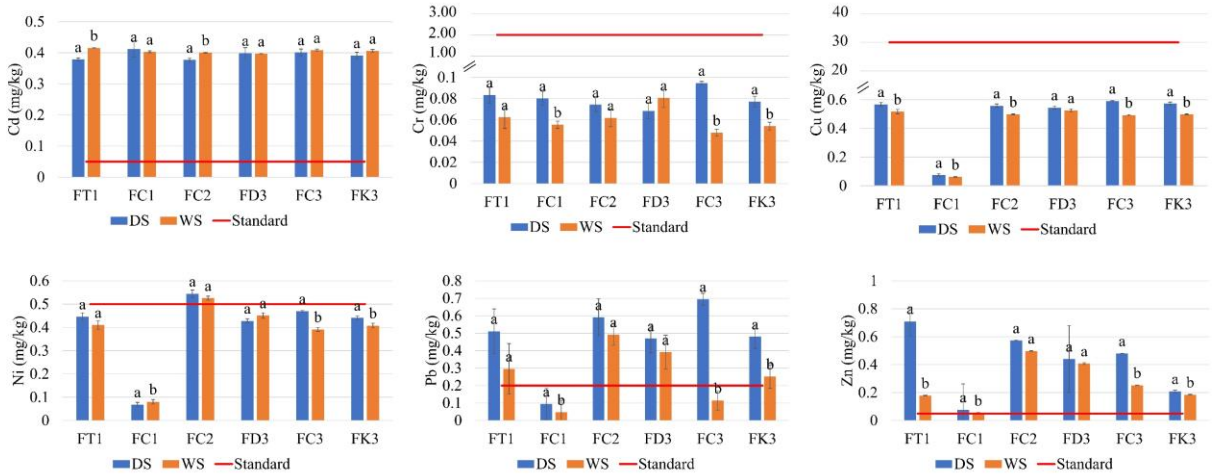


Figure 20. Heavy metals concentration in fish samples

FT1: Channa striata in site 1; FC1: *Trichopodus trichopterus* in site 1; FC2: *Trichopodus trichopterus* in site 2; FD3: *Oxyeleotris marmorata* in site 3; FC3: *Trichopodus trichopterus* in site 3; FK3: *Trichopsis vittata* in site 3; DS: dry season; WS: wet season; bar values are mean ± Standard error; different letters are significantly different within the same sample ($p < 0.05$). Standard refers to the permissible limit of heavy metal concentration in fish of FAO (Hossain et al. 2022) and WHO (Effah et al. 2021).

On average, the concentrations of Cd and Zn in all fish samples exceeded the WHO permissible limit values of 0.05 mg/kg (Effah et al. 2021). The Cd levels were similar across fish species and were approximately eight times higher than the safety limit. Notably, during the WS, the levels of Cd in *Channa striata* (FT1) and *Trichopodus trichopterus* (FC2) were significantly higher than those in the DS. In contrast, Zn levels were significantly higher in the DS, particularly for *Channa striata* (FT1), *Trichopodus trichopterus* (FC1 and FC3), and *Trichopsis vittate* (FK3). The level of Pb was also higher in the DS compared to the WS. Among all fish species, *Trichopodus trichopterus* (FC1 and FC3) during the WS had the lowest levels of Pb, falling within the acceptable limit for human consumption recommended by the WHO at 0.2 mg/kg (Effah et al. 2021). However, the levels of Pb in other fish samples exceeded the permissible limits for human consumption set by the WHO, ranging from 0.37 to 0.54 mg/kg. In a previous study conducted by Holm et al. (2010), higher concentrations of Cd and Pb were found in Blackskin catfish caught from Choeung Ek wastewater drainage system, with values of 0.456 and 0.41 mg/kg fresh weight, respectively. On the other hand, Hossain et al. (2018) reported even higher levels of Pb (7.9 mg/kg)

and Zn (69.7 mg/kg) in Tilapia fish collected from leachate-treated pond. The average levels of Ni in *Trichopodus trichopterus* (FC2) exceeded the permissible limit of 0.5 mg/kg set by the FAO (Hossain et al. 2022), and the one-way ANOVA analysis showed no significant different between the seasons. However, Ni levels in other fish species were below the permissible limit. The concentrations of Ni in fish bodies reported in the study by Hossain et al. 2018 were twice as high as those observed in this study. Additionally, the concentrations of Cr and Cu in all fish species were well below the permissible limit values of 2 and 30 mg/kg, respectively, as set by the WHO (Effah et al. 2021). In contrast, another study conducted in a leachate-treated pond recorded high levels of Cu (10.6 mg/kg) in Tilapia fish (Hossain et al. 2018).

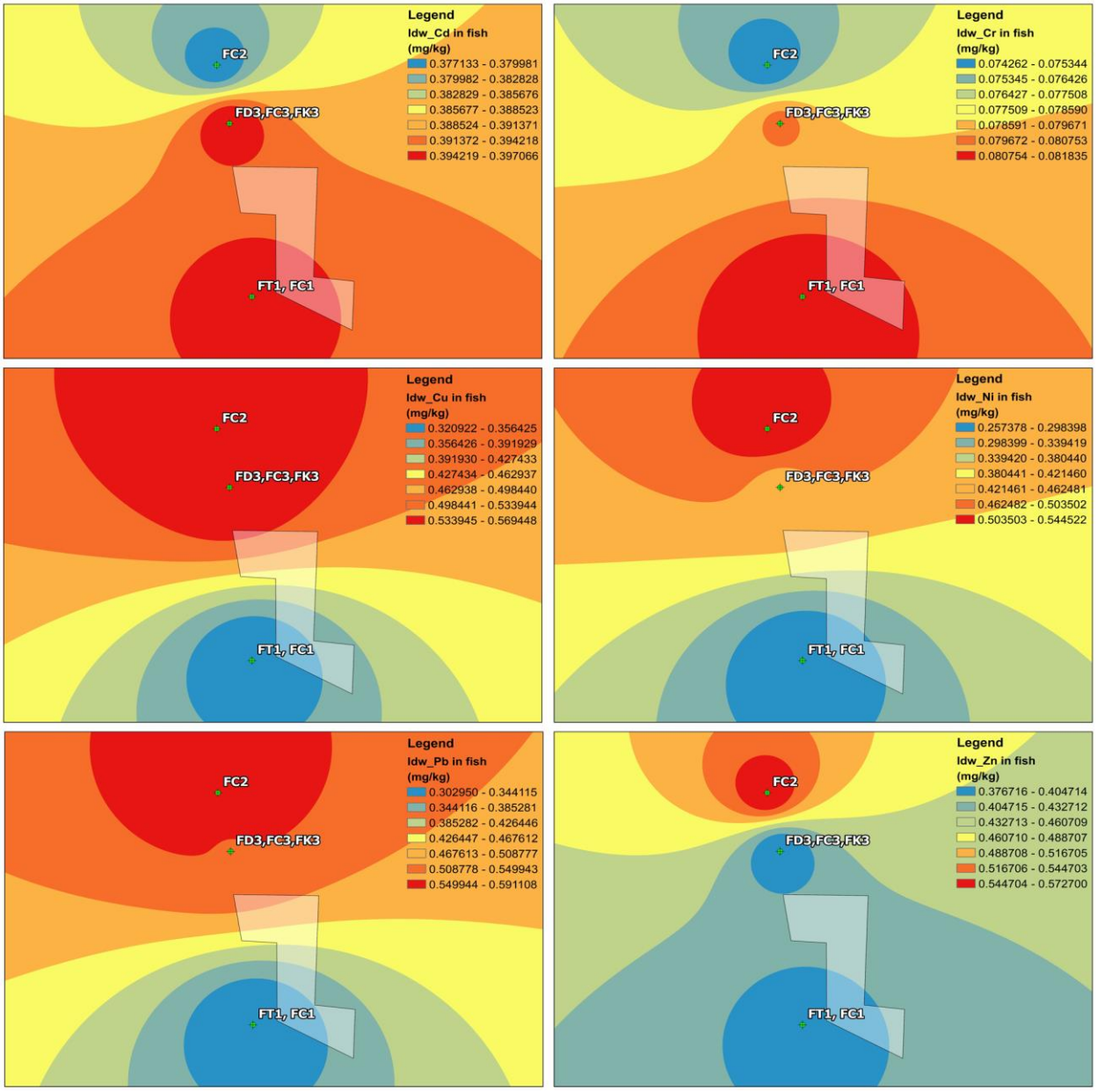


Figure 21. Heavy metal spatial dispersion map in fish

As shown in Figure 21, the concentrations of heavy metals in fish samples were consistent with those found in plant samples. The elevated levels of heavy metals observed in this study may be associated with their accumulation in surface water and soil. Aquatic organisms possess the ability to uptake heavy metals, particularly Cd compounds in both water-soluble and sediment forms, which are then transferred indirectly to fish bodies through the food chain (Perera et al. 2015). Cd exhibits the highest bioaccumulation in various parts of the fish body, including the liver, kidney, gills, and skin (Garai et al. 2021). The high concentrations of heavy metals, particularly Cd, Ni, Pb, and Zn, in fish bodies pose potential health risks for humans when consumed.

2.4. Limitations of the study

This study encompassed a diverse range of samples, such as surface water, groundwater, soil, plants, and fish. Samples were collected in both the WS and DS to assess the significant variation in heavy metal concentrations among the two seasons. However, it is important to note that the number of samplings conducted in this study is relatively low for conducting robust statistical tests. A larger sample size would have allowed for a more reliable statistical analysis and increased the generalizability of the results. Additionally, increasing the number of sample sizes would have provided a more comprehensive understanding of the heavy metal concentrations in the study area and enhanced the reliability of the findings. Therefore, future studies with a larger number of samples are recommended to further validate and strengthen the conclusions drawn from this study.

2.5. Conclusions

This study examined the concentrations and distribution of heavy metals in water, soil, plants, and fish in the Dangkao MSW landfill and its vicinity. The results revealed a higher accumulation of heavy metals in surface water was found in the canal, particularly in middle and downstream areas, indicating that these contaminants had migrated from the landfill site and accumulated downstream. Groundwater located in the landfill site exhibited a high concentration of heavy metals, implying a strong correlation with proximity to the landfill site. Furthermore, the Dangkao landfill has a maximum pit depth of 30 m, thus there is a high possibility of leachate leakage into the groundwater aquifer due to improper liner installation and the absence of leachate collection and treatment system. The presence of certain heavy metals in groundwater exceeded the allowable limits, making it unsuitable for drinking purposes. Furthermore, the concentration of heavy metals was observed to systematically accumulate in soils, plants, and fish, indicating a progressive buildup of contamination. This clearly indicates that the soil in the vicinity of the landfill was contaminated with high levels of certain heavy metals, which were subsequently transferred to plants and fish.

Among the analyzed heavy metals, Cd exhibited the highest levels and exceeded the permissible limits in all samples. This finding suggests a high possibility of Cd mobility from the landfill site to water and soil and bioaccumulate in aquatic organisms. The high concentration of Cd in groundwater, plants, and fish signifies a potential health risk for individuals who regularly consume these contaminated sources. Furthermore, Cd concentrations in the soil samples have surpassed the intervention value, indicating a severe risk to the functionality of soil. Remediation measures are necessary to address this issue and restore soil health. Bioremediation and phytoremediation should be considered as they are cost-effective and natural approaches for removing contaminants in soil and water. The concentration of Cr, and Ni were also found to exceed the standard limits in surface water, groundwater, and fish samples. Additionally, the concentrations of Pb and Zn in most of fish samples were found to be higher than the allowable limit. The accumulated level of these metals was observed to be higher during the DS, which can be attributed to factors such as lower water volume, reduced dilution, and higher water temperatures. These conditions can lead to increased concentrations of pollutants, including heavy metals, in water. Therefore, fish inhabiting these water bodies are more susceptible to higher levels of heavy metal exposure through their feeding and respiration processes.

These findings strongly indicate that there is a high likelihood of heavy metal contamination in the surrounding environment of the Dangkao landfill due to potential leakages. The migration of these contaminants, coupled with the low-lying topography, played a significant role in the accumulation of heavy metals in water and soil, posing a potential risk to both aquatic ecosystems and human health. Overall, the environment condition surrounding the landfill is unfavorable. Hence, regular monitoring of heavy metals in water and soil is crucial to safeguard both the environment and human well-being. Implementing sustainable waste management practices and measures to minimize the release of heavy metals into the environment can help mitigate these issues. Future studies should also consider conducting human health risk assessments as well as cancer risk assessments.

CHAPTER 3

ACCOUNTING FOR GREENHOUSE GAS EMISSIONS FROM MUNICIPAL SOLID WASTE LANDFILLING

3.1. Introduction

Landfilling remains a commonly used method for disposing of municipal solid waste (MSW). Biodegradable wastes in the landfill site decompose over time, generating LFG which is primarily composed of CH_4 and CO_2, along with a small proportion of non-methane organic compounds (Amini et al. 2013; Machado et al. 2009). Fugitive CH_4 emissions from disposal sites have a high global warming potential (GWP), accounting for about 5% of the world's anthropogenic GHGs (Bogner et al. 2007). CO_2 emitted from landfills is of biogenic origin and is therefore not considered a GHG (IPCC 2006). LFG generation varies depending on the physical composition, permeability, moisture, temperatures, and landfill management practices (Lizik et al. 2013; Spokas et al. 2021). Some countries have regulated the rules for landfill management to mitigate the negative impacts on the environment, including, but not limited to, the installation of LFG collection, separation of organic waste, and the soil cover for bio-oxidation of CH_4 as it passes through the top cover (Mohsen, Abbassi, and Zytner 2020). Denmark, for example, has implemented various methods for landfill gas management, such as utilizing CH_4 for electric energy and heat, flaring, microbial oxidation of CH_4, and constructing biofilters, as regulated by their government (Scheutz and Kjeldsen 2019). In the USA, the Clean Air Act requires certain landfills to install and operate a gas collection and control system. As a result, many landfill owners voluntarily collect LFG for flaring and take advantage of it as a renewable energy resource. By 2021, 550 LFG-to-energy projects were operating in the USA, producing electricity (70%), direct use (17%), and renewable natural gas (RNG) (13%) (LMOP 2021). Meanwhile, in the United Kingdom (UK), the emission of LFG accounts for about 20% of the country's total CH_4 emissions, and the UK government has implemented regulations to manage GHG emissions by increasing the use of enclosed flares, improving LFG collection efficiency, and expanding the number of LFG utilization projects (Environment Agency 2002). However, despite the presence of highly effective LFG collection systems, some amount of LFG is still released (Barlaz et al. 2009), either through the bio-oxidation of CH_4 when the landfill surface is covered by soil, leakage of the collection system, or the leachate collection system (Amini et al. 2013).

Collecting the LFG generated from landfill sites for flaring or energy purposes can help reduce fugitive CH_4 emissions, thereby reducing direct GHG emissions resulting from landfill management. Moreover, electricity generated from LFG can replace energy from conventional sources like coal and fossil fuel, indirectly reducing CO_2 emissions. It is worth noting that not all carbon-containing materials in landfills undergo degradation. Certain degradable carbon compounds persist and can be stored in landfills for an extended period (Barlaz et al. 1997; Kumar and Sharma 2014b). The biogenic carbon storage in landfills brings environmental benefits and should be included in GHG emissions assessments (Friedrich and Trois 2013a). These avoided emissions can offset the indirect GHG emissions resulting from landfill operation (e.g. compaction, excavation, and soil cover, which consume diesel), leachate treatment (e.g. water, chemicals, and electricity), and construction (e.g. gravel and synthetic liner) (Friedrich and Trois 2013a). Studies in European countries have demonstrated that utilizing LFG and carbon sequestration can reduce overall GHG emissions, resulting in net GHG emissions being reduced to zero or even negative values in conventional landfills (Manfredi et al. 2009).

In Cambodia, rapid urbanization, coupled with socio-economic development and population growth, has led to a significant increase in MSW generation. The lack of treatment infrastructure, technology, and management has added to the environmental burden of MSW, particularly in major cities like Phnom Penh. Numerous landfills have been created to accommodate the excessive increase in MSW volumes. Phnom Penh is the most developed city, facing challenges with the rapid increase in waste disposal. MSW generation in Phnom Penh municipality accounts for about a quarter of the total generation nationwide (Dek et al. 2022). In 2022, approximately 3,538 tMSW was collected daily and sent to a landfill without intermediate treatment. Organic matter with high moisture content is the most predominant waste fraction disposed of at landfills, accounting for more than 50% (Seng et al. 2018). The degradation of carbon content in degradable wastes poses a high potential risk, including an increase in global warming through GHG emissions, soil contamination, surface water and groundwater pollution by leachate, human health risks through disease spreading, and fire and explosion hazards by LFG (Kumar and Sharma 2014b; Seng et al. 2013). It is important to note that the current landfill is operating without leachate treatment and LFG capture systems, which are mostly found in the majority of cities in Cambodia. Consequently, studies on GHG emissions from a landfill site are very limited in Cambodia. Seng et al. (2013) investigated CH_4 emissions from a landfill using the IPCC model. However, the researchers considered only the emissions of CH_4, which is state of the art in the IPCC model, while the emission of N_2O and other NMOC compounds from the landfill and emissions from landfill

operations were not included. Carbon storage has the potential to significantly offset GHG emissions from landfills, but it is often overlooked in the studies of GHG emissions from landfill sites.

The MSW landfills in Cambodia are currently experiencing an improvement phase due to economic growth and increased environmental awareness among the population. The extraction and treatment of LFG may become increasingly important in the country's landfill management. Therefore, data on LFG emissions is essential for developing policies to mitigate the environmental impacts of landfills and designing LFG-to-energy projects. This study aimed to quantify GHG emissions resulting from MSW landfilling in Phnom Penh municipality over a 14-year period (2009-2022), considering four different landfill management options.

3.2. Landfill gas generation investigation approaches

Several methods have been developed to evaluate LFG generation from disposal sites, including field measurement methods and mathematical models. Field measurement methods, such as flux chambers, tracer gases technique, Horizontal Radial Plume Mapping Optical Remote Sensing (HRPM ORS), inverse modeling technique, differential absorption light detection and ranging (LiDAR), micrometeorological eddy covariance (EC), helicopter-borne spectroscopy, have been used to investigate CH_4 collection and fugitive CH_4 emissions from landfills (Babilotte, Lagier, and Fiani 2010). However, these methods are time-consuming and costly and introduce uncertainties when measuring in a large-scale landfill due to spatial and temporal fluctuations in CH_4 flow and its components (Amini et al. 2013). Sample site selection and uncontrolled leakage may also lead to uncertainty in on-site measurements (Ghosh et al. 2019).

In the meantime, mathematical models have been developed to estimate LFG emissions based on waste disposal data, waste composition, moisture content, landfill cover material, and LFG collection (Amini, Reinhart, and Mackie 2012). A significant number of models have been developed over the years and have drawn the attention of many researchers in the industry, including but not limited to the IPCC default model, Modified Triangular method (MTM), Dutch Multiphase First-order model, AMPM, GASSFILL, Scholl Canyon Fir-order model, Rettenberger First-order model, E-PLUS model, Zero-order German EPER model, IPCC First-order model, US EPA Landfill Gas Emissions Model (LandGEM), Afvalzorg model, and Gassim (Majdinasab, Zhang, and Yuan 2017). Among those, LandGEM is widely used for assessing LFG and other air pollutants from the decomposition of landfilled waste. The model was first developed in 2005 by

the US EPA based on a first-order decay (FOD) rate (Alexander, Burklin, and Singleton 2005). Users can either input site-specific data or use the default values if site-specific data is not available. The default data is based on empirical data from various landfills in the USA. Another model commonly used by many research scholars is the IPCC model, which consists of two methods for quantifying GHG emissions from solid waste disposal sites (IPCC 2006). The Tier 1 method of the IPCC model estimates CH_4 emissions from the mass balance of waste, while Tier 2 uses the FOD method, which produces highly accurate estimated results. Hence, the FOD model is recommended for CH_4 emissions estimation. The model estimates CH_4 emissions based on waste compositions and assumes a slow degradation of organic matter over decades. Challenges regarding the accuracy of models have been raised, associated with the input parameters, including the amount of waste disposed of, the physical composition of landfilled waste, moisture content, temperature, and lag time in gas generation (Amini et al. 2013). However, the models have several advantages over field measurement methods when investigating CH_4 emissions in a large-scale landfill (Amini et al. 2013).

3.3. Methodologies

3.3.1. Data collection

The data of landfilled waste was obtained from the Dangkao landfill office for a period of 14 years, from 2009 to 2022. This historical waste disposal data was used to calculate LFG emissions in the estimation models, namely LandGEM and IPCC FOD models. Daily records of MSW disposal data were available and showed a dramatic increase from 177,224 t/year in 2009 to 1,288,223 t/year in 2022 (Table 8). In 2022, the MSW disposal rate was estimated at 1.34 kg/capita/day, which had increased from 0.70 kg/capita/day in 2009.

Table 8. Population and MSW disposal from 2009 to 2022

Year	Population[a] (thousand)	Waste landfilled (t/year)	Per capita disposal (kg/capita/day)	Year	Population (thousand)	Waste landfilled (t/year)	Per capita disposal (kg/capita/day)
2009	1,393	177,224[b]	0.70	2016	1,947	717,435	1.01
2010	1,461	409,336	0.77	2017	2,043	808,530	1.08
2011	1,533	442,469	0.79	2018	2,143	965,944	1.24
2012	1,608	492,380	0.84	2019	2,282	1,015,980	1.22
2013	1,687	532,471	0.86	2020	2,394	1,035,878	1.19
2014	1,770	617,489	0.96	2021	2,511	1,012,039	1.10
2015	1,856	681,905	1.01	2022	2,634	1,288,223	1.34

[a] Population was estimated based on Generation Population Census 2019 (National Institute of Statistics 2020)

[b] The value is six months period; hence, per capita disposal in 2009 assumed the same proportion in the first six month

The physical characteristics of landfilled waste were taken from Seng, Fujiwara, and Seng (2018). As shown in Figure 22, the organic fraction accounted for the highest proportion of the disposed waste, at 55.87%. The second-highest component was recyclables, comprising plastics (21.13%), mixed paper (6.54%), glass (1.42%), and metals (1.05%).

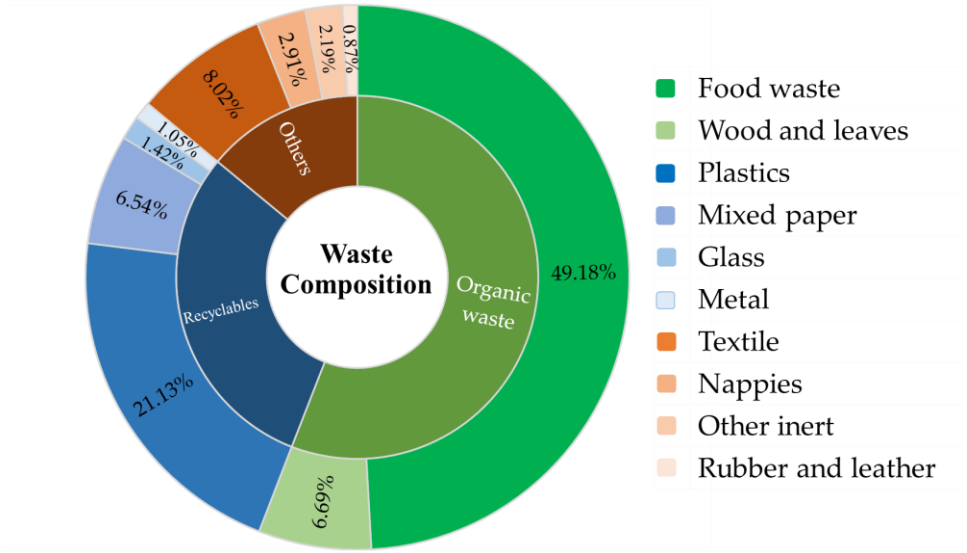


Figure 22. Landfilled municipal solid waste composition in Phnom Penh (Seng et al. 2018)

3.3.2. Landfill management scenarios

The present management of the Dangkao landfill involves simple dumping without treating leachate and collecting LFG for energy or flaring. This method is widely used across the country because it is cost-effective and easy to operate. However, this approach may adversely affect the environment due to LFG emissions and the potential for leachate infiltration, which could contaminate soil, surface water, and groundwater resources. Four scenarios for managing the landfill were developed, including the current management practice in scenario 1, which was compared with three improved scenarios (Figure 23, Table 9). In scenario 1, the landfill operates without extracting LFG and treating leachate. The leachate is stored in a lagoon for recirculation without treatment. In this scenario, emission sources include direct fugitive CH_4 emissions that are not captured by the gas capture system, as well as emissions from diesel used for heavy-duty equipment, such as excavation and compaction. Biogenic carbon sequestration potential was also

taken into account as the landfill's potential for offsetting GWP. In scenario 2, landfill management remains the same as in scenario 1, but the leachate generation is collected and treated instead of recirculating. Indirect GHG emissions from electricity consumption and chemical production used for leachate treatment are additionally included in this scenario. Scenario 3 is more advanced than scenario 2 since LFG is captured for flaring. Hence, GHG emissions under scenario 3 are expected to recover 50% of CH_4 for flaring. The optimal scenario is presented in scenario 4, where LFG is captured and utilized for energy production. GHG emissions quantification under scenario 4 is similar to that in the first three scenarios. However, due to the application of LFG-to-energy technology, electricity generation can offset the generation from conventional energy sources. Therefore, indirect GHG emissions savings from electricity substitution are accounted for in scenario 4. In the scenario comparison, the study considered the three main GHGs, such as non-biogenic CO_2, CH_4, and N_2O, using a 100-year GWP of 1, 25, and 298, respectively (IPCC 2006).

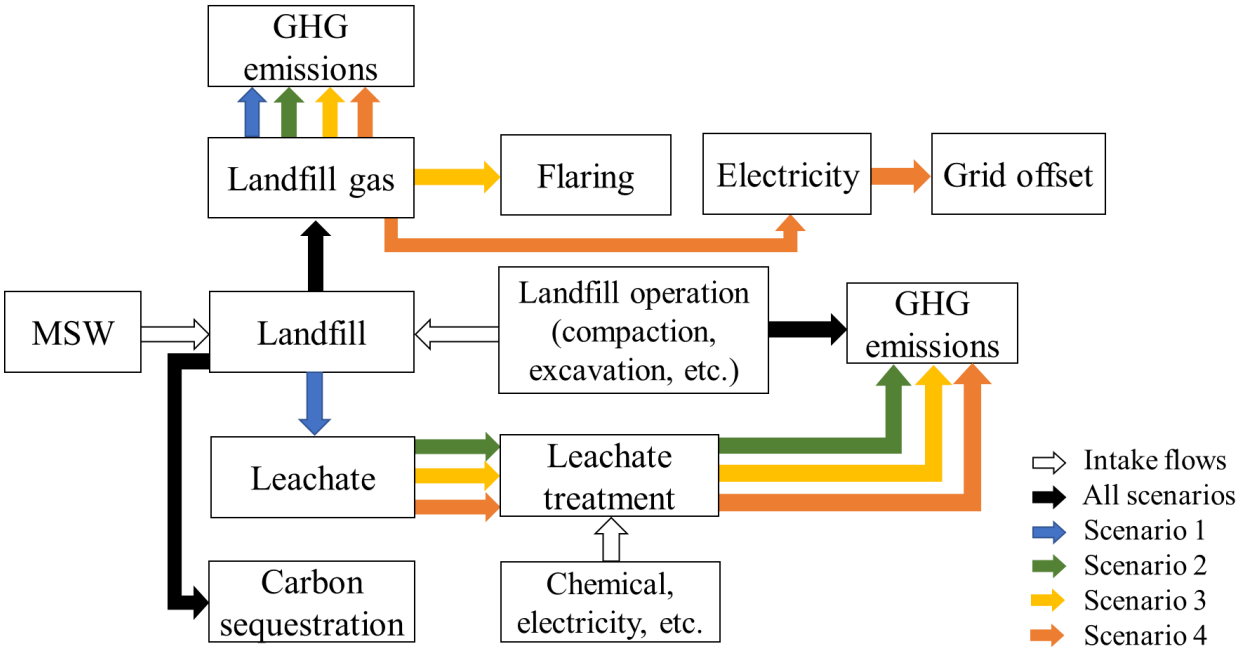


Figure 23. System boundary for GHG emissions quantification

Table 9. Setting of different emission reduction scenarios

Scenario name	**Technical management**	**Scenario description**	**Parameter setting**
Scenario 1	Simple dumping	The method of waste disposal being described is simple dumping, where only a partial layer of soil	Bio-oxidation is not considered in this scenario, as a partial soil cover is used. Only the diesel consumption for waste excavation and compaction is taken into

		cover is applied, and there is no treatment of leachate or the collection of LFG.	account, at a rate of 0.43 L/tMSW. Carbon sequestration may help to reduce the emissions produced.
Scenario 2	Leachate treatment	The waste management facility employs a daily soil cover, as well as a leachate collection and treatment system, to control odors and prevent the spread of contaminants.	Bio-oxidation in the topsoil cover is estimated to be 10%, resulting in a reduction of CH_4 emissions through the oxidation process. Leachate generation is calculated to be 2.2527 m^3/tMSW, with a collection efficiency of 40%. Carbon sequestration may help to further reduce the amount of emissions produced.
Scenario 3	Leachate treatment and flaring	The landfill site follows the best practices of waste management, which includes a daily soil cover with a leachate treatment system, as well as a flaring system to manage landfill gas emissions.	A substantial amount of CH_4 emissions is reduced through the implementation of bio-oxidation in the soil cover (10%) and a flaring system (50%). Leachate generation is estimated to be 2.2527 m^3/tMSW, with a collection efficiency of 40%. Carbon sequestration may help to further reduce the amount of emissions produced.
Scenario 4	Leachate treatment and energy recovery	The landfill site has implemented an advanced waste management system, which includes a daily soil cover with a leachate treatment system, as well as an LFG-to-energy system that converts landfill gas into usable energy.	The implementation of bio-oxidation in the soil cover (10%) and an energy recovery system (75%) have significantly reduced CH_4 emissions at the landfill site. Carbon sequestration and electricity substitution measures can further contribute to emission reduction. The emission savings from electricity substitution are equivalent to the national grid emission factor, estimated to be 0.586 $kgCO_2$-eq/kWh. Leachate generation is estimated to be 2.2527 m^3/tMSW, with a collection efficiency of 40%.

3.3.3. Calculation of LFG emissions

In the quantification of GHG emissions, landfill CH_4, the most critical GHGs, was calculated using the LandGEM and IPCC FOD models from 2009 to 2022. Different compositions of waste can generate varying amounts of CH_4 due to their carbon content. For example, waste containing cellulose degrades quickly under landfill conditions, while waste with lignin decomposes slowly or not at all (Friedrich and Trois 2013a). This study only considered wastes that can be degraded, such as organic waste, paper, textiles, nappies, and leather and rubber. Other types of waste were not included in the models used in the study.

LandGEM model

The landfill CH_4 generation was calculated using the US EPA LandGEM 3.02 model (US EPA 2005). The model calculates the CH_4 gas based on the FOD approach as given in Equation (1), which considers the degradation of biodegradable matters over time. The model has two important parameters: the CH_4 generation rate per year, k, and the CH_4 generation potential, L_0 (m^3/Mg).

$$Q_{CH_4} = \sum_{i=1}^{n} \sum_{j=0,1}^{1} k \times L_0 \times \left(\frac{M_i}{10}\right) \times e^{-kt_{i,j}} \quad (1)$$

Where, Q_{CH_4} is the annual methane generation in the year of calculation (m^3/year); i is the one-year time increment; j is the 0.1-year time increment; n is the duration of waste acceptance at the landfill (year); M_i is mass of waste disposed of in year i (t); and $t_{i,j}$ is the time in year j^{th} section of waste M_i accepted (year).

The calculation of CH_4 generation potential (L_0) follows Equation (2).

$$L_0 = MCF \times DOC \times DOC_f \times F \times \frac{16}{12} \quad (2)$$

Where, MCF is the CH_4 correction factor, taken as 0.8 for unmanaged landfill deeper than 5 m (Table 10); DOC is the degradable organic yielded on the CH_4 in landfill gas; DOC_f is the fraction of degradable organic carbon which decomposes, taken as 0.77 (IPCC 2006); F is the fraction of CH_4 in landfill gas, taken as 0.5; and 16/12 is the conversion factor from methane to carbon.

The degradable organic carbon (DOC) is calculated using Equation (3).

$$DOC = (0.4 \times A) + (0.17 \times B) + (0.15 \times C) + (0.30 \times D) \quad (3)$$

Where, A is paper and textile; B is nappies; C is food waste; and D is wood and leaves. The value of these waste compositions is presented in Figure 22.

Table 10. The value of MCF recommended in the IPCC 2006 guidelines

Type of solid waste disposal site	MCF default values
Managed – anaerobic	1.0
Managed – semi-aerobic	0.5
Unmanaged – deep (>5 m waste) and/or high water table	0.8
Unmanaged – shallow (<5 m waste)	0.4
Uncategorized landfill	0.6

The value of the CH_4 generation rate constant (k) reflects the degradation rate of disposed waste composition. The k value in this study is derived from Equation (4).

$$k = \sum_{i=1}^{n}(k_i \times W_i) \quad (4)$$

Where, k_i is the degradation rate of decomposable waste composition i and W_i is the fraction of decomposable i. This study used the default values of k_i recommended in the IPCC 2006 guidelines for the moist and wet tropical climate region (Table 11).

Table 11. The values of k_i and DOC used in model calculation

Type of waste	**Temperate climate**				**Tropical climate**			
	Dry		**Wet**		**Dry**		**Moist and Wet**	
	Default	**Range**	**Default**	**Range**	**Default**	**Range**	**Default**	**Range**
Paper/textile	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	0.02	0.01–0.03	0.03	0.02–0.04	0.025	0.02–0.04	0.035	0.03–0.05
Garden and other non-food organic	0.05	0.04–0.06	0.1	0.06–0.1	0.065	0.05–0.08	0.17	0.15–0.2
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

IPCC FOD model

The IPCC FOD model was also employed to estimate the landfill CH_4 emissions and compared the result with the LandGEM model. Calculation of CH_4 generation using the IPCC FOD model is given in Equations (5-13) (IPCC 2006).

$$DDOC_m = DDOC_m(0) \times e^{-kt} \quad (5)$$

Where, $DDOC_m$ is the mass of DDOC at any time and $DDOC_m(0)$ is the mass of decomposable degradable organic carbon (DDOC) at the start of the reaction (t=0 and e^{-kt}=1). From Equation (5), at the end of the year, the mass of DDOC left not decomposed in the landfill site was calculated as follows:

$$DDOC_m(1) = DDOC_m(0) \times e^{-k} \quad (6)$$

And the mass of DDOC decomposed into CH_4 and CO_2 was estimated based on below Equation (7)

$$DDOC_{mdecomp}(1) = DDOC_m(0) \times (1 - e^{-k}) \quad (7)$$

However, in actual situations, decaying reactions might have started in previous years of waste deposition in the landfill site, so a separate calculation for the deposition years was needed. To calculate the mass of decomposable DOC ($DDOC_m$) from the amount of waste material, the following equation was used:

$$DDOC_{md}(T) = W(T) \times DOC \times DOC_f \times MCF \quad (8)$$

The amount of deposited $DDOC_{mrem}$ remaining (not decomposed) at the end of the deposition year T was calculated as follows:

$$DDOC_{mrem}(T) = DDOC_{md}(T) \times e^{(-k\times((13-M)/12)} \quad (9)$$

and the amount of deposited $DDOC_m$ decomposed during deposition year T was calculated as:

$$DDOC_{mdec}(T) = DDOC_{md}(T) \times \left(1 - e^{(-k\times((13-M)/12)}\right) \quad (10)$$

The amount of $DDOC_m$ accumulated in the landfill at the end of year T was calculated as:

$$DDOC_{ma}(T) = DDOC_{mrem}(T) \times (DDOC_{ma}(T-1) \times e^{-k}) \quad (11)$$

The total amount of $DDOC_m$ decomposed in year T was calculated as:

$$DDOC_{mdecomp}(T) = DDOC_{mdec}(T) + \left(DDOC_{ma}(T-1) \times (1 - e^{-k})\right) \quad (12)$$

The amount of CH_4 generated from DOC decomposed was:

$$Q_{CH4} = DDOC_{mdecomp}(T) \times F \times \frac{16}{12} \quad (13)$$

where, T is the year of inventory; $W_{(T)}$ is the amount of waste deposited in year T; MCF is the CH_4 correction factor; DOC is the degradable organic carbon (under aerobic conditions); DOC_f is the fraction of DOC decomposing under anaerobic conditions (0.0–1.0); $DDOC$ is the decomposable degradable organic carbon (under anaerobic conditions); $DDOC_{md(T)}$ is the mass of $DDOC$ deposited year T; $DDOC_{mrem(T)}$ is the mass of DDOC deposited in inventory year T, remaining not decomposed at the end of year; $DDOC_{mdec(T)}$ is the mass of DDOC deposited in inventory year T decomposed during the year; $DDOC_{ma(T)}$ is the total mass of DDOC left at the end of year T; $DDOC_{ma(T-1)}$ is the total mass of DDOC left decomposed at the end of year $T-1$; $DDOC_{mdecomp(T)}$ is the total mass of DDOC decomposed in year T; Q_{CH4} is the CH4 generated in year T; F is the fraction of CH_4 by volume in generated landfill gas (0.0–1.0); 16/12 is the molecular weight ratio of CH_4/C.

3.3.4. Calculation of fugitive CH_4 emissions

The landfill in Phnom Penh currently operates without the function of a CH_4 capture system, resulting in the release of all generated CH_4 into the atmosphere. Nevertheless, not all of the produced CH_4 is emitted into the atmosphere, as a fraction of it undergoes oxidation in the topsoil cover. For this study, a CH_4 oxidation rate of 10% was adopted following the IPCC 2006 guidelines. As the current landfill operation only partially applies soil cover, scenarios 1 and 2 assumed no CH_4 oxidation. Fugitive CH_4 emissions in these scenarios were calculated by multiplying the CH_4 yield obtained from the LandGEM and IPCC FOD models by the GWP for CH_4, which is 25. In scenarios 3-4, fugitive CH_4 emissions were determined based on CH_4 collection efficiency, oxidation rate, and CH_4 burning-out efficiency, and were calculated following Equations (14-15):

$$E_{FM} = F_{CH_4} \times GWP_{CH_4} \quad (14)$$

$$F_{CH_4} = [Q_{CH_4} \times (1-\lambda) \times (1-OX)] + [Q_{CH_4} \times \lambda \times (1-\zeta)] \quad (15)$$

Where, E_{FM} is the GHG emissions from the fugitive emissions; F_{CH_4} is the amount of CH_4 released from the landfill site; Q_{CH_4} is obtained from the LandGEM and IPCC models, and ζ is the burn-out rate of CH_4 either by flaring and LFG to energy, taken as 91.1% (Plant et al. 2022).

3.3.5. Calculation of avoided emissions from carbon sequestration

Some biodegradable waste containing biogenic carbon may not be fully degraded even after 100 years of being disposed of in a landfill and could remain stored within the landfill body. This sequestration biogenic carbon is considered as emission savings in the quantification process

(Yang et al. 2013). This study accounted for the avoided emissions from carbon sequestration, following previous works (Friedrich and Trois 2013a; Manfredi et al. 2009; Yang et al. 2013), and these can be calculated as shown below (Entreprises pour L'Environnement 2013):

$$E_{CS} = CSF \times \frac{44}{12} \quad (16)$$

$$CSF = W_{(T)} \times DOC \times \left(1 - DOC_f\right) \times MCF \quad (17)$$

Where, E_{CS} is the GHG emissions saving because of carbon sequestered in the landfill; *CSF* is the carbon sequestered factor (kg C/tMSW).

3.3.6. Calculation of N_2O emissions

N_2O and other NMOCs emitted from landfills also contribute to the GWP. However, NMOCs are typically found in very low concentrations (less than 0.1% v/v) and are not considered in GHG accounting (Yang et al. 2013). N_2O emission from landfills is not available in LandGEM or IPCC FOD models. However, N_2O significantly impacts GWP as its potency is 298 times higher than CO_2 (IPCC 2006), and its atmospheric lifetime extends up to 120 years, necessitating its consideration (IPCC 1995). Despite being negligible, it should be taken into account (Seng et al. 2013). N_2O generation has been found to have a substantial relationship with CH_4 in the waste layer, and landfills in tropical climate zones exhibit higher N_2O emissions (Ishigaki et al. 2016). According to Yang et al. (2013), the emission factor for N_2O ranges from 0.5 to 2 g/kg of CH_4 emitted. Therefore, this study assumes an N_2O generation rate of 2 g/kg of CH_4 due to our climatic zone, and it can be calculated as follows:

$$E_{N_2O} = F_{CH_4} \times EF_{N_2O} \times GWP_{N_2O} \times 1000 \quad (18)$$

Where, E_{N_2O} is the GHG emissions due to the emissions of N_2O ($kgCO_2$-eq/tMSW); EF_{N_2O} is the emission factor for N_2O, taken as 2 g/kg of fugitive CH_4 (Yang et al. 2013); and GWP_{N_2O} is the global warming potential of N_2O, taken as 298 (IPCC 2006).

3.3.7. Calculation of emissions from landfill operation

GHG emissions from landfill operations result from the consumption of electricity, diesel, and auxiliary materials such as HDPE and gravel, which are used for liner, leachate collection, and LFG capture systems. Currently, the landfill operation only uses diesel to power heavy-duty equipment such as excavators, bulldozers, etc. The daily amount of diesel fuel used for this purpose is 1,435 L, equivalent to 0.43 L/tMSW (Dangkor landfill Authority 2021). However, additional

diesel is required for daily on-site operations when upgrading the landfill with leachate treatment and LFG collection systems. In the absence of available data, this study adopts the average values reported in the study by Manfredi et al. (2009): 2 L/tMSW of diesel used for daily operation in scenarios 2-4, and 5, 8, and 12 kWh/tMSW of electricity for scenarios 2, 3, and 4, respectively. Other auxiliary materials, such as HPDE liner and gravel, are used under scenarios 2-4 and are taken as 1 and 100 kg/tMSW, respectively. To calculate GHG emissions from landfill operations, the following equation can be used (Yang et al. 2013):

$$E_{LO} = \sum_{i=1}^{i=0} A_{i,LO} \times EF_i \quad (19)$$

Where, E_{LO} is the emissions from the operation process of landfill ($kgCO_2$-eq/tMSW); $A_{i,\,LO}$ is the amount of the i^{th} auxiliary material or energy used during landfill operation; and EF_i is the emission factors for the provision of the i^{th} auxiliary material or energy and presented in Table 12.

Table 12. Emission factors used in the study

Emission factor	Value	Unit	Reference
EF for diesel fuel	2.70	$kgCO_2$-eq/L	This study
EF for electricity	0.586	$kgCO_2$-eq/kWh	IGES (2022)
EF for HDPE	1.9	$kgCO_2$-eq/kg	Friedrich and Trois (2013)
EF for gravel	0.0027	$kgCO_2$-eq/kg	Yang et al. (2013)
EF for Water	0.0002	$kgCO_2$-eq/kg	Yang et al. (2013)
EF for HCl	0.8	$kgCO_2$-eq/kg	Yang et al. (2013)
EF for NaOH	1.04	$kgCO_2$-eq/kg	Yang et al. (2013)

3.3.8. Calculation of emissions from leachate treatment

GHG emissions from leachate treatment mainly result from electricity, water, and chemical (HCl and NaOH) consumption during the treatment process. The emissions from leachate treatment are calculated following the method proposed by Yang et al. (2013) and are given in Equation (20).

$$E_{LT} = L \times \lambda \times \left[(W \times EF_w) + \left(C_i \times EF_{C_i}\right) + \left(EC \times EF_{grid}\right)\right] \quad (20)$$

Where, E_{LT} refers to the emissions from leachate treatment ($kgCO_2$-eq/tMSW); L, λ, W, C_i, and EC represent the leachate generated over 100 years of landfilling (2.2527 m^3/tMSW); the leachate collection efficiency for treatment (40%); water used for leachate treatment (83 L/m^3 leachate), the chemicals used for leachate treatment (3 and 5 kg/m^3 leachate for HCl and NaOH, respectively),

and electricity consumption for treating leachate (14.24 kWh/t leachate) (Yang et al. 2013). The emission factors for water (EF_w), chemicals (EF_{ci}), and electricity (EF_{grid}) are shown in Table 13.

Table 13. Parameters used in quantifying GHG emissions from landfilling (Yang et al. 2013)

Parameter	Definition	Unit	Value
L	Amount of leachate generated from the landfill	m^3/tMSW	2.2527
λ	Leachate collection efficiency for treatment	%	40
W	Amount of water used for leachate treatment	L	83
C_i	Amount of chemicals used for leachate treatment, which considered HCl and NaOH	kg/m^3	3, 5
EC	Amount of electricity consumption for leachate treatment	kWh/t	14.24

3.3.9. Calculation of avoided emissions from electricity substitution

The energy generated from landfill CH_4 is utilized to replace electricity produced from conventional fuels. The GHG emissions saved from electricity substitution can be calculated as follows:

$$E_{SE} = Q_{CH_4}/0.667 \times LHV_{CH_4} \times \zeta \times \eta \times \lambda \times CF/3.6 \times EF_{grid} \quad (21)$$

Where, E_{SE} is the GHG emissions saving from electricity offset; LHV_{CH_4} is the lower heating value of CH_4, taken as 37.2 (Ayodele, Ogunjuyigbe, and Alao 2017); η is the electricity conversion efficiency, taken as 30% (Amini et al. 2013); λ is the CH_4 collection efficiency, taken as 75% (Amini et al. 2013); and CF is the capacity factor of an internal combustion engine, taken as 85% (Hadidi and Omer 2017).

3.3.10. Calculation of overall emissions from landfill management technologies

The overall GHG emissions from different landfill management under the four scenarios can be calculated as:

$$E_{GHGs} = E_{FM} + E_{LO} + E_{LT} + E_{N_2O} - E_{SE} - E_{CS} \quad (22)$$

Where, E_{GHGs} is the total GHG emissions from the landfill management process.

3.3.11. Determination of uncertainty in CH_4 emissions estimation

According to the IPCC (2006), estimating CH_4 emissions involves potential uncertainties arising from both activity data and parameters. Our study addressed the uncertainties in activity data by

examining the variability of waste composition (±30%). Additionally, we evaluated parameter uncertainties by accounting for the variations in the MCF (±20%) and the fraction of CH_4 in generated LFG (±5%).

3.4. Results and discussion

3.4.1. LandGEM model

3.4.1.1. Methane generation rate constant (k) and potential methane generation (L_0)

The k value is influenced by several factors, including waste composition, moisture content, temperature, waste depth, density, pH, and other environmental conditions (Garg, Achari, and Joshi 2006; Machado et al. 2009; US EPA 2005). Different waste compositions have varying degradation rates (Machado et al. 2009). This study estimated a k value of 0.21 per year, falling within the range of other studies (Anh et al. 2021; Machado et al. 2009; Wangyao et al. 2010). Machado et al. (2009) conducted field measurements and laboratory tests on landfilled MSW in Brazil and found a good agreement of k value at 0.21. Meanwhile, Wangyao et al. (2010) and Anh et al. (2021) obtained relatively higher decay rates (k) from field measurements in Thailand and Vietnam at 0.33 and 0.355 per year, respectively. The LandGEM model recommends a default k value ranging from 0.02 to 0.7 per year, while the IPCC-recommended default value for tropical areas is 0.17 per year for bulky waste (IPCC 2006). The CH_4 generation potential (L_0) was calculated following the IPCC 2006 guidelines resulting in a value of 90 m^3/t. This value is higher than that reported in Vietnam (Anh et al. 2021) due to the higher organic fraction of MSW in Phnom Penh. However, the estimated L_0 value for Phnom Penh is lower than the recommended 96 m^3/t for the inventory wet landfill in the LandGEM model. Overall, the k and L_0 values obtained in this study fall within the acceptable ranges reported for other landfills operating in tropical regions, as shown in Table 14.

Table 14. Comparison of k and L_0 values reported in different studies

Location	k (year^{-1})	L_0 (m^3/t)	Reference
Phnom Penh, Cambodia	0.21	90	This study
Nam Binh Duong, Vietnam	0.355	81	Anh et al. (2021)
Four landfills, Thailand	0.33	-	Wangyao et al. (2010)
Sanitary landfills, Malaysia	0.072–0.136	151.7	Abushammala et al. (2014)
Delhi, India	0.05	130	Srivastava and Chakma (2020)
Andhra Pradesh, India	0.05	110	Ramprasad et al. (2022)
Salvador, Brazil	0.21	70	Machado et al. (2009)

3.4.1.2. Landfill gas generation based on LandGEM model

Typically, the methanogenesis stage occurs at least six months after waste is disposed of at a landfill site (Oukili, Mouloudi, and Chhiba 2022). However, the LandGEM model considers that waste degradation beings in the early second year after waste disposal. Therefore, the model does not account for LFG production during the first year of waste disposal. The LandGEM was used to simulate the total LFG, CH_4, CO_2, and NMOC for the Dangkao landfill over a period of approximately 140 years after its opening. The Dangkao landfill received 177,224 tMSW in 2009 and increased to 1,288,223 tMSW in 2022.

Figure 24 demonstrates the total LFG, CO_2, CH_4, and NMOC emissions from the Dangkao landfill throughout the years. CO_2 and CH_4 are the predominant gases found in the landfill. LFG emissions from the Dangkao landfill increase exponentially, corresponding to an increase in waste deposition over time, peaking one year after the landfills' closure. The highest emissions of CO_2 and CH_4 occur in 2023, with values of 111 M kg/year (60 M m^3/year) and 40 M kg/year (60 M m^3/year), respectively. This is because the landfill closure year was set to 2022, and waste continues to be deposited until the end of 2022. LFG production continues even after the landfill is closed, although it starts to decline as all degradable waste deposited in the landfill degrades. The emissions of CH_4 are relatively lower than those of CO_2 due to the difference in mass between the gases. With the application of soil cover, some CH_4 is microbiologically oxidized to CO_2, thereby reducing CH_4 emissions to the atmosphere. Factors contributing to the soil's CH_4 oxidation capacity include soil moisture content (optimum oxidation occurs at 15-20% moisture content), soil temperature (optimum oxidation occurs at 30-35 °C in soil), the presence of methanogen bacteria in the landfill, soil type, and the thickness of the cover (Njoku, Edokpayi, and Odiyo 2020). The lowest amount was related to NMOCs gas. The lowest amount of NMOC gas was observed, with a release rate of approximately 1% of CH_4 emissions. The concentration of NMOCs depends on the type of waste in the landfill and the reaction of different compounds during the anaerobic degradation process (Moghadam et al. 2021).

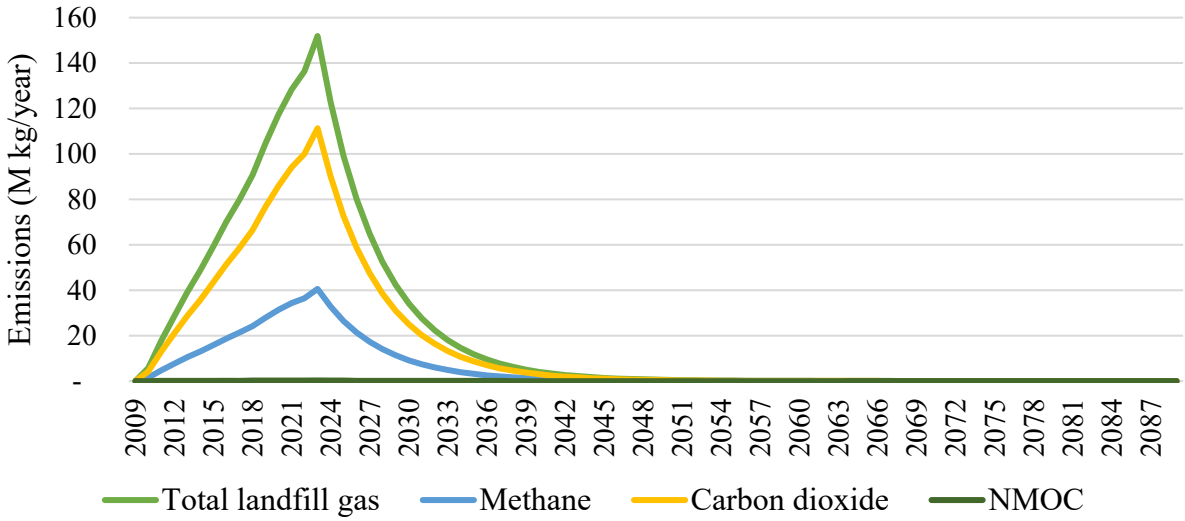


Figure 24. LFG generation simulated from the LandGEM model

The high slope of LFG generation over time in Figure 10 provides evidence supporting the fact that approximately 74% of MSW deposited in the landfill is biodegradable, resulting in a rapid initial rate of decomposition (Moghadam et al. 2021). Even after the landfill's closure in 2023, LFG production continues but at a decreasing rate since no new waste is added to the landfill. The emission of LFG gradually declines over 120 years, with emissions becoming negligible around 80 to 100 years post-closure. Despite this decline, trace amounts of LFG can still be emitted from the site for many years afterward (Njoku et al. 2020). Table 15 presents a summary of the minimum, maximum, and average LFG emissions from the Dangkao landfill between 2010 and 2022. During this period, the LandGEM model simulated average CO_2 and CH_4 emissions of 52.26 and 19.05 M kg/year, respectively.

Table 15. LFG emissions between 2010 and 2022

Emission	Unit	Total LFG	CO_2	CH_4	NMOC
Average	M kg/year	71.31	52.26	17.73	0.12
	M m^3/year	57.01	28.55	28.55	0.03
	Av ft^3/min	3,836.48	1,918.24	1,918.24	2.30
Minimum	M kg/year	5.73	4.20	1.53	0.01
	M m^3/year	4.59	2.29	2.29	0.0027
	Av ft^3/min	308.33	154.16	154.16	0.19
Maximum	M kg/year	136.42	99.98	36.44	0.24
	M m^3/year	109.24	54.62	54.62	0.07
	Av ft^3/min	7,339.54	3,669.77	3,669.77	4.40

3.4.2. Landfill gas generation based on IPCC FOD model

According to the IPCC FOD model, only CH_4 generation is calculated based on the individual waste fraction. The emission of CH_4 from the MSW landfill in Phnom Penh has been steadily increasing from 2010 to 2023, as shown in Figure 25. The results indicate that CH_4 emissions rose from 2.17 M kg in 2010 to 48.06 M kg in 2023. Similar to the LandGEM model, CH_4 generation declined from 2024 onward, as no more waste will be accepted at the landfill. Among the different waste fractions, food waste accounted for approximately 73% of the total CH_4 generation, followed by paper (9%) and textiles (7%).

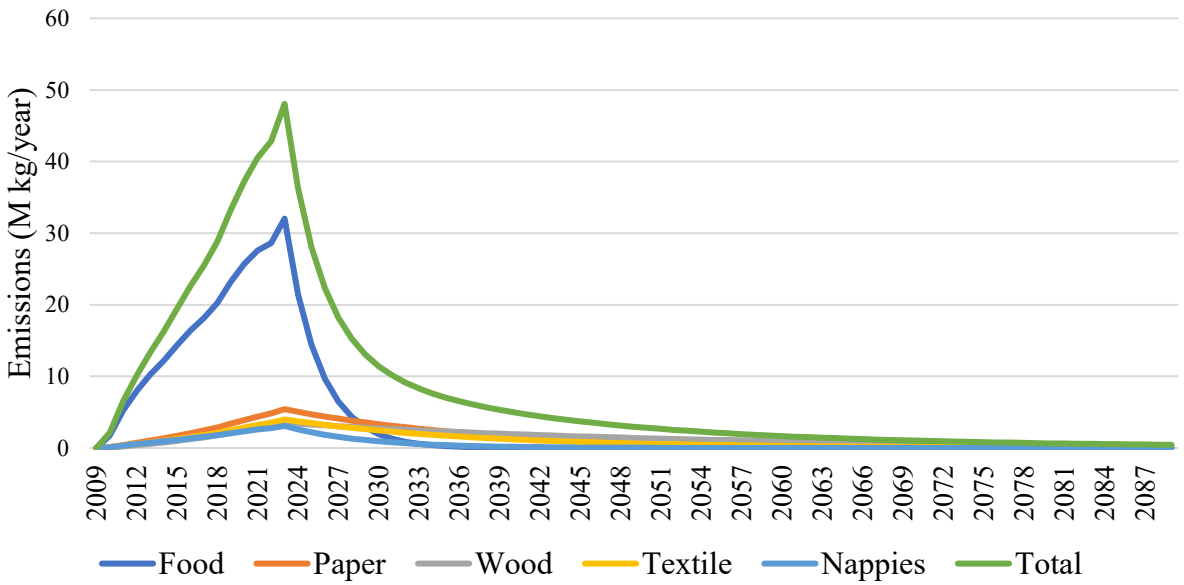


Figure 25. CH_4 generation simulated from the IPCC FOD model

3.4.3. Comparison of results from the LandGEM and IPCC FOD models

The CH_4 emissions from the Dangkao landfill were estimated using the LandGEM and IPCC FOD models based on available data of landfilled waste recorded at the landfill from 2009 to 2022. The LandGEM model typically considers the most common three landfill gases, CH_4, CO_2, and NMOC. However, since CO_2 is of biogenic origin and NMOC levels are negligible (less than 0.1% compared to CH_4), they are not included in GHG emissions quantification. Meanwhile, the IPCC FOD model potentially estimates CH_4 emissions, while CO_2 and other gases are excluded from the model. According to the LandGEM and IPCC FOD models, waste degradation begins in the early part of the second year after disposal, so LFG emissions are assumed to be zero in the initial year.

Figure 26a shows that the production of CH_4 gas increased rapidly in the early stage as MSW accumulated in the landfill. Both models showed a similar trend in CH_4 generation, but the IPCC

FOD model generated relatively higher values, 25% greater than the LandGEM model from 2010 to 2022. The potential landfill CH_4 generation was 1.54 and 2.17 M kg in 2010, increasing to 36.50 and 42.83 M kg in 2022 as calculated by the LandGEM and IPCC FOD models, respectively. However, there was a notable difference between the two models regarding when the CH_4 gas was produced. The LandGEM model predicted that approximately 53% of CH_4 generation would occur during the landfill's operation, while only 47% of CH_4 would remain after the landfill's closure. In contrast, the IPCC FOD model, which used site-specific waste composition, predicted that only 47% of CH_4 would be generated during the landfill's operation from 2010 to 2022, with approximately 53% remaining after landfill's closure. The results from both models indicate the rapid degradation of degradable matter due to the high moisture content of waste, leading to the production of more LFG in a short period, as typically found in tropical regions (Anh et al. 2021; Machado et al. 2009; Wangyao et al. 2010). Many Asian countries are known to produce high levels of LFG emissions, mainly due to the large amounts of food waste in their waste streams, the moist tropical climate, and the high precipitation in this region (Ishigaki et al. 2011). In contrast, the LFG generation rate in most European countries is lower due to the high proportion of slowly degradable fractions such as paper, wood, and yard waste, which take longer to decompose (Christensen et al. 2010). Additionally, the temperate climates and lower precipitation levels in Europe significantly impact emissions from solid waste disposal sites (Ishigaki et al. 2011).

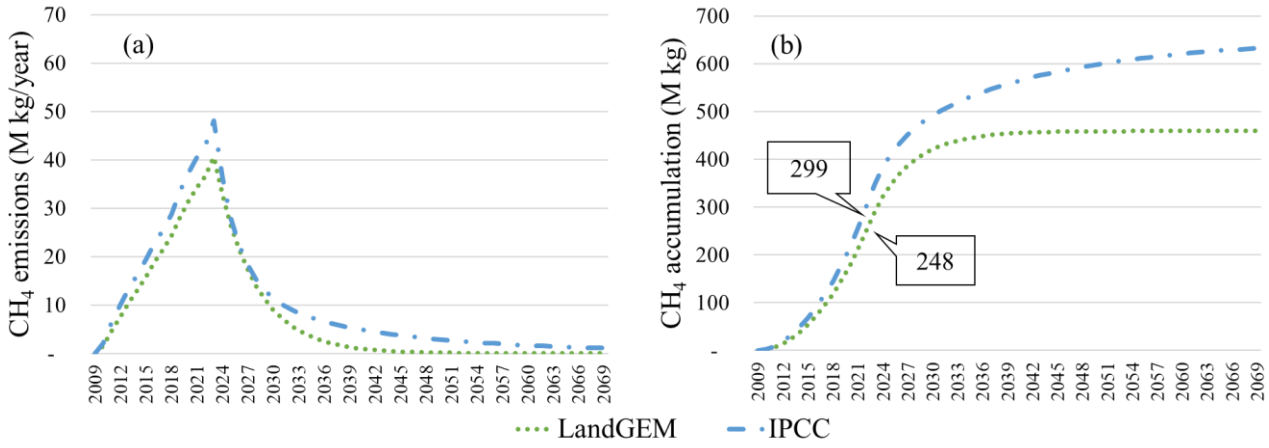


Figure 26. CH_4 emissions based on the LandGEM and IPCC FOD models

Between 2010 and 2022, CH_4 emissions from the Dangkao landfill totaled 248 M kg according to the LandGEM model, with an average emission of 19 M kg/year. According to the IPCC FOD model, the total CH_4 emissions were 299 M Kg CH_4, with an average annual emission of 21 M kg (Figure 26b). Ghosh et al. (2019) estimated CH_4 generation from three landfill sites in Delhi and found that the LandGEM model predicted over twice as much CH_4 compared to the IPCC FOD

model, which is in contrast to the present study's findings. Since there were no LFG collection systems in place and no on-site measurements data available in Cambodia, this study compared the estimated CH_4 generation results with field measurement studies conducted in neighboring countries with similar conditions. Table 16 revealed that the results obtained from the LandGEM model in the present study and the closed flux chamber method in Thailand were comparable, at 24 and 22 $kgCH_4$/tMSW, respectively. However, the IPCC FOD model in the present study estimated higher CH_4 emissions, at 29 $kgCH_4$/tMSW. On the other hand, measurements taken at the LFG collection system in Nam Binh Doung landfill showed an average of 42 $kgCH_4$/tMSW. It is important to note that the higher value in Nam Binh Doung landfill is likely associated with the measurement being taken during the peak period of CH_4 generation, which occurred one year after landfill closure. If CH_4 measurements were taken during landfill operation, the average CH_4 generation would likely be lower. Thus, these findings demonstrate the similarity between the results of this study and the field measurement results from other landfills in similar conditions. However, it is worth noting that the results of landfill CH_4 emissions can vary depending on factors such as landfill management practices, waste quantity, and waste composition (Fallahizadeh et al. 2019).

Table 16. Comparison of CH_4 emission with field measurement studies

Methods	**Location**	**Annual waste acceptance (tMSW/year)**	**Annual CH_4 emissions (tCH_4/year)**	**CH_4 emissions per unit disposal ($kgCH_4$/tMSW)**	**References**
LandGEM model	Dangkao landfill	728,379	17,727	24	This study
IPCC FOD model	Dangkao landfill	728,379	21,341	29	This study
Direct measurement at LFG collection system	Nam Binh Doung landfill	149,850	6,225	42	Anh et al. (2021)
Closed flux chamber	95 landfills in Thailand	4,444,605	98,140	22	Chiemchaisri, Juanga, and Visvanathan (2007)

Gollapalli and Kota (2018) investigated LFG emissions from a landfill in India using a flux chamber and compared them with results calculated in the Modified Triangular Method (MTM), IPCC default model, and LandGEM model. Their study found that MTM, IPCC default model, and LandGEM model predicted 1.9, 1.4, and 1.6 times higher than those measured on-site.

Furthermore, Chakraborty et al. (2011) compared the simulation results using the IPCC default model, MTM, and FOD model and found that the FOD model gave a better result, comparable to that measured in the field. Based on these previous studies, using the models to predict LFG is more likely to obtain a higher result than the field measurement. Ghosh et al. (2019) pointed out that a lower CH_4 measured on-site may be due to uncertainties in sampling selection and unaccounted-for uncontrolled emissions. However, Amini et al. (2012) noted that the LandGEM model underestimates CH_4 production. Kumar and Sharma (2014) compared several landfill models and concluded that LandGEM is the most advantageous model due to its ability to provide accurate results. These previous studies suggest that different models may yield varying levels of accuracy, with some models overestimating or underestimating CH_4 production. On the other hand, it is important to carefully plan field measurements, consider spatial and temporal variability, use appropriate measurement techniques, ensure proper calibration, and account for potential uncontrollable emissions. Combining field measurements with modeling approaches can provide a more comprehensive understanding of CH_4 emissions from landfills.

3.4.4. Estimation of energy recovery potential from LFG recovery

CH_4 generated from landfills can be harnessed for various purposes, including power production, direct use, and conversion into fuel for vehicles (Kumar and Sharma 2014b). However, this study specifically focuses on the utilization of CH_4 for electricity generation. Table 17 presents the energy generation potential estimated based on the LandGEM and IPCC FOD models. Between 2010 and 2022, the LandGEM model suggests a range of 4.10 to 97.31 GWh, with an average of 50.89 GWh/year. On the other hand, the IPCC FOD model indicates a range of 5.78 to 114.18 GWh, with an average of 61.27 GWh/year during the same period. A portion of the electricity generated can be used for on-site operations at the landfill, reducing electricity consumption costs. Any excess electricity beyond the site's requirements can be sold to the national grid, contributing to the overall energy supply. Energy recovery not only offers economic benefits but also aids in GHG mitigation through electricity substitution. According to Institute for Global Environmental Strategies (2022), the emission factor for the national grid in Cambodia is 0.586 $kgCO_2$-eq/kWh. Therefore, the electricity generation from LFG recovery could potentially avoid approximately 30 and 36 M $kgCO_2$-eq/year of GHG generated by the electricity sector in the country, based on calculations using the LandGEM and IPCC FOD model, respectively.

Table 17. Estimation of energy generation and GHGs avoided due to electricity substitution

Year	LandGEM			IPCC FOD		
	CH_4 generated (m^3/year)	**Energy recovered (GWh)**	**GHGs avoided due to electricity substitution (M $kgCO_2$-eq)**	**CH_4 generated (m^3/year)**	**Energy recovered (GWh)**	**GHGs avoided due to electricity substitution (M $kgCO_2$-eq)**
2009	-	-	-	-	-	-
2010	2,304,208	4.10	2.40	3,250,374	5.78	3.39
2011	7,182,351	12.77	7.49	9,832,574	17.49	10.25
2012	11,551,496	20.55	12.04	15,179,269	27.00	15.82
2013	15,727,845	27.97	16.39	20,008,041	35.59	20.85
2014	19,620,866	34.90	20.45	24,326,238	43.27	25.35
2015	23,869,270	42.45	24.88	29,133,918	51.82	30.37
2016	28,136,721	50.04	29.33	33,936,510	60.36	35.37
2017	32,043,991	56.99	33.40	38,230,519	68.00	39.85
2018	36,382,906	64.71	37.92	43,201,962	76.84	45.03
2019	41,932,569	74.58	43.71	49,889,424	88.73	52.00
2020	47,063,634	83.71	49.05	55,855,022	99.35	58.22
2021	51,464,896	91.54	53.64	60,785,458	108.11	63.36
2022	54,708,299	97.31	57.02	64,196,796	114.18	66.91
Average	**28,614,543**	**50.89**	**29.82**	**34,448,162**	**61.27**	**35.90**

It should be noted that LFG recovery has the potential to produce both electricity and heat using the combined heat and power (CHP) technology. By considering CHP, the overall energy efficiency of LFG recovery can be greatly enhanced (LMOP 2021). In addition to electricity, the heat generated can be utilized in industrial areas, thereby promoting economic benefits. This utilization of heat can also directly mitigate GHG emissions resulting from the combustion of firewood in boilers. In Cambodia, the use of firewood as a fuel source in boilers is common in garment sector. By diverting the heat generated from LFG recovery to these boilers, the demand for firewood can be reduced, resulting in reduced deforestation and increased forest carbon storage. These additional benefits can significantly improve energy efficiency from landfill management and contribute to sustainable development (Liang et al. 2022).

3.4.5. Quantification of emission factors for landfill scenarios

In this section, all values are expressed per ton of MSW and converted to Carbon Dioxide Equivalent (CO_2-eq) using 100-year GWP of 25 and 298 for CH_4 and N_2O, respectively. Landfills have direct GHG emissions due to the degradation of decomposable wastes under landfill conditions and the operation of the landfill site, as well as indirect emissions from the construction and installation of landfill equipment. Biogenic CO_2 emissions resulting from waste degradation in the landfill were not included in the GHG emissions accounting models. In contrast, carbon sequestration and electricity substitution, providing environmental benefits of the landfill, were taken into account in the model calculations. Table 18 indicates the emission factors for the four landfill management scenarios. Scenario 2 has the highest emissions, with net GHG emissions of 757.72 and 941.49 $kgCO_2$-eq/tMSW, according to the LandGEM and IPCC FOD models, respectively. The factor that led to higher emissions in scenario 2 over the present landfill management scenario (scenario 1) was mainly associated with the application of leachate treatment, which requires the installation of liner materials and the use of chemicals and electricity for leachate treatment, which were not present in scenario 1. This result is comparable to that found in China, where emissions ranged from 619.5 to 940.7 $kgCO_2$-eq/tMSW when applying the same landfill management practices and excluding the collection and transportation of MSW (Yang et al. 2013). The main reason for a similar result between the two countries could be associated with high CH_4 generation due to waste characteristics, with about 55% of the waste being organic.

Table 18. Quantification of emission factors for landfill scenarios ($kgCO_2$-eq/tMSW)

Activity	**Scenario 1**		**Scenario 2**		**Scenario 3**		**Scenario 4**	
	LandGEM	**IPCC**	**LandGEM**	**IPCC**	**LandGEM**	**IPCC**	**LandGEM**	**IPCC**
Fugitive CH_4	820.09	998.86	820.09	998.86	405.53	493.94	190.67	291.42
Landfill operation	1.16	1.16	10.16	11.32	11.92	13.08	14.27	15.43
Leachate treatment	-	-	14.38	14.38	14.38	14.38	14.38	14.38
N_2O emission	17.60	21.43	17.60	21.43	8.80	10.72	4.40	5.36
Electricity offset	-	-	-	-	-	-	–51.89	–63.20
Carbon sequestered	–104.51	–104.51	–104.51	–104.51	–104.51	–104.51	–104.51	–104.51
Total	**734.33**	**916.94**	**757.72**	**941.49**	**336.12**	**427.61**	**115.91**	**157.85**

Fugitive CH_4 emissions have been seen as a key factor contributing to GHG emissions from the landfill, accounting for more than 87% of GHG emissions under the four landfill scenarios. These high emissions are mainly associated with organic composition in landfilled MSW (Nordahl et al. 2020). Other contributors to GHG emissions from landfill management technologies include N_2O emissions, leachate treatment, and landfill operations. Carbon sequestration in the landfill was quantified and considered a negative emission. The amount of carbon sequestration depends on the quantity and waste composition buried in the landfill. In the present study, carbon sequestration was estimated at approximately –104.51 $kgCO_2$-eq/tMSW, which falls within the range reported by other studies (Friedrich and Trois 2013a; Yang et al. 2013). Although simple dumping in scenario 1 resulted in lower GHG emissions than scenario 2, it has the potential to pollute groundwater with untreated leachate that can seep through the soil. In contrast, scenario 2 involves installing liners and regularly applying soil cover, which can reduce liquid penetration through the waste and decrease leachate generation by approximately 50% (Yang et al. 2013).

Given that fugitive CH_4 emissions are a key contributor to GHG emissions from landfills, reducing these emissions through LFG-capturing systems for flaring or electricity generation could significantly minimize their potential contribution to global warming (Kumar and Sharma 2014b). Scenario 3 involves installing an LFG collection system for flaring, which has the potential to reduce GHG emissions by at least 55%. Previous research has shown that LFG capture systems can mitigate GHG emissions by approximately 50% (Nordahl et al. 2020). On the other hand, scenario 4 is the most effective option for reducing GHG emissions. This scenario involves capturing and utilizing landfill CH_4 as a renewable energy source for generating electricity, thereby offsetting emissions from conventional high-emission energy sources. GHG emissions in scenario 4 were reduced by at least 83%. Therefore, scenario 4 represents a promising approach for reducing GHG emissions from landfill management. However, it is important to note that installing these systems requires significant financial resources and technical expertise, which are currently unavailable in the city. While generating renewable energy from CH_4 has the potential to generate income, it remains unclear whether the LFG-to-energy project would be economically feasible for Phnom Penh.

3.4.6. Overall GHG emissions from 2009 to 2022

The emission of CH_4 from landfill sites significantly contributes to the impact of global warming because its global warming potential is 25 times greater than that of CO_2, according to IPCC (2006). When quantifying GHG emissions from the landfill site, not only the direct emissions from the

landfill operation and leachate treatment are taken into account, but also the emissions avoidance from carbon sequestration and the emission savings from electricity substitution. In the landfill management process, emissions of CH_4 that escape into the atmosphere comprise approximately 99 % of total GHG emissions under the present scenario (Figure 27). Therefore, mitigating these emissions could effectively reduce the potential contribution of these gases to global warming (Kumar and Sharma 2014b). The second-highest emissions came from leachate treatment, which consumes significant electricity, water, and chemical inputs. The use of heavy-duty vehicles for landfill management was the third-largest emission source. GHG emissions due to N_2O emissions are minimal, accounting for roughly less than 1% in all scenarios.

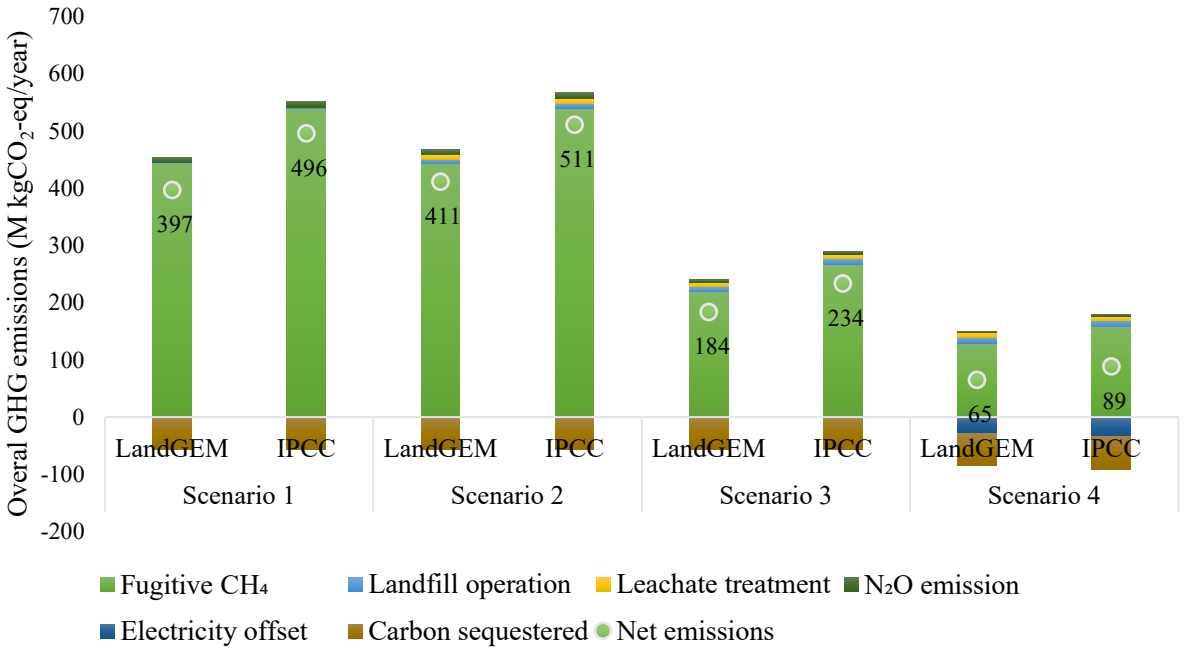


Figure 27. Overall GHG emissions under different scenarios based on the two models

Carbon sequestration in the landfill was estimated and considered as a negative emission. The amount of carbon sequestration depends on the quantity and waste composition buried in the landfill. The avoided emissions resulting from carbon sequestration could potentially save 56.48 M $kgCO_2$-eq/year under each of the four scenarios. The CH_4 generated can be collected and utilized as a renewable energy resource. The present study considered collecting 75% of CH_4 for electricity production, which provides both environmental benefits and contributes to socio-economic development. In scenario 4, GHG emissions saved from electricity substitution amounted to 12.32 M $kgCO_2$-eq/year and 15.01 M $kgCO_2$-eq/year, corresponding to the CH_4 collected under the LandGEM and IPCC FOD models, respectively.

Figure 28 indicates that the average GHG emissions resulting from the current landfill management practices (scenario 1) between 2009 and 2022 were 397 and 496 M $kgCO_2$-eq/year, as determined by the LandGEM and IPCC FOD models, respectively. With the implementation of leachate collection and treatment in scenario 2, GHG emissions increased and averaged 409 M $kgCO_2$-eq/year for the LandGEM model and 509 M $kgCO_2$-eq/year for the IPCC FOD model. Scenarios 3 and 4 led to a significant reduction in GHG emissions due to the destruction of CH_4 through flaring and electricity generation. According to the LandGEM model, scenario 3 resulted in an average of 182 M $kgCO_2$-eq/year emissions. In contrast, the value calculated based on the IPCC FOD model was higher, at 231 M $kgCO_2$-eq/year. By using electricity substitution in scenario 4, GHG emissions were further reduced to 63 and 86 M $kgCO_2$-eq/year, based on the LandGEM and IPCC FOD models, respectively.

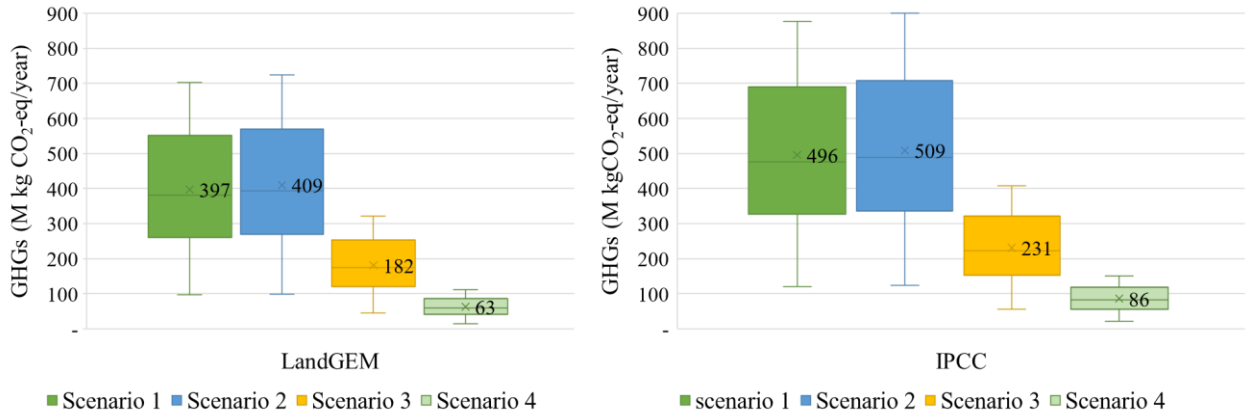


Figure 28. Average annual GHG emissions based on the LandGEM and IPCC FOD waste models

3.4.7. Uncertainty assessment

Based on the above findings, it is evident that landfill CH_4 predominately contributes to GHG emissions from landfill management. Therefore, uncertainties in CH_4 estimation could influence GHG emissions accounting. The uncertainty in predicting landfill CH_4 emissions is mainly caused by a lack of precise and reliable data (IPCC 2006). One of the primary sources of uncertainty is the characteristics of landfilled MSW. The composition of MSW disposal can vary over time and is influenced by factors such as consumption habits, income status, socio-economic, etc. (Dek et al. 2022). Improvements in waste collection systems and proper source segregation can also impact MSW characteristics. The changing composition of MSW affects the total amount of DOC in the landfill (IPCC 2006), which in turn affects the estimation of CH_4 emissions. The uncertainty in CH_4 emissions due to MSW composition has been estimated to be around ±30%, as recommended

by the IPCC 2006 guidelines. As shown in Figure 29, the LandGEM and IPCC FOD models exhibit different levels of uncertainty in predicting CH_4 emissions when waste composition varies. The LandGEM model demonstrates higher sensitivity to changes in waste composition, indicating that CH_4 yields could double with a 30% increase in decomposable waste, while a 30% decrease in biodegradable waste could lead to a 45% reduction in landfill CH_4 generation. On the other hand, the IPCC FOD model shows a lower rate of CH_4 variation when waste composition fluctuates. A 30% decrease in biodegradable waste composition resulted in a 28% reduction in CH_4 yields, and vice versa. These results clearly highlight the significant influence of MSW characteristics, particularly food waste, on landfill CH_4 generation. Therefore, minimizing the landfilling of degradable waste becomes crucial in reducing the substantial amount of landfill CH_4 emissions.

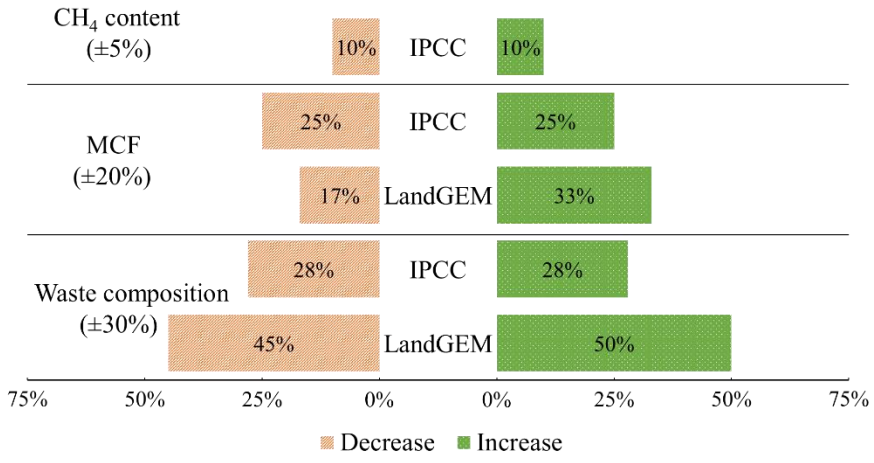


Figure 29. Uncertainty assessment in landfill CH_4 estimation

Another source of uncertainty in CH_4 prediction is associated with the selection of landfill types. In the present study, an unmanaged-deep landfill (MCF=0.8) was chosen, as the landfill pit depth ranges from 10 to 30 m. The LandGEM model demonstrated that shifting the MCF to 0.6 for uncategorized landfill reduced the CH_4 generation potential by 17%. Conversely, increasing the MCF to 1 for a managed-aerobic landfill increased the CH_4 potential by 33%. In the IPCC FOD model, the uncertainty caused by the variation in the MCF resulted in a variation of CH_4 emissions of approximately ±25%.

According to IPCC (2006), the fraction of CH_4 in generated LFG can vary by ±5% from the default value of 50%, depending on several factors. The uncertainty in CH_4 emissions resulting from variations in the CH_4 content ranged between 19 and 24 M kg/year based on the IPCC FOD model. Therefore, the availability of data can have a significant impact on the output value, as indicated

by the results, showing that uncertainty in available data can lead to huge variations in results (Ghosh et al. 2019).

3.5. Limitations of the study

Variation in waste composition can result in high uncertainty in CH_4 estimation results. Waste composition in Phnom Penh may have changed over time; however, the time series data on waste composition is not available at the Dangkao landfill. Furthermore, due to difficulties in arranging manpower, coordinating waste sampling with the timing of waste disposal, and other logistical challenges, the determination of waste compositions was not carried out in this study. Therefore, the physical composition of MSW in Phnom Penh from a scientific study conducted in 2014-2015 was used to simulate CH_4 generation. This could be considered one of the limitations of the study. Another limitation in the present study is the lack of field measurements to validate the estimated results. On-site measurements can help identify and account for factors that may not be adequately captured in the estimation models. However, the Dangkao landfill does not have a landfill gas collection system and has partial soil cover, leading to high uncontrolled emissions. Therefore, caution should be exercised when selecting a field measurement method. In future research, it would be beneficial to incorporate actual measurements to improve the accuracy of CH_4 estimation.

3.5.1. Comparison of advantages and disadvantages of the Models

LandGEM model	**IPCC FOD model**
- The model simulates approximately 51 out of the 300 gases that are emitted from a landfill.	- The model can only simulate the CH_4 emissions from a landfill, which can pose a challenge when using IPCC FOD model since it cannot simulate CO_2 and other trace gases.
- The model does not provide an estimate of the volume percentage of NMOC; it only allows for the estimation of CH_4 and CO_2.	- The model only accounts for CH_4 gas.
- The model is capable of simulating LFG emissions for up to 140 years after landfill is opened.	- The model simulates CH_4 gas emissions for a period of 80 years only.
- The model does not permit the estimation of LFG emissions from individual waste compositions.	- The model is capable of estimating CH_4 generation based on the different compositions, which helps to better

	understanding the degradability of the waste deposited.
- The model does not allow for variation in the CH_4 content in LFG; as it is fixed at 50%.	- While the model allows for CH_4 content adjustment, the recommended variation ±5% can be applied.
- The model does not consider seasonal changes at the landfill site.	- The model does not consider seasonal changes at the landfill site.

3.6. Conclusions

This study estimated GHG emissions resulting from landfill management, which includes direct emissions of landfill CH_4, indirect emissions from landfill operation, N_2O emissions, and GHG emissions avoidance from carbon sequestration and CH_4 utilization. Firstly, landfill CH_4 generation was predicted for the year 2009 to 2022 using two well-known models, namely LandGEM and IPCC FOD models. Both models indicated a similar trend of rapid CH_4 degradation soon after disposal, but the LandGEM model produced lower estimates than the IPCC FOD model. Landfill CH_4 generation was estimated at approximately 19 and 21 M kg/year based on the LandGEM and IPCC FOD models, respectively. The potential of CH_4 emission could substantially affect global warming, given that its GWP is 25 times greater than CO_2. In quantifying net GHG emissions, four landfill management scenarios were proposed. Scenario 1 represented the current situation with no leachate treatment and no landfill gas collection, while scenario 2 involved leachate treatment but no landfill gas collection. Scenarios 3 and 4 included leachate treatment and LFG collection for flaring and electricity production. GHG emissions in scenario 2 were the highest, averaging 409 and 509 M $kgCO_2$-eq/year according to the LandGEM and IPCC FOD models, respectively. The high emissions in scenario 2 were mainly due to additional emissions from the leachate treatment process. However, through LFG collection for flaring and electricity production, scenarios 3 and 4 could significantly reduce GHG emissions by at least 55% and 83%, respectively. Estimating CH_4 emissions from landfills can be challenging due to factors such as waste composition, selection of proper landfill type, and CH_4 content in LFG. Therefore, field measurements of CH_4 emissions are necessary to validate the model calculations and reduce uncertainty. Scenario 4 was the most favorable option for GHG mitigation and energy recovery. Further study is needed to evaluate the economic feasibility of the LFG recovery project.

CHAPTER 4

EVALUATION OF POTENTIAL IMPACTS OF MUNICIPAL SOLID WASTE MANAGEMENT ALTERNATIVES

4.1. Introduction

MSW management is an emerging concern that has drawn the attention of not only managers and stakeholders but also nations worldwide. Globally, about two billion tMSW are generated annually (Amoo and Fagbenle 2013; Ghosh et al. 2019), and this figure is expected to increase to 9.5 billion by 2050 (Pham et al. 2015). The rapidly increasing trend of MSW generation is influenced by high urbanization, population growth, and socio-economic development (Chen and Lo 2016). Waste management contributes about 5% of the total anthropogenic GHG emissions (Bogner et al. 2007). GHGs, primarily CO_2 of biogenic and fossil origins, CH_4, and N_2O, can be generated throughout the entire process of the MSWM system, from temporary storage to the final disposal (Gentil, Christensen, and Aoustin 2009). Among the various processes, transportation was found to have minor environmental impacts on GHGs (Chen and Lo 2016; Kristanto and Koven 2020; Xin et al. 2020).

Landfilling is the most commonly used method for MSW management in developing countries, thanks to its low cost and simple operation, which does not require skilled experts or advance technologies (Christensen et al. 2010; Kumar et al. 2004; Kumar and Sharma 2014a). However, landfills are the primary source of GHGs, mainly anthropogenic CH_4 (50-60%) and CO_2 (40-50%) (Scheutz and Kjeldsen 2019). It is worth noting that CO_2 emissions from landfills are considered biogenic rather than anthropogenic GHG (IPCC 2006). In developed countries, MSW has been used as a resource to produce energy, heat, and fuel (Ghosh et al. 2019). Numerous MSW treatment technologies have been developed to recover resources from waste, such as AD, pyrolysis, composting, incineration, mechanical biological treatment, gasification etc. These treatment technologies have the potential to mitigate GHG emissions and reduce the environmental burden at landfills (Chen and Liu 2021; Kristanto and Koven 2020; Xin et al. 2020).

Many studies have been conducted to assess the GHG emissions from the MSW management systems by proposing alternative management strategies and comparing the results of environmental impacts with the baseline scenario (Bernstad and La Cour Jansen 2012; Chen and Lo 2016; Kristanto and Koven 2020; Liu et al. 2017; Thanh and Matsui 2013; Xin et al. 2020).

Among these studies, researchers have reported that unmanaged and managed landfills produced the highest GHG emissions compared to composting, incineration, AD, and material recovery facilities (MRFs). Previous studies have suggested that incineration, AD, and composting are alternative technologies to reduce waste landfilling and mitigate LFG emissions (Bernstad and La Cour Jansen 2012; Kristanto and Koven 2020; Liu et al. 2017). Recycling of recyclables and electricity generation from incineration and AD can significantly reduce a vast amount of GHGs compared to their emissions (Chen and Lo 2016; Kristanto and Koven 2020; Xin et al. 2020). However, none of these studies included emissions from operations that use fuel and electricity in quantifying GHG emissions from treatment processes. Furthermore, very few researchers included GHG emissions from transportation in their study.

Cambodia generates 4.78 M tMSW, with approximately 44% of it being collected and sent to landfills, while recycling, composting, and incineration account for less than 10% (Dek et al. 2022). The remaining waste ends up in rivers, vacant land, or is burned (Dek et al. 2022; Spoann et al. 2019). Phnom Penh municipality, being the largest city in terms of population, contributes around 25% of the country's MSW generation. The municipality is facing a significant challenge as daily waste disposal at the landfill has more than doubled in the last decade (Dangkor landfill Authority 2021). Waste separation is limited despite the municipality’s rule for separation obligation, leading to the disposal of commingled waste. Food waste makes up a large portion (49%) of the MSW landfilled, and it has a high potential for GHG generation through biological decomposition (Mallinson, Russell, and Barker 2016).

There have been few studies on GHG emissions from the MSW management system in Cambodia. Most of the research has focused on emissions from one or two management methods rather than considering the entire system. Additionally, many studies lacked an analysis of uncertainties associated with various parameter values. For example, Hoklis and Sharp (2014) conducted a study on GHG emissions from the landfill and compared the results with the environmental impacts of composting. They proposed two alternative scenarios with variations in the disposal of recyclable materials and organic waste in the landfill, but they did not consider the environmental benefits of recycling and energy substitution. Similarly, Seng et al. (2013) investigated the benefits of composting by utilizing organic waste from households, markets, restaurants, schools, and hotels and compared it with GHG emissions from the landfill. Meanwhile, Seng et al. (2018) analyzed the characteristics of MSW in the landfill and identified suitable treatment technologies based on the chemical and physical characteristics of the waste. However, there is a need for more

comprehensive studies that consider the entire MSW management system and include uncertainties in the analysis of GHG emissions.

Therefore, this study aims to investigate the GHG emissions from the current MSW management practices and explore alternative scenarios that have the potential to reduce the sector's environmental impact. The quantification of GHG emissions from MSWM follows a life cycle assessment approach and utilizes the 2006 IPCC guidelines. This analysis is based on field data collected, various databases, and reliable publications.

4.2. Materials and Methods

4.2.1. Status of MSWM in Phnom Penh municipality

Phnom Penh municipality has a population of 2,281,951 (as of 2019) (National Institute of Statistics 2020). In 2022, a total of 3,530 tMSW was sent to Dangkao landfill daily, which amounts to approximately 1.38 kg/capita/day. With a collection efficiency of 92%, the MSW generation rate was estimated to be about 1.50 kg/capita/day, showing an increase from 1.32 kg/capita/day in 2018 (PPCA et al. 2018). The composition of MSW in Phnom Penh municipality is predominantly organic waste (55.87%), followed by plastic (21.13%), textile (8.02%), and paper (6.54%), as shown in Table 19. Organic waste, such as food waste, wood, and leaves, can be sent to composting and AD plants, while plastics, paper, glass, and metals can be collected for recycling.

Table 19. MSW characteristics and its properties

Composition	Fraction in wet basis (%) [a]	Moisture (%) [a]	LCV (MJ/kg) [a]	Total carbon in dry weight (%) [b]	Fossil carbon fraction (%) [b]
Food waste	49.18	78.77	0.33	38.00	-
Wood and leaves	6.69	57.12	0.56	49.00	-
Mixed paper	6.54	63.61	4.04	46.00	1.00
Plastic	21.13	18.37	20.70	75.00	100.00
Metals	1.05	-	-	N/A	N/A
Glass	1.42	-	-	N/A	N/A
Rubber and leather	0.87	18.09	22.37	67.00	20.00
Textile	8.02	44.28	14.87	50.00	20.00
Nappies	2.91	58.29	4.49	70.00	10.00
Others	2.19	22.73	3.84	3.00	50.00

[a]Seng et al. (2018); [b]IPCC (2006); N/A, data not available

4.2.2. System boundary and functional unit

The present study comprehensively assessed GHG emissions arising from various sources within the MSW management system, encompassing the following components: (1) transportation of MSW from its generation sources to the designated landfill site and treatment facilities, (2) operation and decomposition of MSW in composting plant, AD plant, and landfill, (3) combustion processes, including both open burning and incineration, (4) emission savings resulting from electricity generation through incineration and biogas plants, and (5) emissions avoidance achieved through recycling, substituting virgin products (Figure 30).

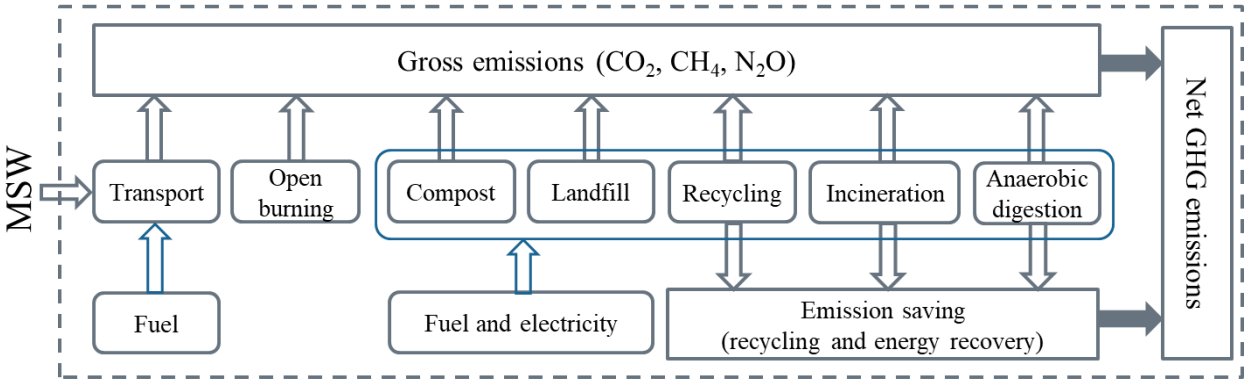
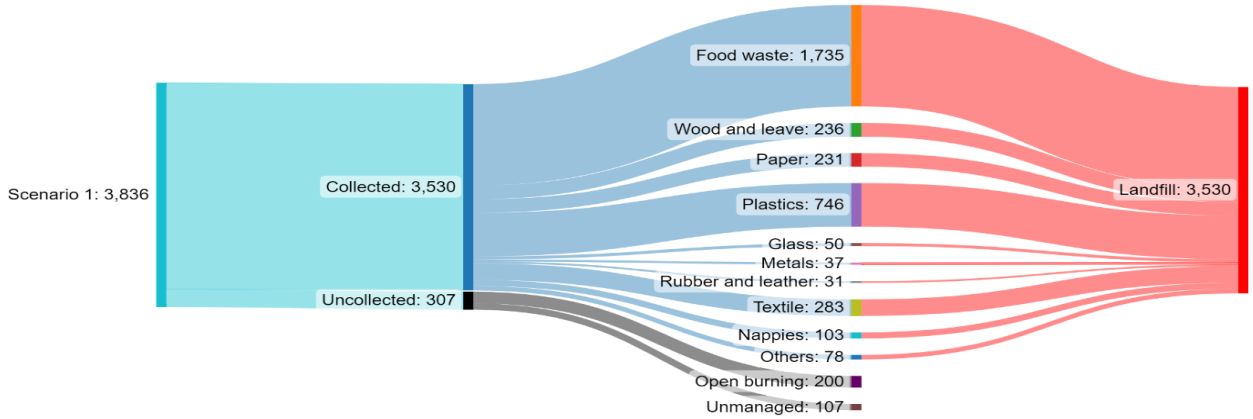


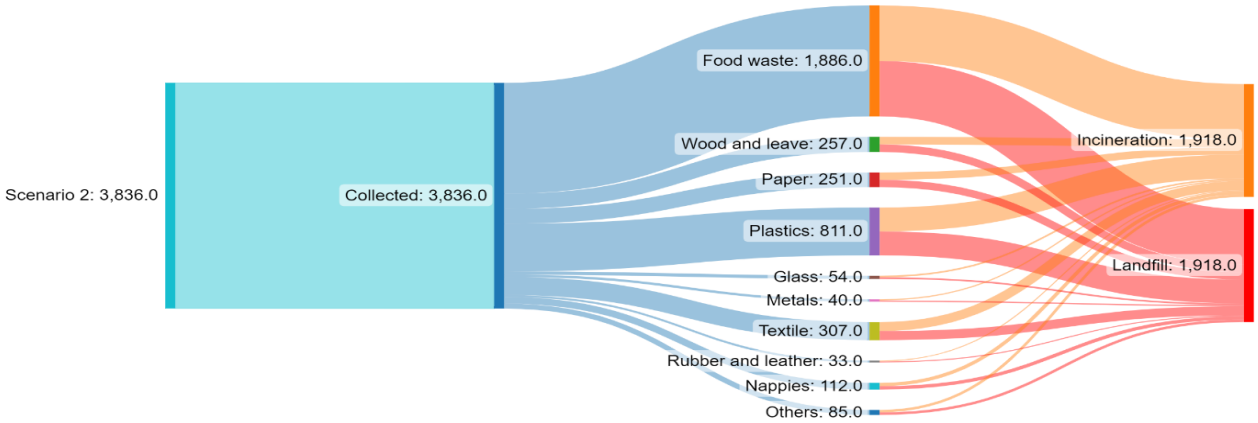
Figure 30. The system boundary of the study

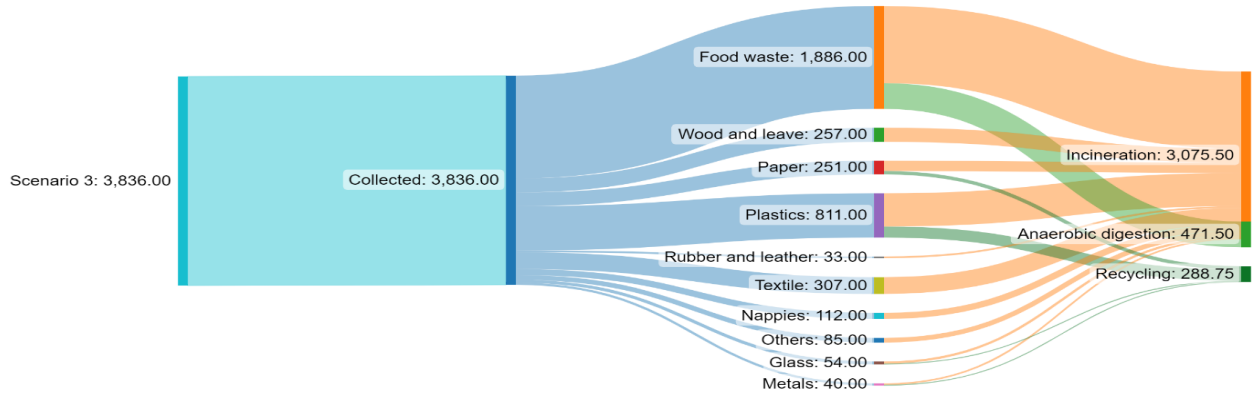
In order to quantify GHG emissions accurately, the study adopted a functional unit of 3,530 tMSW/day. This standardized unit facilitated a consistent and reliable basis for comparing and evaluating the environmental impacts of various MSW management strategies.

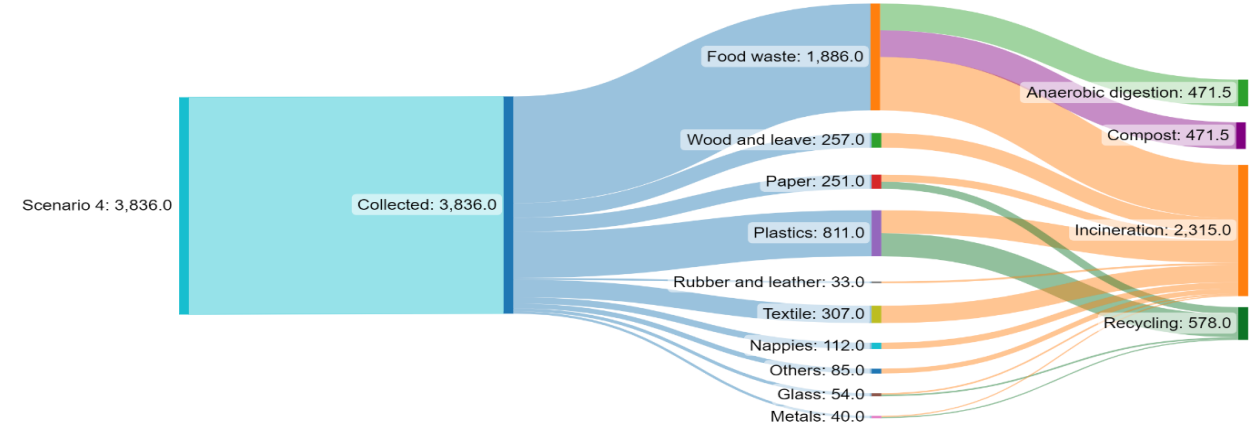
4.2.3. Scenario design

Five distinct MSW management scenarios were formulated and analyzed in this study. Scenario 1 represents the current MSW management system, featuring a collection rate of 92%. Scenarios 2-5 are devised with a 100% collection efficiency and are guided by the waste composition for suitable treatment options. In scenario 2, mixed waste disposal is considered, while in scenarios 3, 4, and 5 involved the segregation of MSW into two categories: food waste and recyclables. The material flows managed by each treatment method under the various scenarios are visually summarized in Figure 31.


Scenario 1: 3,836
Collected: 3,530
Uncollected: 307
Food waste: 1,735
Wood and leave: 236
Paper: 231
Plastics: 746
Glass: 50
Metals: 37
Rubber and leather: 31
Textile: 283
Nappies: 103
Others: 78
Open burning: 200
Unmanaged: 107
Landfill: 3,530


Scenario 2: 3,836.0
Collected: 3,836.0
Food waste: 1,886.0
Wood and leave: 257.0
Paper: 251.0
Plastics: 811.0
Glass: 54.0
Metals: 40.0
Textile: 307.0
Rubber and leather: 33.0
Nappies: 112.0
Others: 85.0
Incineration: 1,918.0
Landfill: 1,918.0


Scenario 3: 3,836.00
Collected: 3,836.00
Food waste: 1,886.00
Wood and leave: 257.00
Paper: 251.00
Plastics: 811.00
Rubber and leather: 33.00
Textile: 307.00
Nappies: 112.00
Others: 85.00
Glass: 54.00
Metals: 40.00
Incineration: 3,075.50
Anaerobic digestion: 471.50
Recycling: 288.75


Scenario 4: 3,836.0
Collected: 3,836.0
Food waste: 1,886.0
Wood and leave: 257.0
Paper: 251.0
Plastics: 811.0
Rubber and leather: 33.0
Textile: 307.0
Nappies: 112.0
Others: 85.0
Glass: 54.0
Metals: 40.0
Anaerobic digestion: 471.5
Compost: 471.5
Incineration: 2,315.0
Recycling: 578.0

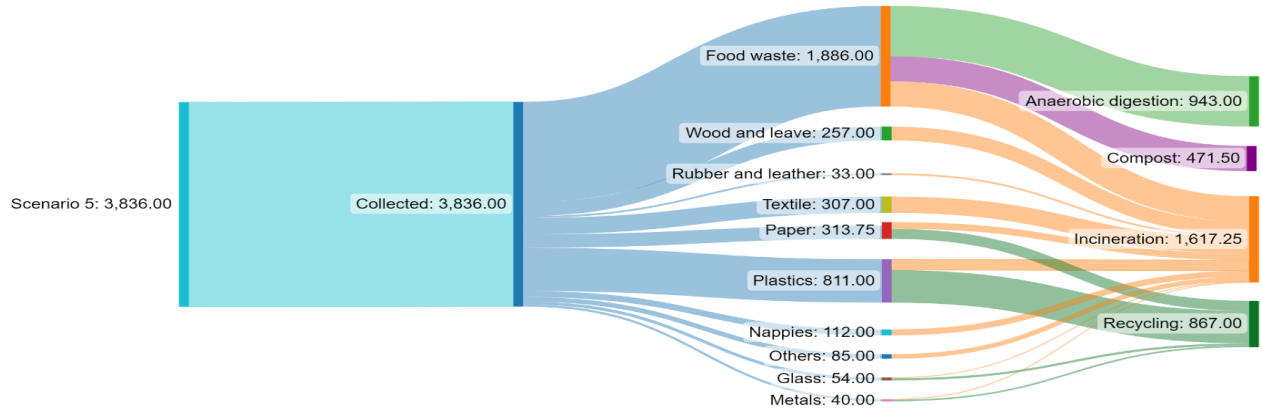


Figure 31. MSW management under different scenarios

Scenario 1: This is the baseline scenario in which the collected MSW is landfilled (3,530 tMSW/day) and open burned (200 tMSW/day). Open burning is assumed to be 64% uncollected (Seng et al. 2018). Transportation of waste is considered only for the collected waste, with an average distance of 18 km from the central city to the landfill. Uncollected waste is not taken into account when quantifying GHG emissions from transportation, since it's self-disposal and does not require transportation.

Scenario 2: This scenario assumes no waste separation is in place. Incineration is used to process 50% of mixed waste, and the remaining waste is sent to landfill without treatment. This scenario does not consider open burning, but transportation is considered for sending waste to incineration plants and the landfill. Approximately 20% of bottom ash is also sent to the landfill. The average distance to each location is assumed to be 18 km.

Scenario 3: This scenario assumes a 25% waste separation rate. Waste is separated into food waste and recyclable. The separated food waste and recyclables are sent to the AD and recycling facilities, respectively, while the remaining comingled waste is used as feedstock for the incineration plant. Transportation is considered for sending waste to each treatment facility as well as for sending the rejects from AD (50%) and recycling (13%) and incineration bottom ash to the landfill. It is assumed there are no further emissions from these residuals.

Scenario 4: This scenario aims to improve the separation efficiency by 50%. In this scenario, food waste is treated by composting and AD in equal proportion. The amount

of recycling has increased, along with improved waste separation. The remaining comingled waste is incinerated for energy. However, the amount of incinerating is lower than in scenario 3, owing to increased separation. The quantification of GHG emissions from transportation is similar to scenario 3, but there is additional transportation of waste to composting plants, as well as the reject from composting (40%)[1], which needs to be disposed of at the landfill.

Scenario 5: This scenario aims to achieve optimal resource use by increasing the separation rate to 75%. Like scenario 4, the separated waste is treated by composting, AD, recycling, and incineration. The amount of waste composted remains the same, but the amounts for recycling and AD increase. In contrast, the amount of feedstock sent to the incineration plant is lower than in scenarios 2, 3, and 4. All rejects and bottom ash are sent to the landfill for final disposal.

4.2.4. GHG emissions from the MSWM system

The GHG emissions from the MSW management system represent the net GHG emissions (expressed as a CO_2-eq value) resulting from the gross emissions being subtracted by emissions saving. The quantification of GHG emissions was given as follows.

$$E_{Net} = E_{Gross} - E_{saving} \quad (23)$$

$$E_{Gross} = E_i + \left(FC \times LHV_{fuel} \times EF_{fuel}\right) + (EC \times EF_{electricity}) \quad (24)$$

$$E_{saving} = ERP_i \times EF_{electricity} \quad (25)$$

Where, E_{Net} is the net GHG emissions; E_{Gross} is the gross GHG emissions; E_{saving} is the GHG emission avoidance/saving; E_i is the GHG emissions from the treatment process *i*; FC is the volume of diesel consumption; LHV_{fuel} is the lower heating value of diesel (36.12 MJ/L); EF_{fuel} is the emission factor for diesel (0.0741 $kgCO_2$/MJ); EC is the electricity consumption; $EF_{electricity}$ is the emission factor for electricity (0.586 $kgCO_2$-eq/kWh) (Institute for Global Environmental Strategies 2022); and ERP_i is the energy recovery potentials from WTE plant.

4.2.5. Emissions from MSW transportation

To estimate the emissions from transportation, we used a method developed by (Chen and Lo 2016), given in the following equations, where E_{tran} represents the transportation emissions, *d* is

[1] http://www.epem.gr/waste-c-control/database/html/Composting-01.htm

the average distance (km), EF_{tran} is the emission factor for transportation. The fuel consumption rate (FC) was taken as 0.1858 L/km/tMSW (Outapa and Na Roi-et 2018). The transportation distance between each treatment facility was assumed to be equal to the distance to the landfill site, 18 km.

$$E_{tran} = \sum(EF_{tran} \times d \times MSW) \quad (26)$$

$$EF_{tran} = (FC \times LHV_{fuel} \times EF_{fuel}) \quad (27)$$

4.2.6. Emissions from compositing

Following the IPCC 2006 guidelines, the EF for CH_4 and N_2O for composting was set as 4 kg CH_4/tMSW and 0.3 kg/tMSW (IPCC 2006). The diesel and electricity consumptions for one ton of waste composting are taken as 0.065 L and 5.7 kWh (Lu et al. 2021). Calculation of GHG emissions from composting plant is given as:

$$E_{compost} = M_{compost} \times (EF_{CH_4} \times GWP_{CH_4} + EF_{N_2O} \times GWP_{N_2O}) \quad (28)$$

4.2.7. Emissions from open burning and incineration

The emissions from open burning and incineration were calculated following the IPCC 2006 guidelines and given as follows:

$$E_{combustion} = CO_{2emisison} + N_2O_{emission} + CH_{4emission} \quad (29)$$

$$CO_{2emission} = \sum(MSW_i \times dm_i \times CF_i \times FCF_i \times OF) \times 44/12 \quad (30)$$

$$N_2O_{emission} = \sum(MSW_i \times EF_{N_2O}) \times GWP_{N_2O} \quad (31)$$

$$CH_{4emission} = \sum(MSW_i \times EF_{CH_4}) \times GWP_{CH_4} \quad (32)$$

Where, $E_{combustion}$ is the emission factor for open burning and incineration; dm_i is the dry matter content in each waste constituent i; CF_i is the total carbon content in dry matter of waste constituent i; FCF_i is the fraction of fossil carbon in each waste constituent i; OF is the oxidation factor which equals the combustion efficiency and taken as 58% for open burning and 100% for incineration (IPCC 2006); 44/12 is the conversion factor from carbon to CO_2; EF_{N_2O} and EF_{CH_4} are the emission factor for N_2O and CH_4 and taken as 0.15 kgN_2O/tMSW and 6.5 kgCH_4/tMSW for open burning and 0.05 kgN_2O/tMSW and 6 kgCH_4/MSW for semi-continuous stoker incineration (IPCC 2006).

The fractions of waste and their lower heating value significantly influence the GHG emissions and energy recovery potential (ERP) from incineration. Table 20 demonstrates the variations in waste fractions due to enhanced separation rates. Incineration typically yields both heat and electricity; however, this study solely focused on electricity generation, which was computed using the following equation (Xin et al. 2020). The produced bottom ashes are then sent to a landfill and assumed to constitute 20% of the total waste incinerated (Chen and Lo 2016).

$$ERP_{inc} = MSW_i \times LHV_i \times \frac{1000}{3600} \times \eta \times CF \quad (33)$$

Where, ERP_{inc} is energy recovery potential from incineration; LHV_i is the lower heating value of waste fraction i (see Table 1); η is the electricity conversion efficiency which is taken as 25% (Chakraborty et al. 2013); CF is the capacity factor of an incinerator, taken as 80% (Chakraborty et al. 2013); 1000 and 3600 is the conversion factors from kg to ton and kJ to kWh.

Table 20. Variation in waste fraction due to separation efficiencies

Component	Scenario 1	Scenario 2	Scenario 3	Scenario 4	Scenario 5
Kitchen waste	49.18%	49.18%	46.00%	40.74%	30.34%
Wood	6.69%	6.69%	8.34%	11.08%	16.51%
Paper	6.54%	6.54%	6.12%	5.42%	4.03%
Plastics	21.13%	21.13%	19.76%	17.50%	13.03%
Glass	1.42%	1.42%	1.33%	1.18%	0.88%
Metal	1.05%	1.05%	0.98%	0.87%	0.65%
Rubber and leather	0.87%	0.87%	1.11%	1.53%	2.45%
Textile	8.02%	8.02%	10.00%	13.29%	19.79%
Nappies	2.91%	2.91%	3.63%	4.82%	7.18%
Others	2.21%	2.21%	2.76%	3.66%	5.45%

4.2.8. Emissions from recycling

The recyclable materials are predominantly collected by waste collectors and scavengers. Paper waste made up about 34%, while ferrous, plastics, aluminum, and glass account for about 29%, 21%, 15%, and 1%, respectively (MoE 2021a). The GHG emissions and emission savings resulting from the utilization of recycled materials instead of virgin materials were computed using the following equations, with the EF adopted from Turner et al. (2015).

$$E_{operation} = \sum(R_i \times EF_i) \quad (34)$$

$$E_{saving} = \sum(R_i \times E_{saving(i)} \times RC) \quad (35)$$

Where, R_i represents the mass of recyclable component *i*; *RC* is the recyclability of recyclable waste, which is assumed to be 87%, and an average of 13% reject is sent to landfill (Turner et al. 2015); EF_i denotes the emission factor for recycling of materials, and $E_{saving(i)}$ is the emission avoidance from energy saved by using recycled materials *i* instead of virgin materials. The emission factor for recycling is presented in Table 21.

Table 21. Values for quantifying GHG emissions and emission savings from recycling

Recycling materials	Fraction (%)	Rejects (%)	Emission Factors for recycling process ($kgCO_2$-eq/kg recyclable)	Emissions avoidance from using recycled materials ($kgCO_2$-eq/kg recyclable)
Glass	0.46	94	395	709
Aluminum	14.91	95	1,113	9,256
Steel	29.16	86	883	4,460
Plastic	21.08	70	339	1,363
Paper	34.40	90	559	679

4.2.9. Emissions from anaerobic digestion

In this study, only food waste is considered as the feedstock for AD. The biogas yield resulting from experimental studies was found to be 1,550 L/kg of mixed food waste with a CH_4 content of 30% (Al-Wahaibi et al. 2020). The amount of CH_4 generated from food waste digestion and its ERP can be calculated as follows:

$$Q_{CH_4(AD)} = (M_{food\ waste} \times dm \times Yield_{biogas} \times F_{CH_4})/1000 \quad (36)$$

$$ERP_{AD} = (Q_{CH_4(AD)} \times LHV_{CH_4} \times \eta \times \lambda \times CF)/3.6 \quad (37)$$

Where, Q_{CH_4} is the CH_4 generated from AD plant; $M_{food\ waste}$ is the mass of feedstock which is food waste; *dm* is the dry matter content of food waste, taken as 21.23% (Seng et al. 2018); $Yield_{biogas}$ is the biogas yield (1,550 L/kg); F_{CH_4} is the methane content in biogas (30%); ERP_{AD} is the energy recovery potential from AD plant; LHV_{CH_4} is 37.2 MJ/m^3; and η is the conversion efficiency which is taken as 30% for the internal combustion engine (Chakraborty et al. 2013); λ is the CH_4 collection efficiency which is taken as 95% (IPCC 2006); *CF* is the capacity factor of a plant over

year operation and taken as 85% (Hadidi and Omer 2017); and 3.6 is the conversion factor from MJ to kWh.

GHG emissions from the AD plant occur through the emission of CH_4, which is considered to be 5% of the total biogas generated, as well as the consumption of electricity and diesel fuel to run the system. Therefore, the fugitive CH_4 emissions from the AD plant can be calculated as follows:

$$E_{Fugitive} = Q_{CH_4(AD)} \times (1 - \lambda) \times \rho_{CH_4} \times GWP_{CH_4} \quad (38)$$

$$E_{Operation(AD)} = \left(FC \times LHV_{fuel} \times EF_{fuel}\right) + (EC \times EF_{electricity}) \quad (39)$$

Where, $E_{Fugitive}$ is the fugitive emissions of CH_4 from AD plant; ρ_{CH4} is the density of CH_4 in standard temperature (6.67×10^{-4} t/m^3); *FC* and *EC* is the provision of diesel and the electricity consumption for the operation process which are taken as 1.6 L/t input waste and 20 kWh/t input waste, respectively (Møller, Boldrin, and Christensen 2009). The produced biosolids or sludge are sent to landfills and are assumed to be equivalent to 50% of input waste (Chen and Lo 2016).

4.2.10. Emissions from landfill

The CH_4 emissions from the landfill are calculated following the IPCC FOD model. For the FOD model, the gradual decay of degradable organic carbon of MSW occurs over several years. However, for the purpose of this study, only the emission of CH_4 from a fixed amount of MSW disposal was considered. The impact of time and carbon bound in the landfill were not taken into account. The landfill model is described as follows (Xin et al. 2020):

$$E_{CH4} = DDOC \times F \times 16/12 \times (1 - R) \times (1 - OF) \quad (40)$$

$$DDOC = MSW \times DOC \times DOC_f \times MCF \quad (41)$$

$$DOC = \sum DOC_i \times W_f \quad (42)$$

Where, E_{CH4} is the total CH_4 emissions; *DDOC* is the mass of decomposable DOC; *DOC* is the fraction of degradable organic carbon; DOC_i is degradable organic carbon for waste fraction *i*; W_f is waste composition; DOC_f is the fraction of DOC that can decompose, which the default value is 0.5; *MCF* is the methane correction factor for aerobic decomposition, which is 0.8 for unmanaged deep landfill (IPCC 2006); *F* is the fraction of methane in generated landfill gas, which is 0.5; *OF* is oxidation factor, which is 0; and *R* is methane recovered gas. In this study, methane is not recovered, so *R* is set to 0.

In the operation process, only emissions from heavy-duty equipment using diesel are considered. The electricity required for running office buildings at the landfill site is negligible and therefore not included in the study. The daily diesel consumption is 1,435 L, which amounts to approximately 0.43 L/tMSW (Dangkor landfill Authority 2021).

4.3. Results and discussion

4.3.1. Emission factors for MSW transportation

Door-to-door collection is the most common method for collecting MSW in Phnom Penh. However, this method requires more workers and takes more time due to frequent stops. The vehicles used for waste transportation run solely on diesel fuel, and it is assumed that the travel distance to all treatment facilities is 18 km. Diesel combustion can emit CO_2, CH_4, and N_2O. According to Outapa and Na Roi-et (2018), the diesel fuel used for quantifying GHG emissions from waste transportation is 0.1858 L/km/tMSW. The estimated GHG emissions from the transport of waste were on average of 9 $kgCO_2$-eq/tMSW, which is lower than that found in Thailand, 26.33 $kgCO_2$-eq/tMSW (Menikpura, Sang-arun, and Bengtsson 2012), but higher than that in Indonesia, 1.38 $kgCO_2$/tMSW (Kristanto and Koven 2020) (Table 22). The difference in GHG emissions from waste transportation is mainly due to the travel distance and collection method.

Table 22. Emission factors for different MSM treatments ($kgCO_2$-eq/tMSW)

Type of treatment	Gross emissions	Emissions avoidance	Net emissions
Transportation	9	-	9
Open burning	501	-	501
Composting	195	-	195
Recycling	689	2,838	–2,149
Anaerobic digestion	99	145	–46
Incineration	399–553	290–306	109–247
Landfill	1,056	-	1,056

4.3.2. Emission factor for composting and anaerobic digestion

The GHG emissions estimation from the composting facility is determined considering only organic waste in scenarios 4 and 5, with an equal amount (472 tMSW/day). The emission factors for waste composting were 195 $kgCO_2$-eq (Table 22), which falls within the range of value reported in other studies (Andersen et al. 2010; Friedrich and Trois 2013b; Kristanto and Koven 2020; Xin

et al. 2020). The GHG emissions from composting were six times lower than landfilling. Therefore, recycling organic waste through composting has significantly reduced GHG emissions.

Another method in the treatment of organic waste is AD. Unlike composting, the AD plant could potentially generate electricity, which can be used to offset the electricity generated from conventional energy sources, such as coal and diesel fuel. According to IGES (2022), the emission factor for national grid in Cambodia was 0.586 $kgCO_2$-eq/kWh. In this study, electrical energy from biogas was estimated at 247 kWh/t food waste, avoiding approximately 145 $kgCO_2$-eq/t food waste. However, during the AD operation, there is an unavoidable leakage of biogas and emissions of GHGs due to electricity and fuel consumption. By subtracting these emissions, the net GHG emissions resulted in a negative emission, saving approximately –46 $kgCO_2$-eq/t of food waste. The generated electricity is lower than that in Dhanbad city of India, which is 404 kWh/t food waste (Kumar and Samadder 2022), and in Korea, which is 443 kWh/t food waste (Yi, Jang, and An 2018) but higher than in Taiwan, which is 154 kWh/t food waste (Chen and Liu 2021).

4.3.3. Emission factors for recycling

Recycling prevents GHG emissions from material production, as recycled products can be used to substitute raw materials. Therefore, this study considered the GHG emissions and energy reduction resulting from using recycled materials instead of virgin materials. The net GHG emissions from materials recycling were approximately –2,149 $kgCO_2$-eq/t mixed recyclables (Table 22). Recycling aluminum and ferrous materials bring the greatest net GHG emissions reductions, with values of –1,144 and –858 $kgCO_2$-eq/t, respectively (Figure 32).

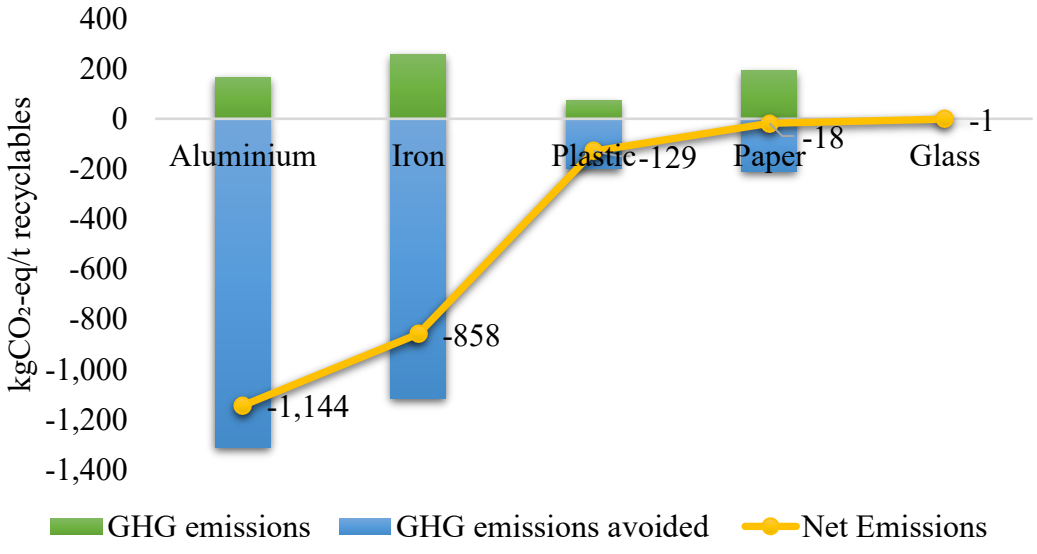


Figure 32. GHG saving from recycling one ton of mixed recyclables

The significant net GHG savings are mainly attributed to the avoidance of emissions from the complicated process of extracting raw materials from nature to produce final products (Turner et al. 2015). Additionally, the processes for aluminum and ferrous materials are much simpler compared to other materials. Recycling exhibits colossal potential for avoiding GHG emissions, as found in studies of GHG emissions factors for recycling in South Africa (Friedrich and Trois 2013b) and the UK (Turner et al. 2015).

4.3.4. Emission factors for open burning and incineration

Both open burning and incineration exhibited gross GHG emissions higher than 500 $kgCO_2$-eq/tMSW. Open burning involves the burning of materials without energy recovery or air pollution control, resulting in no emissions reductions. On the other hand, incineration has the potential to produce electricity, leading to significant reductions in GHG emissions.

In scenario 2, the electricity generation from WTE incineration of mixed waste was 518 kWh/tMSW. Under scenarios 3, 4, and 5, the electrical energy decreased to 512, 502, and 483 kWh/tMSW, respectively, corresponding to an increase in separation rates of 25, 50, and 75%, respectively. The separation of food waste can reduce the moisture content and increase net calorific value (Leckner 2015). However, when food waste is separated along with recyclables, particularly paper and plastic, the energy potential of an incinerator decreases (Horttanainen et al. 2013).

According to Table 23, incineration of mixed waste emitted the highest amount of GHGs. Sorting food waste and recyclables can reduce GHG emissions to 512, 502, and 481 $kgCO_2$-eq/tMSW in response to separation efficiencies of 25, 50, and 75%, respectively. Plastic waste incineration emitted the most GHGs, followed by textile and paper. Other studies also found that incinerating plastic waste emitted the highest GHG emissions (Chen 2018; Monni 2012; Yaman 2020).

Table 23. GHG emissions and energy recovery from incineration

Composition	GHG emissions (CO_2-eq/tMSW)				Energy (kWh/tMSW)			
	Mixed waste	**25% separation**	**50% separation**	**75% separation**	**Mixed waste**	**25% separation**	**50% separation**	**75% separation**
Food waste	7.40	6.92	6.13	4.57	9.02	8.43	7.47	5.56
Wood and leaves	1.01	1.26	1.67	2.48	2.08	2.60	3.45	5.14
Paper	1.32	1.23	1.09	0.81	14.68	13.73	12.16	9.05

Plastics	487.67	456.11	403.97	300.81	408.28	381.85	338.20	251.84
Rubber and leather	2.89	3.68	5.07	8.13	10.81	13.77	18.96	30.45
Textile	16.90	21.07	27.99	41.69	66.25	82.62	109.76	163.47
Nappies	3.42	4.27	5.67	8.44	7.26	9.05	12.03	17.91
Emissions from electricity and diesel consumption: 32.01 CO_2-eq/tMSW								
Total	**552.62**	**526.55**	**483.60**	**398.95**	**518.38**	**512.06**	**502.03**	**483.41**

The data normalization method was employed to assess the environmental performance of the incineration technology. This method standardized the data within the range 0 and ±1. Outliers were removed, and all variables were adjusted proportionally using the following equation (Chen 2018).

$$X_i = \frac{X_i - X_{min}}{X_{max} - X_{min}}$$

In the given equation, X_i represents each data item (such as energy and GHG), while X_{max} and X_{min} represent the maximum and minimum values within the dataset. In the context of incineration technology, energy recovery is considered a positive environmental factor, while GHG emissions are seen as a negative factor. The environmental performance of incinerating various types of waste was assessed using the following equation.

$$EP_{inc} = X_{energy} + X_{GHG}$$

Where, EP_{inc} is the environmental performance of the incineration, X_{energy} denotes the grade of energy recover and X_{GHG} represents the grade of GHG emissions from an incineration plant.

According to Figure 33, incineration of plastic resulted in the highest electricity generation. Ohnishi et al. (2018) suggested that waste-rich in plastic could serve as an efficient fuel for WTE plants. However, it was also found to have the poorest environmental performance, as confirmed by a study conducted by Chen (2018). Incineration of textile and food waste ranked second and third, respectively, in terms of GHG emissions. However, it is worth noting that the energy recovery from incinerating textile waste outweighed its emissions.

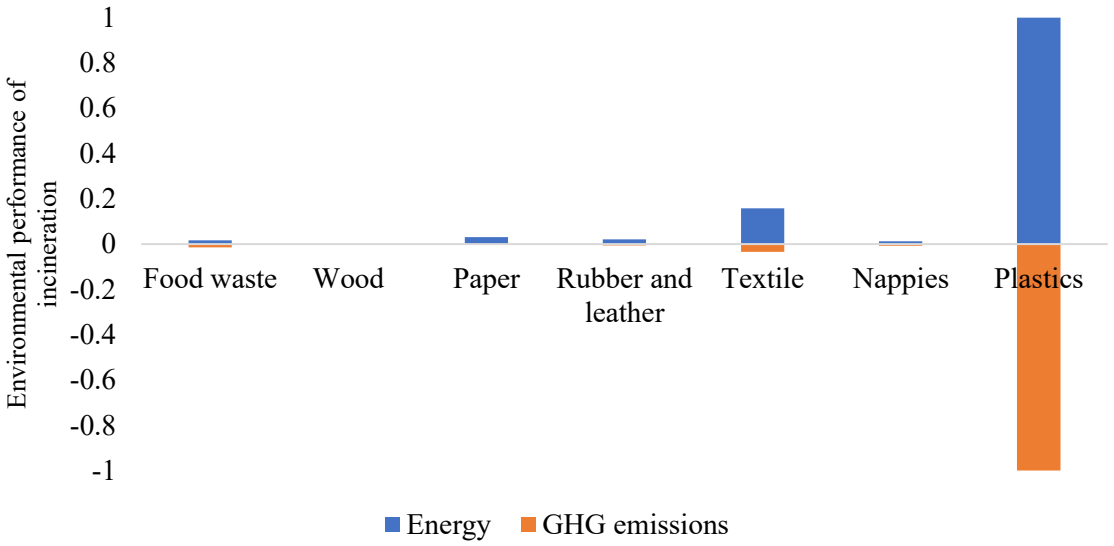


Figure 33. Normalized environmental performance of incineration

The sensitivity analysis conducted to evaluate the effects of plastics separation on the performance of incineration in GHG mitigation revealed significant impacts. The composition of plastics was varied by decreasing its rate by 10% increments. The results showed that, on average, GHG emissions and ERP were reduced by 46.62 $kgCO_2$-eq/tMSW and 36.77 kWh/tMSW, respectively, for every 10% reduction in plastics (Figure 34). This finding highlights the influence of plastics on GHG emissions and the energy potential of WTE incineration. Therefore, separating plastics from other waste components is essential to enhance the potential for GHG mitigation. However, it is important to note that increasing plastic separation significantly impacted electricity production, which may alter the economic feasibility of the incineration plant.

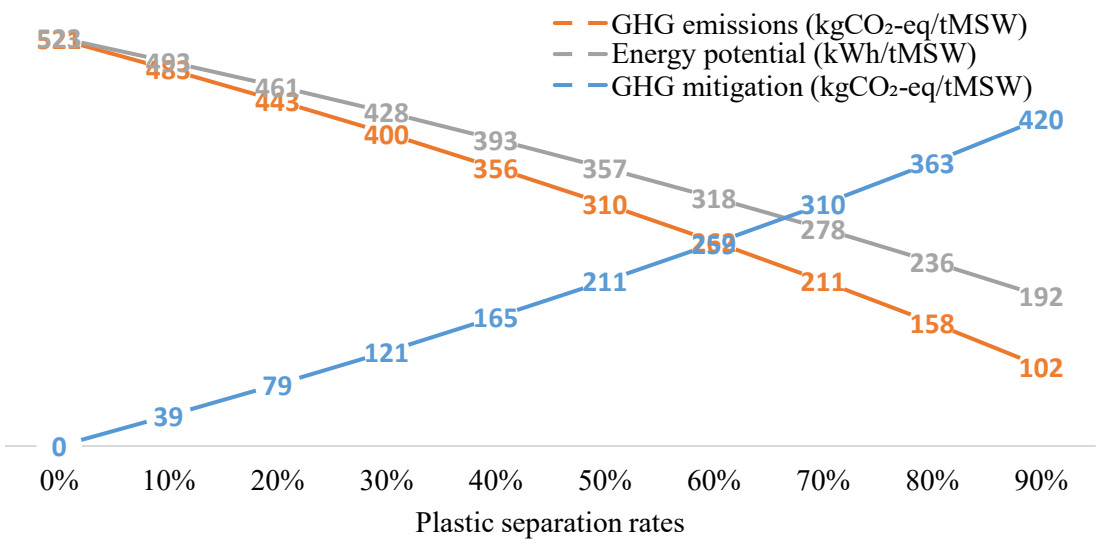


Figure 34. Sensitivity analysis of plastic separation rates of incineration performance

4.3.5. Emission factors for landfill

The MSW landfill operations in Phnom Penh municipality, similar to those in other developing countries, do not employ gas extraction systems (IPCC 2007; Tan, Hashim, et al. 2014). This study evaluated the environmental pollution of MSW landfill based on a fixed amount of MSW disposal, resulting in approximately 1,056 $kgCO_2$-eq/tMSW of GHG emissions. This value is comparable with findings from other studies. Chen and Liu (2021) reported a value of 1,807 $kgCO_2$-eq/tMSW, while Xin et al. (2020) documented a range of GHG emissions from landfills between 1,225 $kgCO_2$-eq/tMSW and 1,280 $kgCO_2$-eq/tMSW. Sarbassov et al. (2020) determined that landfills produced 1,055 $kgCO_2$-eq/tMSW of GHGs. Emissions of LFG vary based on factors such as waste composition, moisture content, and landfill management. Notably, food waste has a significant potential for GHG emissions when disposed of in landfills, as highlighted by Chen and Liu (2021). Furthermore, organic materials take nearly a century to decompose in landfills. Therefore, AD and composting are recognized as the most effective alternatives for treating organic waste (Kristanto and Koven 2020; Kumar and Samadder 2017).

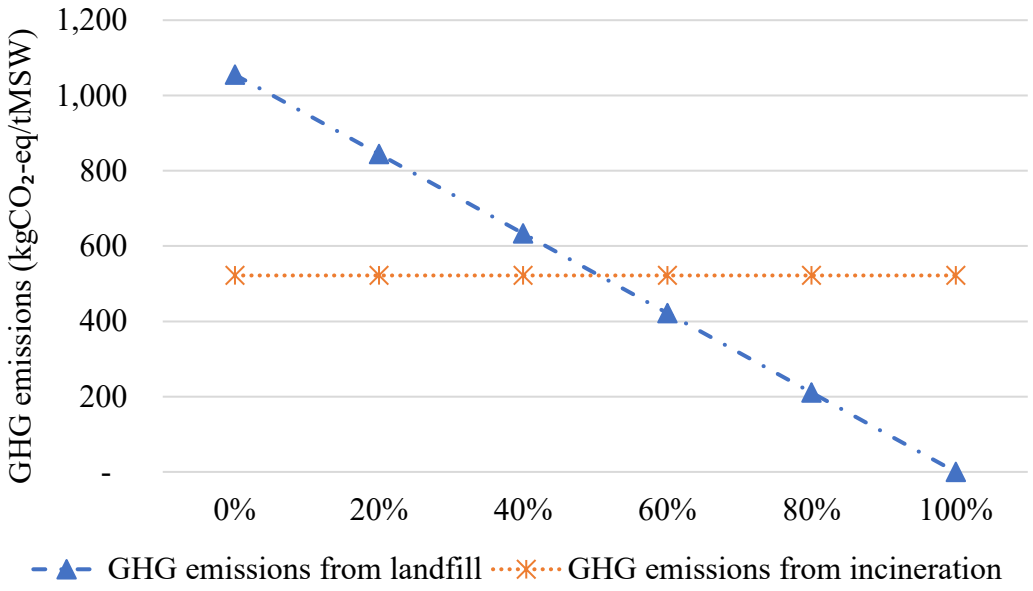


Figure 35. Variation of landfill gas collection for GHG mitigation

The release of CH_4 from landfills gives rise to environmental impacts due to its global warming impact, which is approximately 25 times higher than that of CO_2. In Denmark, regulations have been implemented to manage LFG through methods such as CH_4 collection and utilization, flaring, or other approaches (Scheutz and Kjeldsen 2019). Efficient utilization of CH_4 gas for electricity generation can offer a sustainable option for both economic and environmental benefits in MSWM (Ghosh et al. 2019). However, the landfill in Phnom Penh is not equipped with a CH_4 recovery

facility. Capturing CH_4 gas for electricity can substantially mitigate GHG emissions. In this study, GHG emissions from the landfill are almost 2-fold compared to those from incineration. To achieve a similar level of GHG emissions as incineration, the efficiency of LFG collection should be around 50% (Figure 35). This finding aligns with the observations made in China (Xin et al. 2020).

4.3.6. Comparison of the overall GHG emissions between scenarios

According to Figure 36, the gross GHG emissions varied across the five scenarios, with scenario 1 having the highest emissions and scenario 5 having the lowest. In scenario 1, where no treatment technologies were implemented, the gross GHG emissions reached 3.86 M $kgCO_2$-eq/day. On the other hand, in scenario 5, the emissions were significantly lower at 1.40 M $kgCO_2$-eq/day. When comparing scenario 2 to scenario 1, a reduction of 0.74 M $kgCO_2$-eq/day in gross GHG emissions was observed. Scenario 3 involved the elimination of landfilling and the implementation of recycling, AD, and incineration. This led to a significant decrease in gross GHG emissions, with reductions of 1.95 compared to scenario 1, and 1.22 M $kgCO_2$-eq/day compared to scenario 2. The trend of decreasing emissions continued in scenario 4, with a reduction of 2.16 M $kgCO_2$-eq/day, and scenario 5 with a reduction of 2.42 M $kgCO_2$-eq/day.

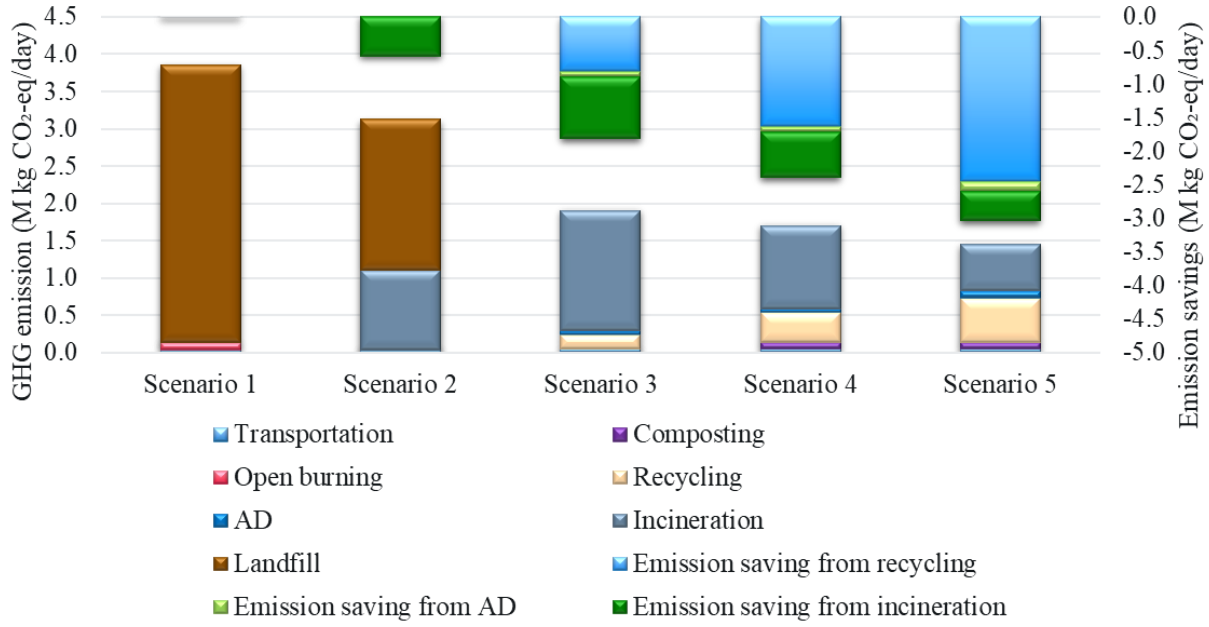


Figure 36. Gross GHG emissions among different scenarios

Among the five scenarios, transportation was found to contribute the least emissions compared to the treatment technologies. In scenario 1, the GHG emissions from transportation amounted to 33,465 $kgCO_2$-eq/day, which was lower than the emissions in the other four scenarios. The higher

GHG emissions observed in scenarios 2 to 5 were primarily attributed to the increased in waste collection rate and transportation of rejects from treatment facilities to the landfill. To address this issue, Kristanto and Koven (2020) suggested consolidating waste treatment operations in a single area to minimize transportation emissions. By centralizing waste treatments, the need for extensive transportation can be reduced, leading to lower overall GHG emissions.

Figure 37 illustrates the net GHG emissions, which represents the emission value obtained after subtracting the emissions avoided from electricity generation and recycling. Recycling helps reduce the use of raw materials and saves GHG emissions from production processes, while AD and incineration generate electricity, resulting in savings in emissions from other sources. Among the five scenarios, scenario 5 achieved the highest GHG emissions savings, while scenario 2 had the lowest. Scenario 1, which involved open burning and landfilling, there were no GHG emissions reductions. Specifically, open burning in scenario 1 emitted 0.10 M $kgCO_2$-eq/day of GHGs from the combustion of 200 tMSW, landfilling contributed 3.73 M $kgCO_2$-eq/day, and transportation accounted for 0.034 M $kgCO_2$-eq/day. In scenario 2, approximately 0.59 M $kgCO_2$-eq of GHGs were avoided through the substitution of electricity generation from WTE incineration. Scenario 3 demonstrated a significant reduction in net GHG emissions compared to scenario 1, with approximately 1.81 M $kgCO_2$-eq/day mitigated through recycling, AD, and incineration plants. Both scenarios 4 and 5 resulted in negative net GHG emissions, indicating emissions savings. Scenario 4 achieved a net emissions savings of approximately –0.60 M $kgCO_2$-eq/day, while scenario 5 achieved a higher net emissions savings of approximately –1.59 M $kgCO_2$-eq/day.

Scenarios 4 and 5 demonstrated significant GHG emissions savings primarily due to waste recycling. The Waste Management Strategy and Action Plan for Phnom Penh 2018-2035 has set targets of achieving recycling rates of at least 50% by 2023 and 95% by 2035. These targets have the potential to avoid approximately –1.19 M $kgCO_2$-eq/day of GHG emissions by 2023 and –3.55 M $kgCO_2$-eq/day by 2035. However, challenges exist in achieving these targets, including low public participation in source segregation and the absence of MRF plant. These challenges have been observed in other countries such as Indonesia (Kristanto and Koven 2020). The absence of an MRF poses a significant barrier to achieving recycling goals. An MRF can recover up to 83% of recyclables from the input waste (Kristanto, Gusniani, and Ratna 2015).

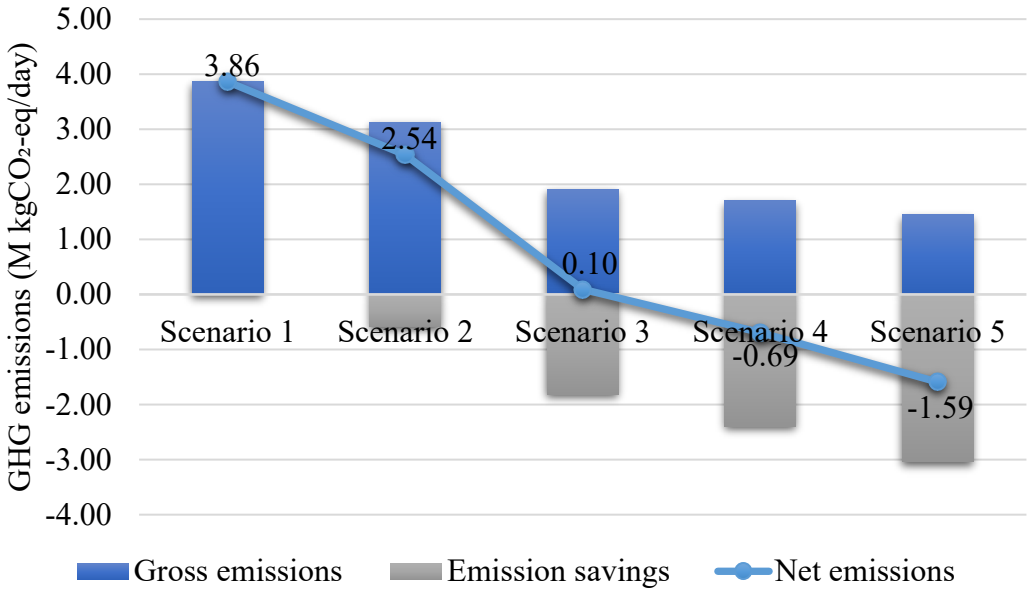


Figure 37. Net GHG emissions among scenarios

In scenarios 4 and 5, about 18% of GHGs were also reduced by converting 472 tons of organic waste into compost instead of sending it to the landfill. The use of compost in agricultural activities instead of chemical fertilizers helps offset GHG emissions associated with fertilizer production. However, this GHG emissions avoidance process is not considered in this study. In scenario 5, an additional environmental benefit was achieved by generating approximately 233 MWh of electricity through the AD of 943 tMSW (50% of the organic waste). This resulted in a reduction of approximately –0.14 M $kgCO_2$-eq/day. If all food waste in Phnom Penh were used for AD, it is estimated that about 466 MWh of electricity could be produced daily. This would be sufficient to supply approximately 134,481 households, assuming an average monthly consumption of 104 kWh per household (Mika et al. 2021).

The net GHG emissions saving in scenario 5 was twice as high as in scenario 4, primarily due to different separation rates, as shown in Figure 37. Therefore, scenario 5 represents a new paradigm for MSW management in Phnom Penh, emphasizing the optimal use of resources from waste and effective mitigation of GHG emissions from the sector.

In scenarios 2, 3, 4, and 5, WTE incineration operations resulted in gross GHG emissions of approximately 1.06, 1.62, 1.12, and 0.62 M $kgCO_2$-eq/day, respectively. However, it is important to consider the positive impact of incineration on reducing GHG emissions through electricity substitution. The net GHG emissions from WTE incineration in these scenarios were approximately 0.48, 0.70, 0.44, and 0.18 M $kgCO_2$-eq/day, respectively. These values take into

account the emissions saved from the electricity generated by incineration, which offsets emissions that would have been produced by conventional energy sources in the national grid. The net emissions represented the actual contribution of WTE incineration to the overall GHG emissions reduction in each scenario, were depicted in Figure 38.

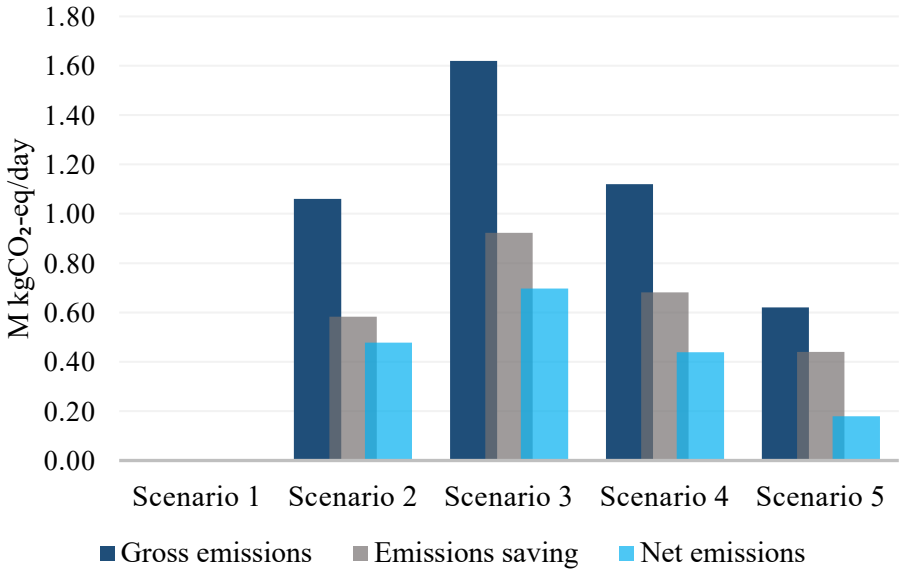


Figure 38. GHG emissions from incineration under different scenarios

4.4. Uncertainties and limitations

Due to the unavailability of specific data on waste recycling efficiency and biogas production in Phnom Penh, certain assumptions were made to estimate the environmental benefits and electrical energy potential in this study. To calculate the GHG emissions from recycling facilities and the avoided emissions from material production, data from another country, as documented in the study by Turner et al. (2015), was utilized. Similarly, for estimating the amount of electricity generation from the AD plant, the biogas yield data from experimental research conducted by Al-Wahaibi et al. (2020) was adopted. It is important to note that these assumptions were made based on available data and may not perfectly align with the waste characteristics and technological applications specific to Phnom Penh. If country-specific data were available, the results might differ accordingly.

4.5. Conclusions

This study focused on assessing the GHG emissions associated with MSW management in Phnom Penh municipality. It specifically examined the direct emissions from treatment and operating processes, as well as avoided emissions from substituting raw materials and electricity products.

The study incorporated GHG emissions from waste transportation to various treatment facilities under five scenarios. However, there were certain aspects that the study did not consider, such as the avoided emissions from using compost instead of chemical fertilizer and the potential for landfill gas to be converted into energy. The results of the study demonstrated the potential for significant negative net GHG emissions in scenarios 4 and 5, where food waste and recyclables were separated at rates of 50% and 75% respectively. In scenario 3, where the separation rate of 25%, the net GHG emissions were 0.10 M $kgCO_2$-eq/day, lower than those in scenarios 1 and 2, where waste separation was not implemented. GHG emissions from waste transportation were the lowest, while landfilling contributed the most GHG emissions, with emissions exceeding 1,000 $kgCO_2$-eq/tMSW. Open burning and composting were the second and third highest sources of emissions, respectively. Incineration emitted fewer GHGs, thanks to the electricity generated during the process, which helped offset other emissions. Recycling and AD potentially showed potential for reducing the overall volume of MSW and resulted in negative net GHG emissions.

The study highlights the importance of waste separation in mitigating GHG emissions. Incineration and AD technologies have the potential to produce electricity, which can substitute conventional energy sources and further reduce GHG emissions. Future studies could focus on assessing the economic feasibility of different WTE energy technologies and comparing their environmental performance.

CHAPTER 5

ENERGY, ECONOMIC, AND ENVIRONMENTAL BENEFITS OF WASTE-TO-ENERGY

5.1. Introduction

In a global context, the generation of MSW has experienced a substantial increase, rising from 1.3 billion tons in 2012 (Hoornweg and Bhada-Tata 2012) to 2.7 billion tons in 2019 (Maalouf and Mavropoulos 2022). This escalation is accompanied by an average generation rate of 0.74 kg/capita/day (Kaza et al. 2018). The projections indicate that by 2050, MSW generation will rise to 3.40 billion tons, wherein low- and middle-income countries are anticipated to contribute over 40% of the total volume (Kaza et al. 2018). This noteworthy trend in MSW generation can be predominantly attributed to various factors, such as population growth, economic development, urbanization, industrialization, and changes in consumption habits (Trindade et al. 2018). Unfortunately, the lack of approximate treatment measures is causing a considerable environmental concern. Approximately 75% of the world's MSW is directly disposed of in landfills and dumping sites without undergoing any treatment processes (Hadidi and Omer 2017). Consequently, this disposal practice leads to the generation of LFG through the biodegradation process, significantly impacting the environment and contributing to climate change.

Fossil fuel-based electricity generation significantly contributes to GHG emissions throughout its entire life cycle, spanning from resource extraction to final consumption. Consequently, there is a growing global emphasis on transitioning from conventional electricity generation sources to greener and more sustainable alternatives. Waste-derived electric energy has emerged as a viable solution to alleviate the waste burden, generate electricity, and mitigate GHG emissions (Ayodele, Ogunjuyigbe, and Alao 2018; Brunner and Rechberger 2015; Cucchiella, D'Adamo, and Gastaldi 2014). A range of WTE technologies, including thermal treatment methods such as incineration, gasification, pyrolysis, and plasma arc, as well as biological treatment methods like AD and LFG recovery, have been developed to covert waste into usable energy (Gómez et al. 2010). By employing these technologies, waste can be effectively utilized as a resource for energy generation.

As of 2018, more than 2,450 WTE plants were operational worldwide, collectively processing approximately 368 M tMSW annually (Asian Development Bank 2020). These plants play a crucial role in diverting waste from landfills and contributing to the production of renewable

energy. By harnessing the energy potential of waste, these facilities not only address the waste management challenge but also help reduce GHG emissions associated with traditional electricity generation.

LFG mainly consists of CH_4 in the range of 50–60% (IPCC 2006). This CH_4 content makes LFG a valuable renewable energy source due to its high calorific value of 37.2 MJ/m^3 (Ayodele et al. 2017). Numerous studies have utilized mathematical models to assess landfill CH_4 generation and its economic potential. For instance, Escamilla-García et al. (2020) employed the LandGEM model to evaluate LFG generation, economic feasibility, and environmental benefits at a landfill site in southern Mexico. Their findings indicated LFG with a CH_4 generation flow rate of 115 m^3/min, capable of producing approximately 32.40 GWh/year of electricity and 63.99 BTU/year of steam. The economic analysis revealed financial profitability with a positive net present value (NPV) over the 15-year lifespan of the project. Similarly, a study by Kumar and Sharma (2014a) focused on the energy recovery potential (ERP) from three landfill sites in India. They also demonstrated a positive NPV, considering a discount rate of 10%. The levelized cost of electricity (LCOE) ranged from 0.12 to 0.17 USD/kWh, which was lower compared to solar power plants and offshore wind energy plants. In another study conducted by Cudjoe, Han, and Chen (2021), the authors estimated the ERP from LFG in three regions of China. Using the LandGEM model and historical landfill data spanning 15 years, they estimated that landfill sites in these regions could generate approximately 12,525 GWh of electricity. The economic assessment conducted also indicated a positive cash flow.

MSW typically consists of various components, including organic fraction, recyclable materials, and non-recyclables. Different technologies, such as incineration and AD, have been employed as alternatives to landfilling in managing these waste streams. Incineration is a process that involves burning waste at a temperature of at least 900 °C (Silva et al. 2020). This thermal treatment method not only generates energy through a steam turbine but also helps destroy hazardous organic substances and minimizes the release of toxic metals via filter (Brunner and Rechberger 2015). Incineration is particularly suitable for non-recyclable waste and can significantly reduce the volume and mass of the waste being discarded. By converting waste into energy, incineration can contribute to a circular economy by minimizing the dependence on natural resources and reducing the environmental and health risks associated with landfilling.

On the other hand, AD is a biological treatment process that specifically targets biodegradable organic waste. This waste is broken down by microorganisms in an oxygen-free environment,

producing biogas as an output. Biogas, primarily composed of CH_4, can be utilized as a renewable energy source. AD not only helps divert organic waste from landfills but also provides an opportunity to generate energy from it. Both incineration and AD technologies offer potential solutions to reduce landfilling and promote a circular economy by effectively utilizing organic waste and non-recyclable materials for energy generation. By converting waste into electricity or biogas, these technologies help minimize the negative impacts of waste disposal on the environment and human health while simultaneously reducing the demand for virgin resources.

In a circular economy, the aim is to minimize waste generation and maximize the value of resources through closed-loop systems where materials are continuously reused, recycled, and recovered. However, it is important to note that the increasing focus on waste recycling may affect the availability of feedstock for WTE incineration. In particular, in developing countries like Cambodia, the lack of source segregation and the disposal of waste in a comingled manner pose challenges for the separation and sorting of recyclable materials. As a result, the recovered materials may be of low quality, damaged or contaminated, such as wet paper and mixed and soiled plastics. The contamination makes these recyclables unappealing or difficult to process for recycling facilities, necessitating their disposal as trash. Additionally, some contaminated waste requires advanced recycling technologies to separate harmful compounds effectively. Despite these challenges, incineration remains an effective method for reducing the mass and volume of waste, especially when dealing with complex streams containing hazardous organic compounds. Proper separation of MSW, particularly segregating food waste, can enhance the efficiency and effectiveness of an incineration facility.

However, incineration has the potential to release various pollutants into the air, including CO_2, N_2O, SO_2, and particular matter. These pollutants can contribute to both local air pollution and global warming, thereby affecting climate change. Additionally, incineration can emit toxic substances such as dioxins and furans (Brunner and Rechberger 2015; Trindade et al. 2018), which are highly hazardous to human health (Chakraborty et al. 2013). Consequently, concerns regarding pollution, health risks, and potential negative impacts on local communities have led to opposition to incineration facilities (Dek et al. 2022).

Developed countries have been adopting alternative thermal technologies, such as pyrolysis and gasification, which utilize more efficient equipment like a combined cycle gas turbine (Lombardi, Carnevale, and Corti 2015). These technologies are considered promising, but they come with higher capital and operating costs and are more technologically complex (Silva et al. 2020). One

notable challenge with these technologies is their limited range of suitable feedstocks, which typically include solid refuse fuels (SRF), plastics, and rubber tires (Lombardi et al. 2015).

Cambodia has experienced a significant increase in MSW generation over the last decade, primarily driven by factors such as economic growth, population growth, improved living standards, and rapid urbanization (Dek et al. 2022). In 2020, the country generated approximately 4.78 M tons, with an average per capita generation rate of 0.78 kg/day (Dek et al. 2022). The current MSW management practice in Cambodia predominantly relies on landfilling and burial at disposal sites, which poses environmental risks due to GHG emissions and leachate generation. This presents a challenge for the government in selecting an effective alternative MSW management system. To address these issues and minimize environmental impacts, the government of Cambodia is considering WTE as an alternative approach. Currently, the country's electricity production mainly depends on hydropower (51.93%) and fossil-fired power plants (41.28%), including those fueled by coal, diesel, heavy fuel oil, and light diesel oil (Electricity Authority of Cambodia 2021). Additionally, a portion of electricity is imported from neighboring countries such as Thailand, Vietnam, and Lao PDR, accounting for 26% of the supply (Electricity Authority of Cambodia 2021). However, the current electricity supply is inadequate to meet the growing consumer demand, particularly during the dry season when the water resource for hydropower plants is diminished. Introducing WTE technologies could help bridge the gaps in electricity supply and mitigate GHG emissions from the waste sector. Therefore, conducting studies on the energy recovery potential of MSW is crucial. Such studies can serve as a valuable reference for decision-making and the development of an effective strategy to address Cambodia’s MSW management challenges while simultaneously contributing to the country’s energy needs and reducing GHG emissions.

The objective of this study was to 1) assess the energy recovery potential, 2) analyze the economic feasibility, and 3) evaluate the environmental performance, considering the global warming potential (GWP) from the three WTE technologies: incineration, anaerobic digestion, and LFG recovery. The study focused on Phnom Penh, the most populated city in Cambodia. The conceptual framework outlining the scope and approach of the study is presented in Figure 39.

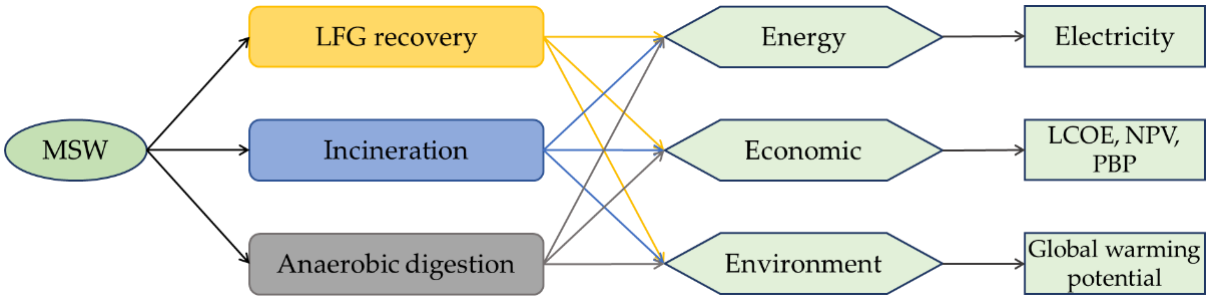


Figure 39. Assessment framework for WTE technologies

5.2. Methodologies

5.2.1. Status of MSWM in Phnom Penh municipality

Phnom Penh, the capital city of Cambodia, is characterized by a population of approximately 2,281,951 and covers a land area of 679 km^2, resulting in a high population density of 3,361 people per square kilometer (National Institute of Statistics 2020). This makes it the most densely city in the country. The city has experienced significant population growth, with an annual growth rate of 4.9% between 2008 and 2019, compared to 2.8% between 1998 and 2008 (National Institute of Statistics 2020). As a result, the increasing population, combined with limited land areas, will present significant challenges in managing MSW in the city.

In 2022, approximately 1.29 M tMSW were collected and disposed of at a landfill site in Phnom Penh without undergoing intermediate treatment. The existing landfill operates without systems for LFG collection and leachate treatment.

5.2.2. Waste generation and characteristics

In this study, the ERP from MSW was evaluated over a 20-year period from 2023 to 2042. The analysis took into account the waste composition, as detailed in Table 24. The study also considered a correlation between population and MSW generation, assuming a constant growth rate for projecting MSW generation within the specified timeframe. The projection of MSW generation potential is given in Equations (43) to (45).

$$MSW_{gen(t)} = P_{(t)} \times W_{Gr} \times 365/1000 \quad (43)$$

$$P_{(t)} = P_{(0)} \times (1 + r)^t \quad (44)$$

$$W_{Gr} = \frac{W_{collected}}{P_{(b)} \times R_{collection}} \times \frac{1000}{365} \quad (45)$$

Where, $MSW_{gen(t)}$ is the forecasted waste generation in year t; $P_{(t)}$ is the projected population over the years t, using geometrical increase method; W_{Gr} is the MSW generation per capita; $P_{(0)}$ is the initial population using the national census 2019; r is the population growth rate; t is the number of years; $W_{collected}$ is the quantity of waste collected in 2022, which is taken as 1,288,223 tons; $P_{(b)}$ is the population in the base years of calculation; and $R_{collection}$ is the collection efficiency. This study used the average population growth rate in the last two decades (1998-2019) (National Institute of Statistics 2020).

Table 24. MSW characteristics in Phnom Penh

Composition	Waste properties						Waste treatment		
	Fraction (%) [a]	Moisture (%)[a]	LHV (MJ/kg)[a]	Carbon content (%)[b]	Fossil carbon (%)[b]	DOC (%)[b]	LFG (%)	Incineration (%)	AD (%)
Food waste	49.18	78.77	0.33	38.00	-	15	49.18	49.18	49.18
Wood and leaves	6.69	57.12	0.56	49.00	-	43	6.69	6.69	-
Mixed paper	6.54	63.61	4.04	46.00	1.00	40	6.54	6.54	-
Rubber and leather	0.87	18.09	22.37	67.00	20.00	39	0.87	0.87	-
Textile	8.02	44.28	14.87	50.00	20.00	24	8.02	8.02	-
Nappies	2.91	58.29	4.49	70.00	10.00	24	2.91	2.91	-
Plastic	21.13	18.37	34.78	75.00	100.00	-	-	21.13	-
Glass	1.42	-	-	3.00	50.00	-	-	-	-
Metals	1.05	-	-	3.00	50.00	-	-	-	-
Others	2.21	22.73	3.84	3.00	50.00	-	-	-	-

[a] Seng et al. (2018), [b]IPCC (2006)

5.2.3. Estimation of energy recovery potential

5.2.3.1. Energy generation from LFG

The LandGEM model (version 3.02) was utilized to estimate the generation of LFG. The details of the LandGEM model used in this study are present in section 3.3.3.

In general, it is not possible to achieve 100% efficiency in collecting landfill CH_4 due to factors such as gas leakage from the collection system, bio-oxidation with covered soil, and improper

capping (Barlaz et al. 2009). According to Amini et al. (2012), the average LFG efficiency of LFG collection ranges from 67% to 90%. In this study, a collection efficiency of 75% is assumed, based on the work of Cudjoe et al. (2021). The ERP for LFG recovery can be calculated using the methodology proposed by Ayodele et al. (2018):

$$ERP_{LFG} = (Q_{CH_4(LFG)} \times (1 - OF) \times LHV \times \eta \times \lambda \times CF)/3.6 \quad (46)$$

Where, *OF* is the oxidation factor in a landfill; *LHV* is the lower heating value of CH_4; η is the electricity conversation efficiency for internal combustion; λ is the collection efficiency of methane from landfill; *CF* is the capacity factor of the plant over the year operation (Table 25); and 3.6 is the conversion factors kJ to kWh.

Table 25. Parameters for calculating energy recovery potential from the three technologies

Plant type	OF (%)	LHV (MJ/m^3)	ŋ (%)	λ (%)	CF (%)
LFG	10[a]	37.2[b]	30[c]	75[d]	85[e]
AD	-	37.2[b]	30[c]	95[a]	85[e]
Incineration	-	(see Table 24)	25[c]	-	80[c]

[a]IPCC (2006), [b]Ogunjuyigbe, Ayodele, and Alao (2017), [f]Chakraborty et al. (2013), [d]Cudjoe et al. (2021), [e]Hadidi and Omer (2017)

5.2.3.2. Energy generation from incineration

The data in Table 24 and Table 25 are used for estimating ERP from incineration using moving grate system, following Equation (47) (Chakraborty et al. 2013).

$$ERP_{inc} = (MSW_i \times LHV_i \times \eta \times CF)/3.6 \quad (47)$$

Where, ERP_{inc} is the energy recovery potential from waste incineration; LHV_i is the lower heating value of waste fraction *i*.

5.2.3.3. Energy generation from AD

Food waste is recognized as a viable feedstock for AD plants to produce electricity. Based on the experiences conducted by Al-Wahaibi et al. (2020), the biogas yield from mixed food waste was determined to be 1,550 L/kg, with a CH_4 content of 30%. Therefore, the quantity of CH_4 that can be obtained from the AD plant can be calculated as follows:

$$Q_{CH_4(AD)} = (M_{food\ waste} \times dm \times Yield_{biogas} \times F_{CH_4})/1000 \quad (48)$$

Where, $Q_{CH4(AD)}$ is the methane generation from AD; $M_{food\ waste}$ is the mass of input waste; *dm* is the dry matter content of food waste, taken as 21.23% (Seng et al. 2018); $Yield_{biogas}$ is biogas yield; and F_{CH4} is the methane content in biogas. The ERP from AD technology and the plant capacity can be calculated below (Ayodele et al. 2018):

$$ERP_{AD} = (Q_{CH_4(AD)} \times LHV \times \eta \times \lambda \times CF)/3.6 \quad (49)$$

Where, ERP_{AD} is the energy recovery potential from AD technology, the value of *LHV* of CH_4, η, λ, and *CF* is shown in Table 25.

In this study, WTE plants are assumed to operate throughout the year. Therefore, the plant capacity LFG recovery, incineration, and AD technologies are determined as follows:

$$G_{P(i)} = ERP_i/24 \times 365 \quad (50)$$

Where, $G_{P(i)}$ is the plant capacity (kW) for WTE technology *i*.

5.2.4. Economic feasibility analysis

The economic assessment is performed based on the LCOE, NPV, and PBP for technology comparison.

5.2.4.1. Net present value (NPV)

NPV is the sum of annual cash flow based on the discount rate over the project's lifetime. The project is considered economically viable when the NPV is positive, and it can be calculated as (Nubi, Morse, and Murphy 2022):

$$NPV = \sum_{n=0}^{y} \frac{P_n}{(1+\alpha)^n} = P_0 + \frac{P_1}{(1+\alpha)^1} + \cdots + \frac{P_y}{(1+\alpha)^y} \quad (51)$$

Where, P_n is the net cash flow rate; α is the annual discount rate taken as 10% (Ayodele et al. 2018); *y* is the economic period of the project; and P_0 is the initial investment cost. P_n can be determined as:

$$P_n = Rev - OPEX - P_{tax} - INVEST_{cost} \quad (52)$$

$$Rev = ERP \times FIT + Fee_{gate} \times MSW \quad (53)$$

$$P_{tax} = Profit \times R_{tax} \quad (54)$$

$$Profit = Rev - O\&M_{cost} \quad (55)$$

Where, *Rev* is the revenue from the WTE plant; *OPEX* is the operation and maintenance cost; P_{tax} is the tax paid on the profit made from the WTE plant; $INVEST_{cost}$ is the initial investment cost; *FIT* is the feed-in-tariff; Fee_{gate} is the gate fee for waste disposal; *Profit* is the accrued profit from the plant; R_{tax} is the annual marginal tax rate of Cambodia (20%). The purchasing cost of electricity for a biomass-fired plant in Cambodia was from 0.095 to 0.120 USD/kWh (Sokrethya et al. 2023); hence, the feed-in tariff was taken as 0.095 USD/kWh.

5.2.4.2. Payback period (PBP)

PBP is the time at which the project can recover the amount invested. It is the maximum period of the year in which a project starts to have a return on investment, and it can be calculated as (Cudjoe et al. 2021):

$$PBP = \frac{TLCC}{Proft} \quad (56)$$

$$TLCC = INVEST_{cost} + \sum_{n=1}^{y} \frac{O\&M_{cost}}{(1+\alpha)^y} \quad (57)$$

Where, *TLCC* is the total life cycle cost of the WTE project.

5.2.4.3. Levelized cost of electricity (LCOE)

LCOE is the minimum cost of the electricity generated (in USD/kWh) at which the system breakeven (Ogunjuyigbe et al. 2017). LCOE serves as a vital economic indicator to measure the economic viability of a project, and it can be calculated as (Ogunjuyigbe et al. 2017):

$$LCOE = \left(\frac{TLCC}{ERP_i}\right) \times \left(\frac{\alpha(1+\alpha)^y}{(1+\alpha)^y - 1}\right) \quad (58)$$

5.2.4.4. Capital investment and operating expenditure

Investment and operating cost for LFG recovery

This study considered the internal combustion engine (ICE), commonly used for electricity generation from LFG recovery and AD plants, because of its low cost and high efficiency. The investment cost of the LFG recovery ($CAPEX_{LFG}$) can be calculated as (Nubi et al. 2022; Ogunjuyigbe et al. 2017):

$$CAPEX_{LFG} = C_{(v)} + C_{(w)} + C_{(k)} + C_{(e)} + C_{(ICE)} \quad (59)$$

Where, $C_{(v)}$ is the installation cost of the vertical gas extraction wells, $C_{(w)}$ is the cost of installing wellheads and pipe gathering, $C_{(k)}$ is the cost for installation of the knockout, blower, and flare system, $C_{(e)}$ is the cost of engineering, permitting, and surveying, and $C_{(ICE)}$ is the cost of installing reciprocating internal combustion engine. These costs can be calculated as follows:

$$C_{(v)} = 85 \times W_n \times (D_{well} - 10) \quad (60)$$

$$C_{(w)} = W_n \times 17{,}000 \quad (61)$$

$$C_{(k)} = \left(FR_{CH_4}\right)^{0.6} \times 4600 \quad (62)$$

$$C_{(e)} = W_n \times 700 \quad (63)$$

$$C_{(ICE)} = \left(1300 \times G_{P(LFG)}\right) + 1{,}100{,}000 \quad (64)$$

Where, W_n is the number of wells dug at the site; FR_{CH4} is the methane flow rate; and D_{well} is the depth of the well assumed to be 15 m.

The operating and maintenance expenditure for LFG recovery ($OPEX_{LFG}$) consists of two components: fixed operation and maintenance of the landfill site cost ($O\&M_{fixed}$) and variable operation and maintenance cost ($O\&M_{variable}$). The calculation of costs associated with operation and maintenance is as the following (Ayodele et al. 2018).

$$OPEX_{LFG} = O\&M_{fixed} + O\&M_{Variable} \quad (65)$$

$$O\&M_{fixed} = O\&M_{cost(LF)} + O\&M_{cost(ICE)} \quad (66)$$

$$O\&M_{cost(LF)} = 2600 \times W_n + 5100 \quad (67)$$

$$O\&M_{cost(ICE)} = 0.025 \times ERP_{LFG} \quad (68)$$

$$O\&M_{variable} = ERP_{LFG} \times 4.4/1000 \quad (69)$$

Where, $O\&M_{cost(LF)}$ and $O\&M_{cost(ICE)}$ are the costs for scheduled operation and maintenance of the landfill and the IEC, respectively, and 4.4 is the cost for unscheduled expenditure and maintenance of the system (Hadidi and Omer 2017).

Investment and operating cost for incineration

The models for estimating the capital expenditure ($CAPEX_{inc}$) and the fixed operating expenditure ($Fixed\ OPEX_{inc}$) of an incinerator were adopted from Alzate-Arias et al. (2018), as given in Equations (70) to (71). The calculation of variable operating cost for incineration follows Equation (69).

$$CAPEX_{inc} = 16{,}587 \times G_{P(inc)}{}^{0.82} \quad (70)$$

$$Fixed\ OPEX_{inc} = CAPEX_{inc} \times 4\% \quad (71)$$

Investment and operating cost for AD

The CAPEX and fixed OPEX were calculated for AD technology as shown in Equations (72) to (73) (Hadidi and Omer 2017; Nubi et al. 2022). The calculation of the variable operating cost of the AD plant follows Equation (69).

$$CAPEX_{AD} = Cost_{Install} \times G_P \quad (72)$$

$$O\&M_{fixed} = CAPEX_{AD} \times 3\% \quad (73)$$

Where, $Cost_{Install}$ is the installation cost of the AD plant, which is taken as 4,339 USD/kW (Nubi et al. 2022).

Table 26. Input parameters for economic analysis

Parameter	Value
Electricity price (USD/kWh)	0.095[a]
Discount rate (%)	10[b]
Gate fee (USD/ton)	1.00
Internal use of electricity (%)	20[c]
Marginal tax rate (%)	20
Variable OPEX for LFG and AD (%)	4.40[d]
Vaiable OPEX for incineration (%)	4.00[d]
General inflation rate (%)	5.48

[a]Sokrethya et al. (2023), [b]Ayodele et al. (2018), [c]Xin et al. (2020), [d]Hadidi and Omer (2017)

5.2.5. Environmental performance evaluation

The environmental assessment in this study focused on evaluating the impact on GWP. The calculation of GWP considered two main factors. Firstly, the study considered the direct emissions associated with fugitive emissions from LFG and AD technologies, as well as stack emissions from the incinerator. These emissions contribute to the release of GHG, such as CH_4 and CO_2, which have a significant impact on climate change. Secondly, the study assessed the emission avoidance achieved through the replacement of electricity generated from conventional sources, particularly coal-fired power plants. In Cambodia, where a substantial portion of electricity production relies on coal-fired power plant (36%). Replacing this electricity with electricity generated from WET technologies can lead to a reduction in GHG emissions. The calculation of GHG emissions followed the IPCC 2006 guidelines, which take into account the 100-year GWP.

5.2.5.1. Direct GHG emissions

Fugitive CH_4 emissions from the LFG recovery and AD processes play a significant role in contributing to GWP. It is important to note that CO_2 emissions released from the landfill and biogas plants are considered of biogenic origin, meaning they come from organic sources, and therefore they are not included in the calculation of GWP. GHG emissions from the two technologies are calculated below:

$$GHG_{LFG} = Q_{CH_4(LFG)} \times (1 - OF) \times (1 - \lambda) \times \rho_{CH_4} \times GWP_{CH_4} \quad (74)$$

$$GHG_{AD} = Q_{CH_4(AD)} \times (1 - \lambda) \times \rho_{CH_4} \times GWP_{CH_4} \quad (75)$$

Where, GHG_{LFG} and GHG_{AD} are the direct GHG emissions from landfill and AD plants, respectively; ρ_{CH4} is the density of CH_4 in standard temperature (6.67×10^{-4} t/m^3).

The direct emissions from waste combustion in an incinerator are calculated following Equations (76) to (78).

$$GHG_{inc} = E_{CO_2} + E_{N_2O} \times GWP_{N_2O} \quad (76)$$

$$E_{CO_2} = MSW \times \sum(W_i \times dm_i \times CF_i \times FCF_i \times OF \times 44/12) \quad (77)$$

$$E_{N_2O} = \sum(W_i \times EF_{N_2O})/1000 \quad (78)$$

Where, GHG_{inc} is the direct GHG emissions from incineration, GWP_{N_2O} is the global warming potential for N_2O, E_{CO_2} and E_{N_2O} are the emissions of CO_2 and N_2O from incinerator; W_i is the

fraction of waste in MSW; dm_i, CF_i, and FCF_i are the dry matter content, total carbon content, and fossil carbon fraction of waste constituent i; OF is the oxidation factor, taken as 100% (IPCC 2006); 44/12 represents the molecular weight ratio of CO_2 to carbon; and EF_{N_2O} is the emission factor for N_2O, taken as 50 gN_2O/t (IPCC 2006).

5.2.5.2. GHG Emissions avoidance

To calculate the emission avoidance, the study used an emission factor for the coal-based power plant in Cambodia. In this case, the emission factor for the coal-based power plant was taken as 0.919 $kgCO_2$-eq/kWh (ACE 2021). The calculation of emission avoidance due to the implementation of WTE technologies is calculated as below:

$$GHG_{avoided} = ERP_i \times EF_{coal-fired\ plant} \quad (79)$$

Where, $GHG_{avoided}$ is the avoided emission of GHGs from electricity replacement, and ERP_i is the energy recovery potential of each WTE technology.

5.2.6. Sensitivity analysis

In this study, a sensitivity analysis was conducted to assess the impact of input parameters on the economic viability of the WTE technologies. The analysis focused on three key parameters: electricity conversion efficiency, discount rate, and electricity sale price. Electricity conversion efficiency measures the effectiveness of converting waste into electricity. A higher efficiency means more electricity can be generated from the same amount of waste input. The sensitivity analysis examined how variations in electricity conversion efficiency affected the economic performance of the WTE technologies. The discount rate reflects the concept of the time value of money and plays a significant role in economic evaluations. It determines the present value of future cash flows, and a higher discount rate reduces the present value of expected revenues. This can make long-term investments, such as WTE projects, less financially appealing. The sensitivity analysis explored different discount rates to assess their impact on the economic viability of the technologies. The electricity sale price directly affects the revenue generated by the WTE projects. It represents the price at which electricity produced from WTE technologies is sold to consumers or the grid. The sensitivity analysis investigated the effect of various electricity sale prices on the economic viability of the technologies. By examining different price levels, the analysis aimed to identify the thresholds at which the projects would achieve financial feasibility and profitability.

5.3. Results and discussion

5.3.1. MSW generation projection

The MSW generation projection for the period from 2023 to 2042 was based on various data sources. Population data was obtained from the general population census report (National Institute of Statistics 2020). Waste disposal data was obtained from the Dangkao landfill office, and waste collection efficiency was sourced from a report by the local government (PPCA et al. 2018). Based on these data, the per capita generation of MSW was calculated to be 1.50 kg/day. This represents an increase compared to the 2016 rate of 1.32 kg/capita/day (PPCA et al. 2018). The United States of America and Abu Dhabi, an emirate of the United Arab Emirates, have recorded higher MSW generation rates, amounting to 2.03 and 2.1 kg/capita/day, respectively (Dek et al. 2022; Paleologos, Caratelli, and Amrousi 2016). Nevertheless, Thailand and Vietnam have lower generation rates at 1.14 and 0.80 kg/capita/day, respectively (Dek et al. 2022). As depicted in Figure 40, the MSW generation in Phnom Penh is expected to reach 1,454,152 tons in 2023 and steadily increase to 2,980,801 tons in 2042. Assuming a collection efficiency of 92%, an average of 1,961,167 tMSW is projected to be collected and disposed of at the landfill annually between 2023 and 2042. This figure is used for the model calculations. In this study, a moving grate-firing system is considered for WTE incineration, which is designed to handle only burnable waste. The incineration capacity of this system is estimated to be 5,122 tMSW/day. This capacity indicates the maximum amount of burnable waste that can be processed and converted into energy through incineration on a daily basis.

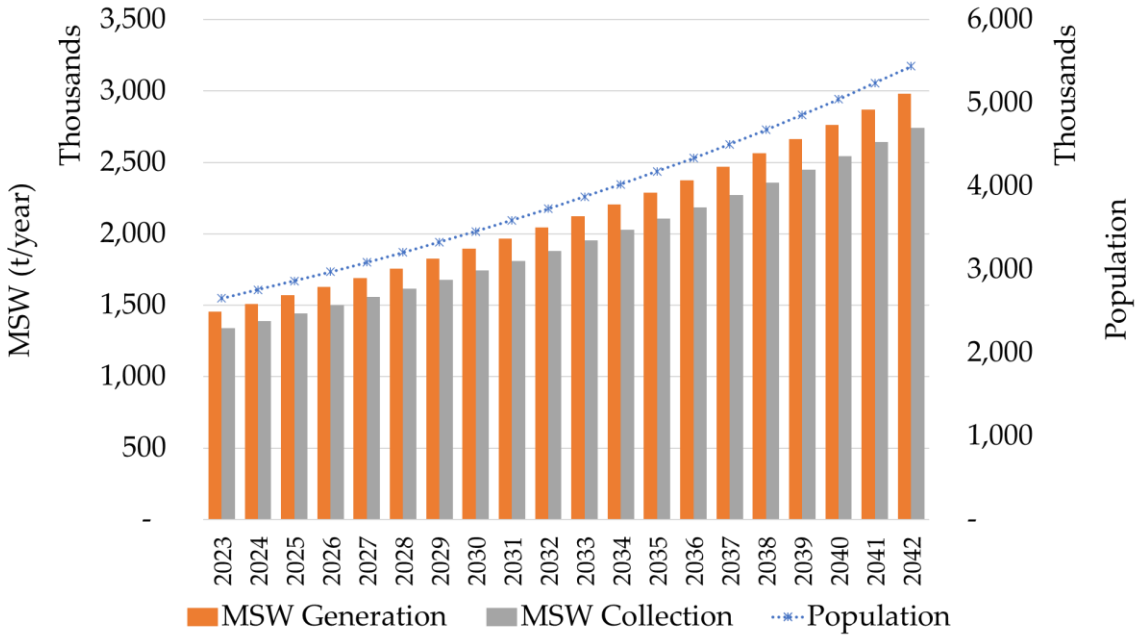


Figure 40. MSW projection for Phnom Penh municipality from 2023 to 2042

5.3.2. Energy recovery potential

The input parameters for the LandGEM model in this study were determined following the recommended values in the IPCC 2006 guidelines, as shown in Table 4. The value of *k* was estimated at 0.21, which aligns with field measurements and laboratory tests conducted in tropical landfills (Machado et al. 2009). This value falls within the range suggested by IPCC (2006) for rapidly degrading waste in moist and wet tropical regions with an annual precipitation of 1,000 mm or more. The L_0, representing the gas generation potential, was determined to be 90 m^3/t, slightly lower than the value used for wet landfill in the LandGEM model (US EPA 2005). In estimate LFG generation, only biodegradable waste types listed in Table 1 were considered. As depicted in Table 27, CH_4 generation is assumed to be zero in 2023, the initial year of waste acceptance, and will gradually increase from 2024 until 2043 as the landfill accumulates waste. After one year of landfill closure, CH_4 generation starts to decline significantly, which can have implications for the economic viability of LFG recovery. Therefore, from an economic perspective, this study considered the utilization of CH_4 for electricity generation for a period of 15 years, specifically from 2028 to 2042, based on relevant literature (Emilio et al. 2022; US EPA 2016).

Table 27. Key parameters for the LandGEM model

Parameters	Unit	Value
Landfill open	year	2023
Landfill closure year (with 80-year limit)	year	2042
Annual precipitation	mm	1550
Methane generation rate constant, *k*	$year^{-1}$	0.213[a]
Potential methane generation capacity, L_0	m^3/ton	90[a]
Non-methane organic carbon concentration (NMOC)	ppmv as hexane	600
Fraction of methane (*F*)	% by volume	50[b]
MCF for unmanaged landfill – deep (>5 m waste)		0.8[b]
Degradable organic carbon (*DOC*)		0.15[a]
Fraction of degradable organic carbon (DOC_f)		0.77[c]

[a] Calculated from Equations (5-7), [b] IPCC (2006), [c] IPCC (2002)

Within the 15-year period of CH_4 utilization for electricity generation (2028-2042), the average annual CH_4 yield was estimated to be 111 M m^3, with an average flow rate of 212 m^3/min as shown in Figure 41. This estimation is comparable to a study in Taiwan (Chen and Liu 2021). In this study, a 75% CH_4 collection efficiency is considered. This means that 75% of the generated CH_4 is

captured and recovered for electricity generation, while the remaining 25% is assumed to be emitted as fugitive emissions. This collection efficiency aligns with other studies (Ayodele et al. 2018; Emilio et al. 2022; Ogunjuyigbe et al. 2017).

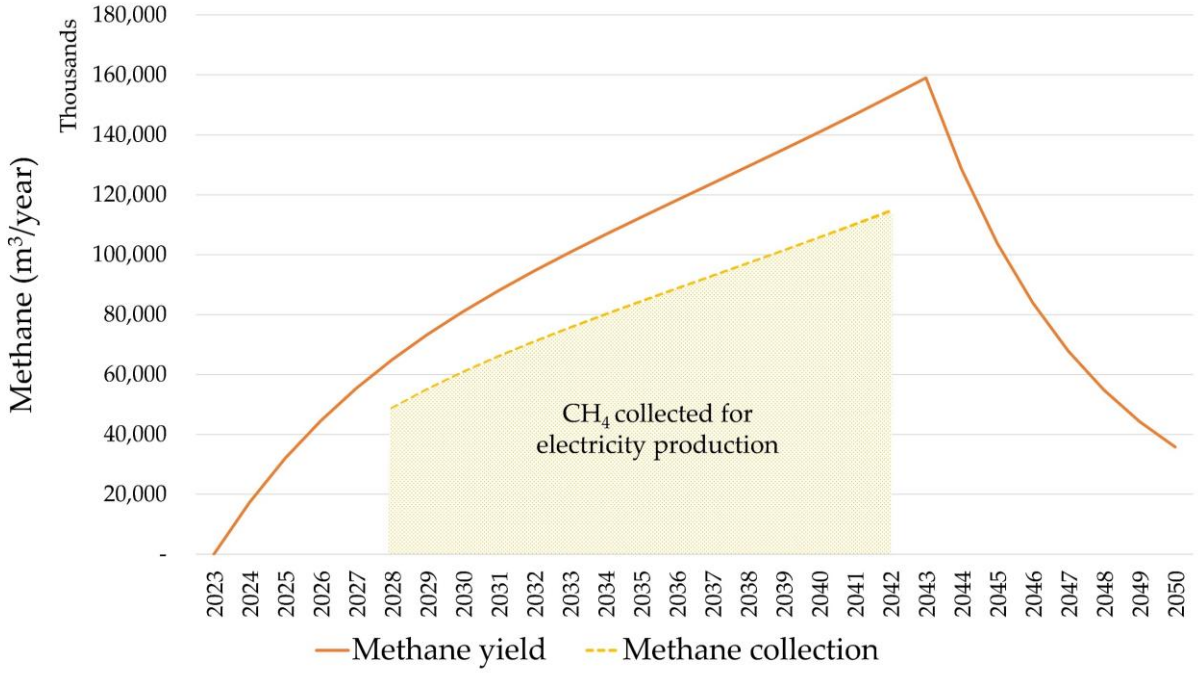


Figure 41. Annual landfill methane generation and collection

The annual ERP from the LFG recovery system was estimated to range between 120.38 and 320.52 GWh over a project lifespan (2028–2042), as shown in Table 28. This estimation consistent with a report by Ogunjuyigbe et al. (2017) when applying a similar electricity conversion efficiency. The average ERP was calculated to be 220.96 GWh/year and an installed capacity of 23 MW as shown in Table 28. This indicates that the LFG recovery system has a substantial energy generation capacity. In comparison, the ERP from a rice straw-fired plant in Cambodia, estimated to be approximately 10 MW, is about 2.5 times smaller (Sokrethya et al. 2023).

In the present study, the ERP from incineration technology, which only considers organic and burnable wastes, is significantly higher compared to that of LFG technology. The annual energy production from incineration is estimated to range between 660.94 and 1354.84 GWh/year. Comparing this with a case in Mexico, where the ERP was estimated at 537.71 GWh from the combustion of 708,900 tMSW (Emilio et al. 2022), the ERP in this study is lower. This difference can be attributed to the LHV of the waste used in the combustion process, which leads to greater energy generation in Mexico.

Table 28. ERP over the lifetime of WTE technologies

Year	LFG (GWh)	Incineration (GWh)	AD (GWh)	Year	LFG (GWh)	Incineration (GWh)	AD (GWh)
2023		660.94	162.59	2033	179.24	964.34	237.22
2024		686.39	168.85	2034	189.80	1001.47	246.36
2025		712.82	175.35	2035	200.06	1040.02	255.84
2026		740.26	182.10	2036	210.16	1080.06	265.69
2027		768.76	189.11	2037	220.18	1121.65	275.92
2028	115.44	798.36	196.39	2038	230.22	1164.83	286.54
2029	130.50	829.09	203.95	2039	240.34	1209.68	297.58
2030	144.11	861.01	211.81	2040	250.62	1256.25	309.03
2031	156.60	894.16	219.96	2041	261.09	1304.61	320.93
2032	168.23	928.59	228.43	2042	271.81	1354.84	333.29

The CH_4 generated from AD technology was estimated at 98.72 m^3/t of food waste. This value falls within the range reported by other studies that have used the same technology (Al-Wahaibi et al. 2020; Alzate, Restrepo-Cuestas, and Jaramillo-Duque 2019; Bicks 2020; Chowdhury 2021). Alzate et al. (2019) reported 71 m^3 of CH4 generated from AD plants, while Bicks (2020) presented a lower CH_4 yield at 50 m^3/t of food waste. Notably, Al-Wahaibi et al. (2020) and Chowdhury (2021) found a higher CH_4 generation rate at 123 and 200 m^3/t of food waste, respectively. The estimated ERP from AD technology in this study is about 238.35 GWh/year. This value is greater than the ERP from LFG recovery but lower than that from incineration technology.

According to the comparison presented in Table 29, incineration technology stands out as a significant electricity producer and has the potential to make a substantial contribution to the national electricity supply. It accounts for 23.63% of imported electricity. To put this into perspective, the Electricity Authority of Cambodia reported that in 2021, 11,092 GWh of energy were sold to 3,244,209 consumers (Electricity Authority of Cambodia 2021), resulting in an average consumption of 3,357 kWh/consumer/year. Based on this data, the electricity generated from incineration could potentially supply approximately 238,220 consumers. This demonstrates the significant role that incineration technology can play in meeting the electricity demands of a considerable number of consumers.

Table 29. Energy and power production

WTE plant characteristics	Unit	LFG	Incineration	AD
Mass of inputs waste	Ton	1,454,989 *	1,869,482	964,502
Operating time	h/year	8760	8760	8760
Lifespan of the WTE projects	Year	15[a]	20[b]	20[b, c]
Average electricity production within 2023-2042	GWh/year	197.89	968.91	238.35
Plant capacity	MW	23	111	27

* Only biodegradable waste fractions included in the LandGEM model.

[a] Emilio et al. (2022); US EPA (2016), [b] Ogunjuyigbe et al. (2017), [c] Ayodele et al. (2018)

5.3.3. Economic feasibility assessment

According to the economic analysis presented in Table 30, the initial investment cost of LFG recovery is USD 31,716,738, which is lower than the capital cost of the AD plant (USD 101,373,259). However, the incineration technology, which utilizes a moving grate system, requires a higher capital investment cost of USD 227,474,483. Both LFG recovery and incineration technologies are deemed economically feasible, as they result in positive NPV. On the other hand, AD technology yields a negative NPV, indicating that it is not financially favorable. The PBP for LFG recovery is calculated to be 7.13 years, while for incineration, it is 8.36 years. Comparatively, a study by Ogunjuyigbe et al. (2017) evaluated the economic feasibility of LFG, incineration, and AD technologies in various cities in Nigeria and found that the PBP for LFG and AD technologies ranged from 4.9 to 7.8 years and 1.2 to 18.6 years, respectively. In their study, incineration had a higher PBP, exceeding 20 years in all cities.

Indeed, the profitability of AD technology can be enhanced by optimizing the income generated from selling digestate for agricultural purposes. Digestate is rich in nutrients and can be used as a valuable fertilizer, providing an additional revenue stream. Abdallah et al. (2018) have highlighted the potential to increase profitability through the utilization of digestate. Tan et al. (2015), reported that approximately 30% of digestate is produced in proportion to the waste input volume. Additionally, adjusting the methane content in biogas can have a significant impact on the economic viability of AD technology. While this study assumed a methane content of 30%, it is worth noting that the methane content can vary depending on the substrate and operational conditions of the AD plant. Holden et al. (2020) reported that the methane content in biogas can be as high as 70%. By increasing the methane content by 10% to reach 40%, for example, the

economic feasibility of the AD technology can be improved, potentially reducing the PBP to 15 years. This adjustment would result in a higher biogas yield and increased electricity generation and potential revenue.

Table 30. Summary of economic feasibility assessment of the WTE technologies

Financial indicators	Unit	LFG	Incineration	AD
Cost				
Initial investment cost	USD	31,716,738	227,474,483	101,373,259
Fixed O&M cost	USD/year	7,426,072	15,829,524	3,041,198
Variable O&M cost	USD/year	916,590	4,080,768	1,048,728
Total life cycle cost	USD	95,232,512	387,186,003	152,593,851
Depreciation cost	USD/year	2,114,449	11,373,724	5,802,066
Tax	USD/year	1,343,205	8,423,313	1,405,515
Benefit				
Net present value (NPV)	USD	25,472,926	169,858,819	-5,556,540
Payback period (PBP)	Year	7.13	8.36	>20
Levelized cost of electricity (LCOE)	USD/kWh	0.070	0.053	0.093
Internal rate of return	%	18.53	16.94	8.08
Net cash flow	USD/year	5,037,019	33,693,254	6,484,177
Profit	USD/year	3,054,723	22,861,135	5,802,066

The LCOE values obtained for the LFG, incineration, and AD technologies in this study, which are 0.070, 0.053, and 0.093 USD/kWh respectively, indicate that these technologies have lower costs of electricity generation compared to the current feed-in tariff for biomass power plants in the country. This suggests that the electricity generated from these technologies can be competitive in the market.

The PBP and IRR were used to assess the investment viability of the technologies. The incineration had a PBP of 8.36 years and an IRR of 16.94%, while LFG recovery had a PBP of 7.13 years and an IRR of 18.53%. These results indicate that both technologies have competitive investment returns and can recover their initial investments within a reasonable timeframe. These findings are consistent with other studies that have reported similar economic indicators for similar projects. For example, Ayodele et al. (2018) reported an LCOE of 0.067 USD/kWh for LFG technology in Nigeria, while Emilio et al. (2022) and Xin-Gang et al. (2016) obtained similar IRRs for incineration plants in Mexico and China at 17% and 18%, respectively. Nubi et al. (2022) found

that incineration is the most promising technology, with an LCOE ranging from 0.046 and 0.062 USD/kWh and the highest IRR (45–63%).

On the other hand, the AD technology in this study showed an IRR of 8.83%, which is lower than the discount rate and indicates financial infeasibility. This result aligns with another study by Abdallah et al. (2018) that reported an IRR of 6.90% for the same technology. However, it is worth noting that Ayodele et al. (2018) obtained better economic feasibility for AD technology with an IRR of 19.3%. It is important to consider that different studies may have variation in project-specific parameters and conditions, leading to differences in economic feasibility outcomes. For example, Ogunjuyigbe et al. (2017) found financial infeasibility for incineration technology, with higher LCOE values compared to this study.

5.3.4. Environmental performance

Table 31 shows that incineration technology has the highest GWP among the three technologies, resulting in 976 M $kgCO_2$-eq/year. This high GWP is primarily attributed to the large quantity of waste being incinerated. In incineration, stack emissions are the main contributor to GWP (Dong et al. 2019). The environmental performance of incineration technology is greatly influenced by electricity generation efficiency. A higher generation efficiency leads to better emission savings. For instance, with a 25% electricity generation efficiency, approximately 1,007 $kgCO_2$-eq/kWh of GHGs is produced from an incineration plant. Increasing the plant efficiency by 5% can reduce GHG emissions by 17%. Another significant factor contributing to the high emissions from incineration technology is the composition of the feedstock. Plastic waste, which accounts for more than 20% of the incoming waste, has a significant impact on GWP due to its fossil carbon fraction, carbon content, and dry matter content. Additionally, the current disposal practice involves commingled MSW without source segregation, leading to high moisture content, which affects the incineration process and emissions. To achieve better economic benefits and minimize GHG emissions from WTE technologies, Tan et al. (2014) suggested implementing a pre-treatment step for the input waste. This pre-treatment can help improve the overall efficiency and effectiveness of the WTE process, reducing emissions, and enhancing economic outcomes.

LFG is the second-largest emission source contributing to the GWP impact. This is primarily due to the uncollected CH_4 emissions, which account for 25% of CH_4 generation from the landfill. Specifically, it is estimated that LFG recovery technology contributes 418 M $kgCO_2$-eq/year of GHGs (Table 31). On the other hand, AD technology is reported to emit the least GHGs among the three technologies. This is because AD plants can achieve a high collection efficiency of CH_4,

up to 95%. As a result, only 5% of the produced CH_4 is released into the atmosphere as emissions (IPCC 2006).

Table 31. GHG emissions and saving from WTE technologies (M $kgCO_2$-eq/year)

Technology	Direct emissions	Emission avoidance	Net emissions
LFG	418	182	236
Incineration	976	891	85
AD	79	219	–140

Figure 42 illustrates that the environmental performance of WTE technologies in terms of offsetting conventional coal-based electricity and their respective net GWP impacts. The study shows that LFG recovery has the highest net GWP, ranging from 137 to 324 M $kgCO_2$-eq/year of GHG emissions. Incineration, on the other hand, has a lower GWP impact, ranging between 58 and 119 M $kgCO_2$-eq/year. Although incineration generates a relatively high amount of GHGs, the benefits of electricity generation can help offset its global warming impacts. In contrast, AD technology has a significantly lower GWP impact, ranging from –195 to –95 M $kgCO_2$-eq/year of GHGs avoided. This suggests that AD technology has a positive environmental impact by reducing the overall emissions compared to conventional coal-based electricity.

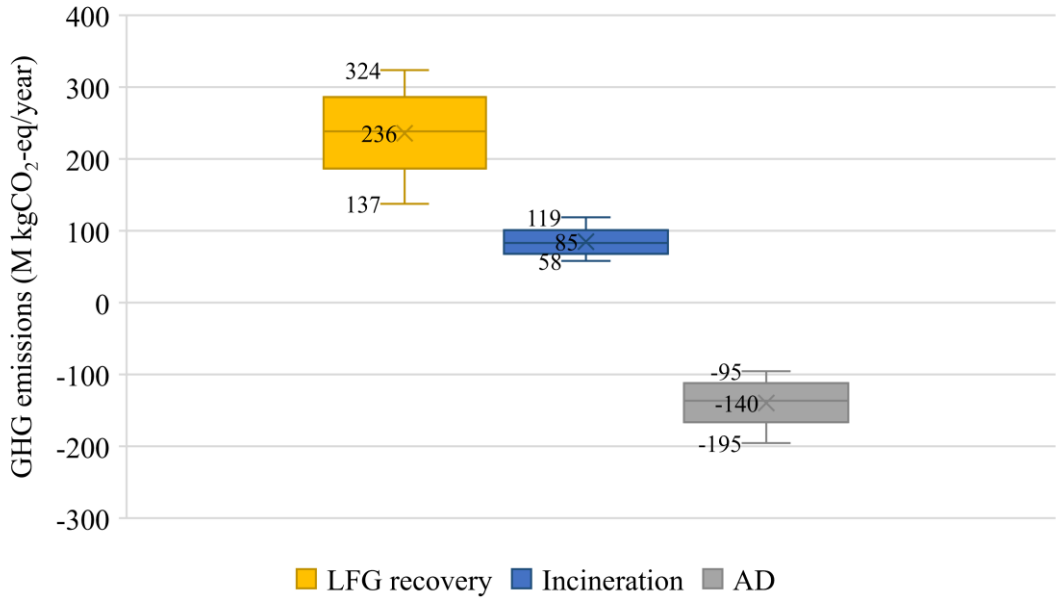


Figure 42. Net GHG emissions of WTE technologies

5.3.5. Sensitivity analysis

The sensitivity analysis conducted in the study examined the influence of the discount rate and electricity generation efficiency on the economic feasibility of WTE technologies. Figure 43 presents the variation of economic parameters such as NPV, LCOE, PBP, and TLCC with changes in electricity generation efficiency. The results of the analysis demonstrate that improving electricity generation efficiency from 15% to 40% leads to a reduction in LCOE values across all technologies. For example, the LCOE for LFG technology ranges from 0.071 to 0.069 USD/kWh, for incineration technology it ranges from 0.057 to 0.049 USD/kWh, and for AD technology it ranges from 0.180 to 0.071 USD/kWh.

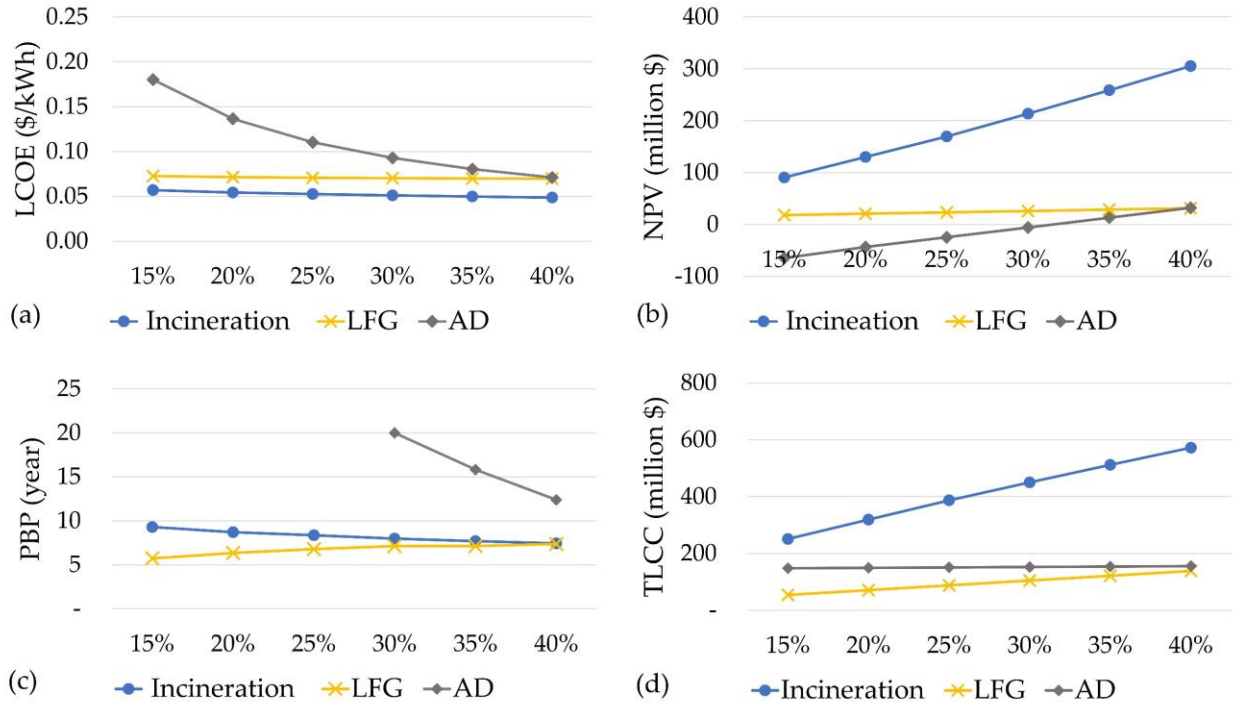


Figure 43. Influence of electricity generation efficiency on LCOE (a), NPV (b), PBP (c), and TLCC (d)

Furthermore, increasing the energy generation efficiency of AD to 32% results in a positive NPV and a reduced PBP of 19.22 years. This suggests that a minimum energy generation efficiency of 32% is necessary to make AD technology economically viable. The NPV for incineration technology shows a clear upward trend as the plant's efficiency increases, indicating that higher efficiency leads to a more favorable economic outcome. However, all WTE technologies have higher TLCC values as plant efficiency improves, implying increased overall costs over the life cycle of the plant. These findings highlight the critical role of energy conversion efficiency in determining the economic viability of WTE technologies. Improving efficiency not only reduces

the cost of electricity generation but also has a positive impact on the NPV and payback period. Therefore, maximizing energy generation efficiency is an essential factor to consider when assessing the economic feasibility of WTE projects.

Figure 44 illustrates the impacts of the discount rate on the economic evaluation of WTE technologies. The figure demonstrates how the NPV and TLCC change with varying discount rates. The results indicate that as the discount rate increases, both the NPV and TLCC consistently decrease for all WTE technologies. This implies that higher discounts lead to lower economic benefits and increased overall costs over the life cycle of the projects. For AD technology, the NPV remains positive when the discount rate is below 10%. However, as the discount rate exceeds 10%, the PBP of AD plant increases to more than 20 years. This suggests that a higher discount rate reduces the economic feasibility of AD technology, making it less attractive from an investment perspective. Additionally, the LCOE and PBP for all technologies increase substantially as the discount rate increases. This indicates that higher discount rates result in higher electricity costs and longer PBP for the projects.

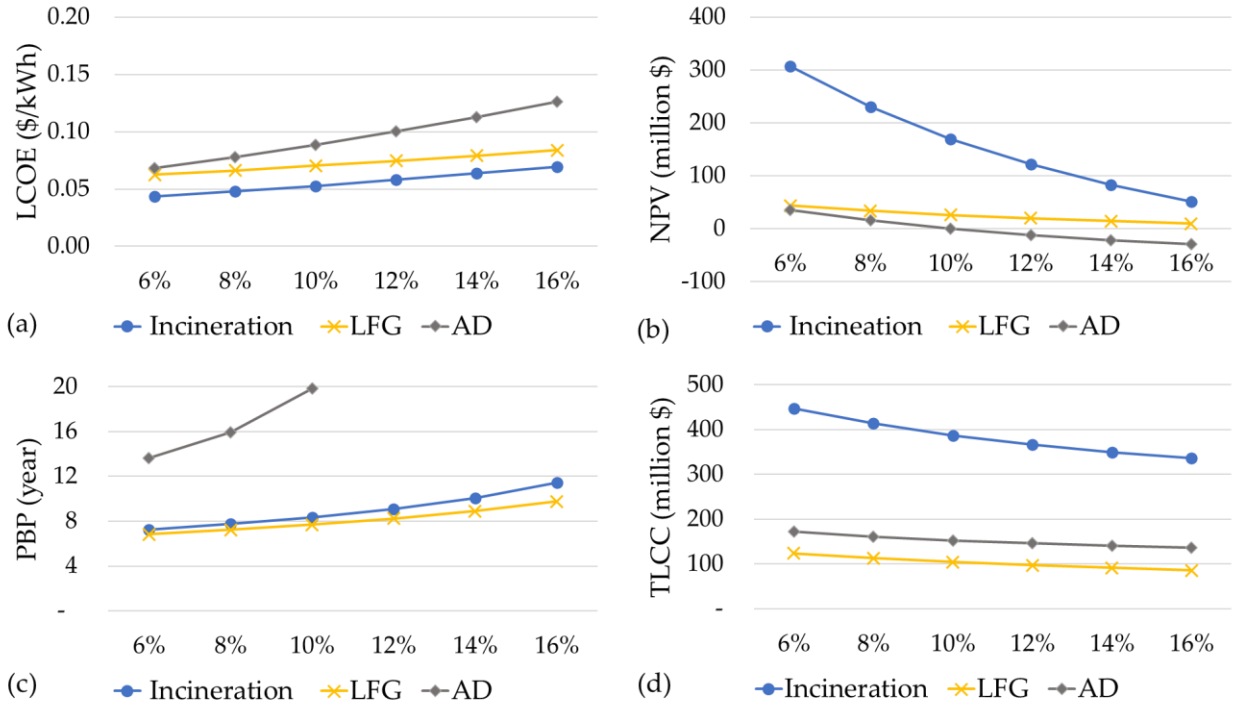


Figure 44. Influence of discount rate on LCOE (a), NPV (b), PBP (c), and TLCC (d)

To enhance the attractiveness of WTE technologies and reduce the PBP, it is crucial to implement strategies that increase profits. This can be achieved by raising the electricity price and tipping fee, as well as reducing internal energy consumption. The impact of fluctuations in electricity price

(±0-30%) on the IRR and NPV of WTE technologies is illustrated in Table 32. The findings highlight the significant influence of changes in electricity prices on both IRR and NPV. Specifically, when electricity prices decreased by 30%, the NPV for LFG recovery and AD technologies turned negative. This indicates that the revenue generated from electricity sales was insufficient to cover the costs associated with these technologies, resulting in a negative NPV. Additionally, the IRR for all three technologies experienced a decline of 10%. However, it is worth noting that increasing electricity prices by a minimum of 10% would reverse the negative NPV value of AD technology and make it positive.

Table 32. Influence of feed-in tariff on NPV, PBP, and IRR

Technology	**–30%**	**–20%**	**–10%**	**0%**	**+10%**	**+20%**	**+30%**
Net present value (USD)							
Incineration	22,461,015	71,593,616	120,726,218	169,858,819	218,991,420	268,124,021	317,256,622
LFG	–1,856,129	7,253,556	16,363,241	25,472,926	34,582,611	43,692,296	52,801,980
AD	–41,815,814	–29,729,389	–17,642,965	–5,556,540	6,529,884	18,616,309	30,702,733
Payback period (year)							
Incineration	16.63	12.38	9.95	8.36	7.22	6.36	5.69
LFG	>20	11.16	8.65	7.13	6.10	5.34	4.75
AD	>20	>20	>20	>20	17.68	14.68	12.62
Internal rate of return (percent)							
Incineration	9.98 %	12.46 %	14.77 %	16.94 %	19.03 %	21.05 %	23.02 %
LFG	7.7 %	11.73 %	15.29 %	18.53 %	21.56 %	24.44 %	27.2 %
AD	2.78 %	4.73 %	6.48 %	8.08 %	9.57 %	10.98 %	12.32 %

The landfill gate fee currently implemented in Cambodia is inadequate and relatively low when compared to other countries. For instance, in the Philippines, the landfill gate fee for waste disposal is reported to be 15 USD/tMSW (Agaton et al. 2020), whereas in the United Arab Emirates, it amounts to 14 USD/tMSW for waste disposal in waste treatment facilities (Abdallah et al. 2018). Alongside advocating for an increase in the landfill gate fee, Dong et al. (Dong et al. 2019) recommended minimizing internal electricity consumption to the greatest extent possible in order to facilitate efficient energy recovery.

WTE technologies offer substantial environmental advantages by effectively mitigating carbon emissions. Furthermore, these technologies have the potential to generate additional income through carbon credits for carbon avoidance and the sale of by-products such as digestate. These

economic aspects should be considered in the overall analysis. Tan et al. (2015) reported that the financial benefit from carbon credits is approximately 15.38 USD/tCO_2, and the sale of one ton of digestate can yield around USD 100. These financial incentives highlight the economic value associated with implementing WTE technologies. To foster increased investment in energy recovery from waste, it is crucial to develop financial and regulatory policies, such as carbon trading, renewable power credits, and renewable power production tax credits. These incentives, as suggested by Amini et al. (2012), would play a pivotal role in attracting attention to the economic benefits offered by WTE technologies.

5.4. Conclusions

This focused on assessing the energy recovery potential, economic viability, and environmental performance of three WTE technologies: LFG recovery, incineration, and AD. The results indicate that incineration exhibits the most promising outcomes in terms of energy generation and financial profitability. It is characterized by a low LCOE, high NPV, and a potential to achieve a breakeven in 8.36 years. Incineration can effectively manage MSW by accepting a wide range of organic and inorganic feedstocks, leading to significant power generation. This power can substitute electricity produced by the coal-based power plant, resulting in substantial GHG emissions savings. Specifically, incineration has the potential to replace 968.91 GWh/year and save approximately 890.75 M kgCO_2-eq/year. On the other hand, LFG recovery demonstrated an attractive investment with a PBP of 7.13 years and a higher IRR of 18.53%. However, LFG recovery technology emits a relatively large amount of GHGs. Additionally, the lifespan of the LFG recovery system is limited, while the LFG generation at the landfill site can persist for up to 100 years. AD technology, which is well-suited for managing organic waste, has the potential to significantly reduce overall GHG emissions. However, from an economic standpoint, the evaluation suggests that AD technology is financially feasible. To increase investment interest in all the WTE technologies evaluated, enhancing energy conversion efficiency, and reducing the discount rate are recommended.

While incineration technology has demonstrated remarkable profitability, there are growing concerns regarding its emissions and bottom ash management. Public acceptance of incineration has also been declining due to the potential risks of disasters and pollution. To address these concerns, it is crucial to ensure compliance with air emission standards and adopt proper bottom ash management practices to mitigate potential health risks. In developed countries, bottom ash is treated and utilized as construction materials instead of being disposed of at landfills. The focus of the present study was primarily on the impact of WTE technologies on GWP. However, it is

essential to conduct further comprehensive studies to examine other hydrocarbon emissions resulting from complete and incomplete combustions in incineration, such as Dioxins, Furans, and Benzene. The associations between these emitted hydrocarbons and human health, ecosystems, and natural resources should be evaluated through life cycle assessments. In Cambodia, the government has recently made effort to encourage investment in WTE technologies. However, clearer guidelines are necessary for the adoption and implementation of these technologies. Regulations and incentive policies, including investment subsidies, tax exemptions, carbon credits, etc. should be implemented to make WTE projects more attractive in commercial schemes. This analysis serves as crucial fundamental information for the development of sustainable MSW management through WTE technology in Phnom Penh, Cambodia.

CHAPER 6

GENERAL DISCUSSION AND CONCLUSIONS

6.1. General discussion

6.1.1. MSW generation

The rapidly increasing trend of MSW generation has become a pressing global issue, drawing attention of countries worldwide regarding its management. In the MSW stream, waste is commonly classified into two main categories: organic and inorganic. Organic waste primarily consists of food waste, wood, and garden waste. These materials are biodegradable and can decompose over time. On the other hand, inorganic waste comprises recyclable materials such as paper, plastic, glass, and metals, as well as other slowly degradable wastes like leather, rubber, nappies, and other inert materials. The composition of MSW can vary significantly depending on several factors. The level of economic development influences consumer behavior and the types of products consumed, thereby affecting the composition of waste. Cultural practices and norms also play a role in waste generation patterns. Geographical location, energy sources, and climate conditions can influence the availability and consumption of certain materials, further impacting waste composition (Hoornweg and Bhada-Tata 2012). High-income countries tend to have a higher proportion of inorganic waste in their MSW stream due to increased consumption of packaged goods, disposable products, and processed materials. In contrast, low- and middle-income countries often have a higher percentage of organic waste in their waste stream, reflecting traditional agricultural practices, limited access to processed goods, and a reliance on locally sourced natural products (UNEP 2015). These countries face significant challenges in effective management of MSW due to limitations in human resources, insufficient financial resources, and a lack of technology.

As the volume of MSW continues to rise, managing MSW has become increasingly complex, necessitating a clear strategy involving the public, private sector, and governmental agencies. The management of MSW is often hindered by a shortage of trained personnel and skilled workers. Limited human resources can impede the implementation of effective waste management practices, resulting in inadequate waste collection, improper disposal methods, and increased environmental risks. Insufficient financial resources pose another challenge for MSW management, particularly in low-income countries. Establishing and maintaining waste management infrastructure, such as waste treatment plants, recycling facilities, and landfill sites, requires substantial investments.

Furthermore, the lack of access to advanced waste management technologies further exacerbates the challenges faced by these countries. Advanced technologies, such as waste-to-energy conversion, composting, and recycling processes, can significantly contribute to waste reduction, resource recovery, and environmental sustainability (Pheakdey, Quan, and Xuan 2023). However, limited access to these technologies restricts the potential for efficient and environmentally friendly waste management practices, leaving the countries heavily reliant on conventional and often unsustainable waste disposal methods.

Cambodia has no exception, with organic waste accounting for more than 50% of MSW generated in the country. In many cases, the lack of proper source separation practices, collection systems, and intermediate treatment facilities results in the mixing of organic waste with recyclables and other inorganic waste, making it challenging to effectively manage and recover valuable resources from waste disposal. This commingled waste is typically sent to landfills or subjected to open burning where the collection services are not available.

6.1.2. Environmental pollution associated with MSW landfilling

Improper management of organic-rich waste, especially at landfill sites, has a profound impact on human health, the local and global environment, and the economy. One of the major environmental issues arising from MSW disposal is the generation of leachate. Leachate is formed through the decomposition of biodegradable waste within landfills and can contain various pollutants. If not properly managed and treated, leachate can seep into the surrounding environment, contaminating water sources and soil, and potentially entering the food chain (Chu et al. 2019; Makuleke and Ngole-Jeme 2020). Of particular concern are the pollutants present in leachate, including heavy metals, which have the potential to cause adverse health effects when they penetrate from the landfill into the surrounding ecosystem (Abdel Gawad 2018). An investigation of heavy metal concentrations in surface water, groundwater, soil, plant, and fish in this study revealed that all samples exceeded the permissible limits for Cd. The elevated levels of heavy metals, particularly Cd, in samples may be attributed to the presence of these metals in MSW disposal (Vongdala et al. 2019). Cd is typically used in batteries and rechargeable Ni-Cd batteries. These Cd contain materials are mostly not recycled and finally end up in the landfill mixing with other household waste (Järup 2003). Improper landfill management has led to penetration of heavy metals into the surrounding environment (Parvin and Taraq 2021). Specifically, the concentration of Cd in the surface water ranged from 0.38 to 0.53 mg/L, four times higher than the levels previously reported in leachate effluent (Xaypanya et al. 2018). Spatial distribution analysis indicates that the

accumulation of all heavy metals in surface water samples was found in the middle and downstream of the canal, clearly indicating that these pollutants have migrated from the landfill into the canal and subsequently flowed downstream. The groundwater collected in this study has been used for cleaning, washing, watering plants, and sometime even for drinking, which poses human health problems due to excessive levels of Cd, Cr, and Ni beyond the WHO drinking quality standards. Among the groundwater sampling sites, GW1, located in the landfill site, exhibited the highest levels of heavy metals. This finding explains the migration of heavy metals from the landfill site, polluting the groundwater that is the closest to landfill (Hredoy et al. 2022). The concentration of Cd in soil samples ranged from 36 to 49 mg/kg, higher than the previously recorded levels of 10-18 mg/kg (Xaypanya et al. 2018). This indicates that soil particles could absorb Cd in contaminated surface water near the landfill, and its concentration may increase over time as more waste materials decompose, persisting in the environment for several decades (Jaishankar et al. 2014). Among the soil sampling locations, SS2, situated in inundation areas adjacent to the closed cell landfill, presented the highest concentrations of Cd, Cr, Cu, and Ni (Figure 16), confirming the findings of a study by Xaypanya et al. (2018). The concentration of heavy metals in the soil can also influenced by fertilizer and pesticide, as well as the discharge of sewage and industrial wastewater (Kumari and Mishra 2020). The land areas near the landfill have been used for paddy cultivation, which involves the application of chemical fertilizer and pesticides to enhance rice productivity, contributing to an increased presence of heavy metals in the soil. Furthermore, due to rapid urbanization, agricultural lands surrounding the landfill sites have been transformed into residential and industrial areas, resulting in a significant rise in the number of factories and property development projects. In the absence of proper treatment systems, industrial wastewater and construction and demolition waste are illegally discharged into nearby drainage systems and vacant land (UNEP 2018). The excessive levels of heavy metals in the soil can lead to soil degradation and a loss of fertility due to a decrease in both microbial and enzymatic activity (Kumari and Mishra 2020). Heavy metals that accumulate in soil can be absorbed by plants. In this study, heavy metal concentrations in plant samples, except for Cd, were found in safety limits for human health. The accumulation of Cd in plants may be attributed to the elevated concentration of Cd in surface water and soil (Hossain et al. 2018) due to the fact that Cd has a rate of transfer from soil to plants (Satarug et al. 2010). *Oryza sativa* is a staple crop for the daily living of Cambodians and *Ipomea aquatica* is a common vegetation consumed by local people and used as animal feed. The concentration of Cd in *Oryza sativa* exceeded the permissible limits of the WHO, and was two-fold higher than the level of Cd recorded in *Oryza sativa* plants in various paddy fields in Cambodia (Hu and Seyfferth 2021). The concentration of Cd in *Ipomea aquatica* in study was

consistently found to be high in a study conducted in Bangladesh by (Hredoy et al. 2022). An excessive level of Cd in these plants may pose potential health risks to humans. Additionally, although the concentrations of Cr, Cu, Ni, Pb, and Zn in soils are currently within acceptable limits, their accumulation in the soil may increase in the future due to improper management of leachate and landfill, combined with other sources of pollution. In the fish samples, Cd, Pb, and Zn were found to exceed the allowable limits of the WHO. The level of Cd and Pb were consistent with those found in Blackskin catfish collected from Choeung Ek wastewater drainage system (Holm et al. 2010). The distribution of heavy metals in plant samples was systematically found in fish samples, indicating that aquatic organisms may uptake these heavy metal compounds in water-soluble and sediment forms, transferring them to fish bodies through the food chain (Perera et al. 2015). Considering that *Oryza sativa, Ipomea aquatica,* and fish are commonly consumed by local people, an elevated level of heavy these food sources can pose health risk. This finding provides evidence of the negative impact of heavy metals leakage on the surrounding environment of the landfill. Furthermore, agricultural activities and urbanization can contribute to an increased level of heavy metals in the landfill surrounding, particularly due to the low-lying areas and discharge from these sources into the upstream canal. Therefore, to mitigate environmental pollution caused by leachate, it is crucial to implement proper waste separation before disposed of at the landfill and ensure appropriate landfill leachate management.

Another consequence associated with MSW landfilling is the emission of LFG, primarily composed of CH_4. The organic waste disposed of in landfills undergoes anaerobic decomposition, leading to the production of CH_4, a potent greenhouse gas that contributes to climate change and global warming. CH_4 emissions from landfills make up approximately 12% of total global CH_4 emissions (EPA 2006b). CH_4 is the second most prevalent GHGs after CO_2, and it has a GWP of 25 times higher than that of CO_2. When assessing landfill CH_4 generation, two approaches are commonly used: on-site field measurement and mathematical model calculations. On-site field measurement can be costly and time-consuming, especially for measuring in large-scale landfills. However, it provides more robust results when proper sampling techniques are employed, and uncontrolled emissions are taken into account. Mathematical models offer advantages over field measurements when investigating CH_4 generation in large-scale landfills and concrete input data is available. Due to the absence of LFG collection system and irregular application of soil cover, there is a high degree of uncontrolled CH_4 emissions. Furthermore, due to difficulty in managing manpower, a lack of CH_4 measurement equipment, and time constraint, this study employed two mathematical models, namely the LandGEM and IPCC FOD models, to estimate CH_4 generation

from the Dangkao landfill and compared the results with field measurement studies conducted in neighboring countries with similar context. The results revealed an average CH_4 generation of 24 and 29 kg/tMSW from 2009 to 2022 using the LandGEM and IPCC FOD models, respectively. These findings in the range found in field measurement results reported in Thailand (22 $kgCH_4$/tMSW) (Chiemchaisri et al. 2007) and Vietnam (42 $kgCH_4$/tMSW) (Anh et al. 2021). Within this period, the LandGEM model estimated an average emission of 19 M $kgCH_4$/year from the landfill, while the IPCC FOD model estimated an average emission of 21 M $kgCH_4$/year. CH_4 generation was observed to be highest during landfill operation as waste accumulated, and it rapidly declined after landfill closure. This natural phenomenon is commonly observed in tropical regions due to the high proportion of food waste, moist tropical climate, and high precipitation (Ishigaki et al. 2011). In quantification of overall GHG emissions from landfill management, all GHGs were normalized into CO_2 equivalent. The current landfill management practice, which lacks leachate treatment and LFG collection systems, emits an estimated amount of 397 and 496 M $kgCO_2$-eq/year based on the LandGEM and IPCC FOD models, respectively. A previous study conducted by Hoklis and Sharp (2014) estimated GHG emissions using the IPCC FOD model, resulting in an average of 331 M $kgCO_2$-eq/year. The lower estimation in their study can be attributed to a lower amount of MSW disposal. The presence of untreated leachate and uncollected LFG can have adverse impacts on the environment and human health. To reduce the environmental burden of leachate generation, a landfill management with leachate treatment scenario was initiated. This scenario resulted in relatively higher GHG emissions compared to the current landfill management scenario. This increase is primarily due to substantial emissions from leachate treatment processes, such as usage of chemicals, water, electricity, and diesel fuel. The similar finding was found in landfill management in China when applying the same method (Yang et al. 2013). Despite the higher emissions, implementing leachate treatment can help reduce environmental pollution associated with the migration of heavy metals and other pollutants. Further assessment was conducted by considering landfill management scenario with leachate treatment and 50% of gas collection for flaring. This approach showed a significant reduction in overall GHG emissions by 55%. Recognizing that landfill CH_4 can be utilized for electricity generation, an advanced landfill management strategy with leachate treatment and LFG recovery for electricity generation was developed and evaluated. In this scenario, it was estimated that an average of 51 and 61 GWh/year of electricity could be generated, based on the LandGEM and IPCC FOD model, respectively. This electricity production has the potential to offset GHG emissions from conventional sources, resulting in a reduction of approximately 30 and 36 M $kgCO_2$-eq/year. Overall GHG emissions from this scenario were reduced by at least 83%,

highlighting the importance of utilizing LFG collection for energy production as a significant means to mitigate GHG emissions. Among the various categories of GHG emissions in landfill management, fugitive CH_4 emissions accounted for the highest proportion, amounting to more than 87% in all scenarios. Although the LFG recovery scenario resulted in a significant reduction in GHG emissions, there is still a portion of CH_4 that remains uncollected, and there is a potential for leachate leakage due to improper liner installation. Therefore, in the context of the circular economy, landfilling is considered the least favorable option in MSW management, and priority is given to recycling and recovery practices.

6.1.3. MSW management options

To mitigate environmental pollution, particularly heavy metals leakage and GHG emissions from MSW disposal, it is crucial to reduce waste landfilling as it serves as a source of pollutants. In a circular economy, the waste hierarchy is emphasized, which includes the following principles: reduce, reuse, recycle, recovery, and final disposal. The priority in the waste hierarchy is to reduce waste generation by minimizing consumption and promoting sustainable production practices. This involves measures such as reducing packaging, promoting durable and long-lasting products, and encouraging responsible consumption habits to prevent waste at its source. By promoting the reuse of items, the need for new production is minimized, leading to reduced waste generation. This, in turn, helps to reduce GHG emissions associated with waste transportation and lowers the expenditure on waste collection services. Recycling plays a crucial role in a circular economy by transforming waste materials into new products or regeneration. Recycling processes help conserve natural resources, reduce energy consumption, and minimize the environmental impact associated with extracting and manufacturing virgin materials. Among the MSW treatment methods examined in this study, recycling emerges as the most favorable treatment option, yielding significant environmental benefits. Recycling not only helps in reducing waste but also contributes to the conservation of natural resources and the preservation of the environment. In this study, recycling can save approximately –2,149 kgCO_2-eq/t recyclables. Furthermore, recycling provides a positive contribution to increasing carbon stock in nature (Turner et al. 2015). In the Phnom Penh Waste Management Strategy and Action Plan 2018-2035, recycling of recyclable is set to be ≥ 50% by 2023 and ≥ 95% in 2035. Therefore, if these targets are achieved, recycling can help savings approximately –1.19 M kgCO_2-eq/day in 2023 and –3.74 M kgCO_2-eq/day in 2035. Furthermore, recycling of organic waste through composting offers significant environmental advantages. In this study, it was found that by employing composting instead of landfilling for organic waste management, approximately 84% of GHGs can be reduced. This finding is consistent with the

results of a study conducted by Seng and Kaneko (2012). Moreover, the utilization of compost in agriculture serves multiple benefits, including the substitution of chemical fertilizers. This not only offsets GHG emissions associated with fertilizer production but also contributes to enhanced carbon storage in the soil, thus promoting soil health and fertility. Recovery is another important aspect of the waste hierarchy, which involves the extraction of energy from waste through processes like waste-to-energy incineration or anaerobic digestion. These methods contribute to generating renewable energy from waste streams. Utilizing energy derived from WTE technologies can serve as a substitute for electricity production from high-pollution sources. This this study, the net GHG emissions from incineration were estimated to be about 111-248 $kgCO_2$-eq/tMSW, taking into account waste separation rates. To enhance energy recovery efficiency, it is crucial to separate food waste from the waste stream due to its high moisture content (Leckner 2015). Among the waste fractions, plastic waste demonstrated the greatest contribution to energy generation from incineration. However, the combustion of plastic waste can also result in the highest GHG emissions. Therefore, to reduce GHG emissions from incineration, it is important to minimize the incineration of plastic waste and instead focus on increasing its recycling. On the other hand, AD was found to have net emissions of approximately –160 $kgCO_2$-eq/tMSW. These findings indicate that AD not only helps to mitigate emissions but also saves a considerable amount of GHGs. Therefore, both WTE incineration and AD play important roles in waste management by reducing waste volume, generating renewable energy, and contributing to GHG emission reduction. Landfilling is regarded as the last resort for final disposal of waste and should only be contemplated for materials that cannot be reused, recycled, or recovered. This study found that GHG emissions from landfilling exceeded 1,000 $kgCO_2$-eq/tMSW, which is consistent with findings in other studies (Chen and Liu 2021; Sarbassov et al. 2020; Xin et al. 2020). Among the five scenarios, the highest GHG emissions were observed in the current MSW management practice (scenario 1), which predominantly relies on landfilling. This scenario released approximately 3.89 M $kgCO_2$-eq/day of GHGs. To address landfill waste reduction, scenario 2 was developed, considering 50% of mixed waste used for incineration and the remaining 50% for landfill. The GHG emissions in this scenario were reduced by 30% compared to scenario 1. In scenario 3, a 25% waste separation for recycling and AD resulted in GHG emissions of approximately 0.10 M $kgCO_2$-eq/day. Similarly, increasing separation rates by 50% in scenario 4 and 75% in scenario 5 resulted in a potential savings of approximately –0.69 M $kgCO_2$-eq/day and –1.59 M $kgCO_2$-eq/day, respectively. These savings were achieved through utilizing composting, AD, recycling, and incinerating. These findings indicate landfilling of MSW represents the worst-case scenario, as it emits the highest amount of GHGs. Furthermore, it is evident that a standalone MSW treatment

approach cannot effectively reduce the environmental burden of MSW. Instead, an integrated MSW management system that emphasizes recycling and resources recovery is highly recommended, as it leads to the most significant reduction in environmental impacts. Furthermore, the successful implementation of such a strategy requires the active involvement of public participation in source separation.

6.1.4. Energy recovery potential and its feasibility

As discussed in Chapter 4, WTE technologies have proven to be effective in both recovering energy resources from waste and mitigating GHG emissions in MSW management system. Furthermore, the electricity generated through these technologies can serve as a substitute for electricity produced by conventional high-pollution sources. Therefore, it is crucial to assess the economic feasibility of WTE technologies to determine the most suitable option for the study area. In the context of this study, an evaluation of the economic feasibility of WTE technologies, including incineration, AD, and LFG recovery, was conducted. The comparison of these technologies was based on three main pillars: energy recovery potential, economic viability, and GHG mitigation. Among the WTE technologies assessed, incineration demonstrated the highest electricity production, followed by AD technology. However, when considering economic feasibility, only incineration and LFG recovery were found to be viable options, while AD had a negative NPV. The LCOE for the incineration, LFG recovery, and AD technologies were 0.053, 0.070, and 0.093 USD/kWh, respectively. These values are lower than those estimated by PPCA et al. (2018) for incineration at 0.12 USD/kWh, while GGGI (2020) estimated it at 0.101 USD/kWh. Despite its economic limitations, AD technology offers potential environmental benefits by reducing a significant amount of GHG emissions through controlled decomposition. Furthermore, the digestate produced from AD technology can be sold to improve soil quality and generate additional income. Unlike incineration and AD technologies, waste disposal in the landfill continues to decompose and generate LFG even 100 years after waste acceptance. As a result, GHGs can be emitted from the landfill even after the closure of LFG recovery project, contributing significantly to global warming. Therefore, AD technology is recognized as the best alternative for treating organic waste (Kristanto and Koven 2020; Kumar and Samadder 2017). By increasing CH_4 productivity by 10%, it could make AD technology economically feasible and reduce the PBP to 15 years. Hence, to enhance CH_4 production in biogas, food waste should be properly separated, or pre-treatment should be carried out to improve the solubility of the organic matter, accelerate the degradation rate, reduce the retention time of AD, and increase biogas productivity (Tan et al. 2015). Furthermore, AD has the potential to generation additional incomes through the sale of

digestate and by accessing carbon credit programs, owing to its significantly contribution to GHG emissions. Overall, incineration weighed a better WTE technology for Phnom Penh as it yielded the highest electricity generation and economic viability, which is similarly found in Malaysia (Tan et al. 2015). However, concerns have been raised associated with hydrocarbon emissions like furan and dioxin and bottom ash management (Dong et al. 2019). The application of recent advanced technologies in flue gas emissions and the proper management of bottom ash would add value to incineration. However, implementing such technologies requires substantial investment. Therefore, it is crucial for the government to implement supporting policies, regulations, and incentives that promote sustainable waste management to enable the development of necessary infrastructure and technologies. Additionally, effective governance and coordination among government agencies responsible for waste management can streamline processes, ensure compliance with regulations, and improve overall waste management performance.

6.2. General conclusions

This Ph.D. dissertation is designed to tackle the environmental issues associated with MSW landfilling. The specific environmental problems and their corresponding solutions are directly integrated into the main objective of the dissertation, as outlined in section 1.7, and summarized below:

- Investigating the potential migration of heavy metals into the surrounding environment of the landfill and assessing the contribution of landfill CH_4 to global warming potential.
- Quantifying GHG emissions from developed alternative MSW management strategies for enhanced resource recovery and reduced environment burden. This study specifically proposes and explores alternative strategies for the management of MSW. Implementing these strategies can minimize the environmental burden associated with landfilling and contribute to a more sustainable waste management approach.
- Assessing the energy recovery, economic viability, and environmental performance (3Es) of WTE technologies. The dissertation evaluates the potential of WTE technologies to recover energy from MSW. It examines the economic feasibility and environmental performance of these technologies, considering factors such as energy generation, cost-effectiveness, and environmental impact. This analysis contributes to understanding the viability and sustainability of implementing WTE technologies as a waste management solution.

Each objective has been addressed in separate scientific papers submitted to peer-reviewed international journal, showcasing the specific findings and conclusions of each study. Limitations and potential avenues for future research are also discussed in each chapter. The main conclusions of the dissertation are described in the subsequent section.

The dissertation provides valuable insight into mitigating environmental issues associated with MSW landfilling, with a particular focus on prioritized issues such as heavy metals leakage and GHG emissions, which contribute to global warming potential. The combined effects of heavy metals contamination in nearby ecosystems and the generation of LFG leading to GHG emissions can result in both local and global pollution, as well as negative impacts on human health. To address these issues, the dissertation proposes potential alternatives for the management of MSW and evaluates their contribution to the global warming potential reduction, following the life cycle assessment method and the IPCC 2006 guidelines. To simplify the quantification of GHG emissions, an Excel spreadsheet was developed. The management of MSW is indeed a complex process that cannot be universally applied from one location to another. This is primarily due to the diverse physical characteristics of MSW, variations in climate conditions, differences in separation and collection efficiencies, availability of technologies, and considerations of resources efficiency. Each region or locality may have unique waste composition, moisture content, density, and chemical composition, which require tailored approaches for effective MSW management.

Another important outcome of this dissertation is the assessment of the economic feasibility of WTE technologies, which have the potential to address challenges in landfill management and the insufficient supply of electricity supply in the city. From a business perspective, priority has been given to evaluating the economic viability of these technologies, while from an environmental standpoint, the focus has been on their contribution to reducing global warming potential. By comparing the energy recovery potential, economic viability, and environmental performance of different WTE technologies, we can confidently select the most suitable technology for managing substantial amount of MSW.

The specific conclusions derived from each main chapter are summarized.

Investigation of heavy metals accumulation in the surrounding environment

- Heavy metal concentrations in surface water, groundwater, soil, plants, and fish samples collected from the landfill site and its vicinity were analyzed, revealing a high level of some heavy metals exceeding the permissible standards.

- Elevated levels of Cd concentration were observed in all samples, indicating that waste disposal may contain Cd and have the potential to penetrate the soil, reaching surface water and groundwater. This metal can accumulate in soil and surface water and substantially transfer to aquatic plants and animals.
- Other heavy metals such as Cr, Ni, and Pb were found to exceed the allowable limits in surface water, while only Cr and Ni were found above the permissible level for groundwater. Additionally, Pb and Zn were found in elevated level which beyond the safety limit for human consumption.
- Fish and plants (*Ipomoea aquatica* and *Oryza sativa*), which are commonly consumed by local people, were found to contain high levels of Cd. This poses potential health risks to humans.
- These findings indicate the presence of heavy metal contamination in the surrounding environment of the landfill site. The exceedance of permissible limits in various samples highlights the need for measures to prevent or reduce the migration of heavy metals and protect human health and the ecosystem.

GHG emissions from landfill management

- The methane constant rate (k) and potential methane generation (L_0) were developed to calculate LFG generation. The value of k and L_0 are comparable to those of other studies conducted in tropical regions.
- The calculated CH_4 emissions were observed to be high during landfill operation as waste accumulated, but rapidly declined after landfill closure. The results of CH_4 estimation were comparable to on-site measurements conducted in the neighboring countries with a similar context.
- Improvements should be made to the existing landfill management practices, such as applying regular soil cover to enhance CH_4 oxidation and regularly recirculating leachate to enhance the biodegradation of waste and reduce the risk associated with leachate leakage. Additionally, LFG collection should be prioritized for either flaring or energy recovery purposes.
- Considering carbon sequestration in quantification models could potentially reduce overall GHG emissions by approximately 10%.

MSW management scenarios

- Waste separation plays a crucial role in mitigating environmental pollution caused by landfilling of MSW and in achieving the circular economy goals by optimizing the utilization of waste resources.
- Implementing effective source separation programs can help divert organic and recyclable wastes from mixed waste streams. This would enable the adoption of appropriate treatment methods such as composting or anaerobic digestion, which can transform organic waste into valuable resources like fertilizer or biogas. Furthermore, the use of recyclables can avoid raw materials extraction, indirectly contributing to significant GHG emissions savings. Therefore, a concrete strategy is necessary for effective MSW management, involving public participation, engagement of the private sector, and contributions from the government. The key focus of such a strategy should be to minimize landfilling and maximize resource recovery from waste.
- Organic waste contributes the highest amount of GHG emissions from the landfill. Prioritizing the proper management of this waste fraction can significantly reduce the environmental impact of landfilled waste, minimize GHG emissions, and harness the potential of organic waste as a valuable resource for sustainable agriculture and renewable energy production.
- Reducing reliance on landfilling while increasing waste recycling and utilizing WTE technologies can help minimize the need for land acquisition and the high costs associated with landfill construction. Therefore, it is important for the government to implement supporting policies that promote waste recycling and resource recovery.

Energy, economic, and environmental performance of WTE technologies

- Incineration has the capability to significantly reduce the mass and volume of MSW while generating a substantial amount of electricity, making it economically viable for implementation in Phnom Penh. However, this technology also carries the risk of releasing pollutants, such as furan and dioxin, which necessitates careful attention and the implementation of appropriate mitigation measures.
- LFG recovery has demonstrated economic feasibility. However, it is important to note that GHG generation at the landfill site can persist for up to 100 years, even after implementing LFG recovery projects.
- Despite being considered economically infeasible, AD technology has shown a significant environmental impact by effectively reducing a substantial amount of GHGs. Furthermore,

AD technology appears to be the most suitable and effective option for managing organic waste.

Overall, in MSW management sector, landfilling remains a major source of environmental pollution, contributing to issues such as leachate pollutants and GHG emissions. To mitigate the environmental burden associated with landfilling, it is crucial to implement proper waste separation practices at the source of generation sources and treat the waste accordingly to its physical characteristics. Organic waste can be effectively treated through methods such as composting or AD technology. Recyclable materials should be prioritized for recycling and regeneration, as this reduces the demand for virgin materials and conserves resources. Furthermore, non-recyclable waste, which cannot be reused or recycled, can be utilized as feedstock for incineration. Incineration can be a viable option for energy recovery, but careful consideration should be given to emission control measures to minimize the release of pollutants and GHGs. Implementing waste separation can improve the quality of recyclable materials, increase the efficiency of recycling processes, and enhance lower heating value of waste intended for incineration. However, addressing the complexities of MSW management requires a multifaceted approach and the involvement of various stakeholders. Public participation and awareness are essential components of a successful waste management strategy. Educating the public about waste reduction, proper segregation, and recycling practices can significantly reduce the volume of waste generated and improve the overall effectiveness of waste management systems. Engaging the private sector is also crucial in MSW management. Collaboration with private companies can foster innovation, enhance waste collection and processing services, and promote the development of sustainable waste management solutions. Public-private partnerships can leverage the expertise and resources of both sectors to establish efficient waste management systems and create economic opportunities through recycling and resource recovery. Moreover, strong governmental support and regulation are vital in driving effective waste management practices. Governments need to establish clear policies, regulations, and incentives to promote sustainable waste management and provide financial support to enable the development of necessary infrastructure and technologies. Additionally, effective governance and coordination among government agencies responsible for waste management can streamline processes, ensure compliance with regulations, and improve overall waste management performance.

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Appendixes

Appendix 1. Heavy metal concentrations in samples

Sample ID	Sample name	Cd	Cr	Cu	Ni	Pb	Zn
		Surface water					
SW1	Canal	0.423	0.076	0.531	0.436	0.425	0.187
SW2	Canal	0.451	0.053	0.516	0.389	0.017	0.171
SW3	Pond	0.400	0.071	0.530	0.433	0.319	0.179
SW4	Pond	0.411	0.074	0.549	0.436	0.266	0.188
SW5	Canal	0.498	0.103	0.531	0.540	0.532	0.208
	Mean	0.436	0.075	0.531	0.447	0.311	0.186
	Minimum	0.381	0.040	0.494	0.376	ND	0.156
	Maximum	0.826	0.250	0.639	0.679	0.940	0.333
	Standard Deviation	0.085	0.045	0.038	0.082	0.378	0.034
	Standard limits	0.050	0.050	1.00	0.20	0.100	2.000
		Groundwater					
GW1	Borehole in landfill	0.405	0.062	0.533	0.413	ND	0.180
GW2	Borehole in village	0.377	0.048	0.506	0.376	ND	0.164
GW3	Borehole in village	0.385	0.045	0.504	0.373	ND	0.164
	Mean	0.389	0.052	0.514	0.387	-	0.169
	Minimum	0.363	0.033	0.482	0.370	-	0.151
	Maximum	0.506	0.152	0.644	0.532	-	0.227
	Standard Deviation	0.029	0.025	0.035	0.037	-	0.017
	Standard limits	0.003	0.050	2.000	0.070	0.010	3.000
Soil							
	SS1	38.642	12.880	50.878	45.041	31.055	31.727
	SS2	45.7875	14.887	59.006	49.273	48.942	35.536
	SS3	40.942	11.567	53.797	44.789	51.348	31.254
	SS4	39.211	12.429	61.054	47.374	51.819	25.039
	SS5	40.815	12.129	57.755	49.305	68.951	21.642
	Mean	41.079	12.778	56.598	47.156	50.423	29.040
	Minimum	35.898	4.683	47.760	38.636	4.996	15.949
	Maximum	71.678	44.051	102.985	77.917	96.443	67.051
	Standard Deviation	8.552	9.306	13.444	9.377	29.416	12.528
	Standard limits	0.800	100	125	35	85	140
Fish							
FT1	*Channa striata*	0.398	0.073	0.542	0.428	0.391	0.445
FC1	*Trichopodus trichopterus*	0.408	0.068	0.535	0.434	0.431	0.391
FC2	*Trichopodus trichopterus*	0.389	0.068	0.528	0.419	0.310	0.184
FD3	*Oxyeleotris marmorata*	0.398	0.075	0.535	0.440	0.483	0.451
FC3	*Trichopodus trichopterus*	0.405	0.072	0.542	0.431	0.404	0.185

FK3	*Trichopsis vittata*		0.399	0.066	0.536	0.425	0.362	0.197
	Mean		0.399	0.070	0.536	0.429	0.397	0.309
	Minimum		0.353	0.038	0.484	0.361	ND	0.160
	Maximum		0.618	0.123	0.601	0.485	0.881	1.341
	Standard Deviation		0.031	0.022	0.043	0.040	0.307	0.300
	Standard limits		0.050	2	30	0.500	0.200	0.050
Plants								
PG1	*Ipomoea aquatica*	Root	0.390	0.059	0.542	0.417	0.264	0.17
		Stem	0.489	0.074	0.576	0.439	0.451	0.181
		Leave	0.388	0.099	0.601	0.475	0.756	0.181
PG2	*Ipomoea aquatica*	Root	0.418	0.081	0.568	0.456	0.507	0.178
		Stem	0.412	0.077	0.564	0.441	0.411	0.175
		Leave	0.422	0.081	0.546	0.431	0.418	0.177
PG3	*Ipomoea aquatica*	Root	0.422	0.062	0.59	0.413	ND	0.198
		Stem	0.392	0.08	0.565	0.462	0.602	0.168
		Leave	0.387	0.064	0.533	0.41	0.228	0.17
PL2	*Nelumbo nucifera*	Root	0.386	0.099	0.596	0.475	0.77	0.169
		Stem	0.380	0.064	0.544	0.425	0.291	0.164
		Leave	0.383	0.059	0.529	0.416	0.255	0.163
PL3	*Nelumbo nucifera*	Root	0.419	0.078	0.572	0.45	0.489	0.179
		Stem	0.388	0.1	0.599	0.487	0.797	0.177
		Leave	0.385	0.064	0.541	0.425	0.221	0.17
PR2	*Oryza sativa*	Root	0.390	0.102	0.618	0.507	0.782	0.189
		Stem	0.367	0.056	0.621	0.436	ND	0.213
		Leave	0.391	0.067	0.56	0.45	0.455	0.165
	Mean		0.401	0.076	0.570	0.445	0.422	0.177
	Minimum		0.367	0.042	0.504	0.159	ND	0.159
	Maximum		0.702	0.104	0.697	0.219	0.884	0.219
	Standard Deviation		0.053	0.022	0.043	0.016	0.322	0.016
	Standard limits		0.050	1.300	10	10	2	0.60

Appendix 2. Methane generation estimation

Year	Waste input (ton)	Waste acceptance (ton)	LandGEM		IPCC	
			(Mg/year)	(m^3/year)	Mg/Year	m^3/year
2009	177,224	177,224	-	-	-	-
2010	409,336	586,560	1,537	2,304,208	2,168	3,250,374
2011	442,469	1,029,029	4,792	7,182,351	6,560	9,832,574
2012	492,380	1,521,409	7,707	11,551,496	10,127	15,179,269
2013	532,471	2,053,880	10,493	15,727,845	13,348	20,008,041
2014	617,489	2,671,370	13,090	19,620,866	16,229	24,326,238
2015	681,905	3,353,275	15,924	23,869,270	19,437	29,133,918
2016	717,435	4,070,710	18,771	28,136,721	22,641	33,936,510
2017	808,530	4,879,240	21,378	32,043,991	25,505	38,230,519
2018	965,944	5,845,184	24,273	36,382,906	28,822	43,201,962
2019	1,015,980	6,861,164	27,975	41,932,569	33,284	49,889,424
2020	1,035,878	7,897,042	31,398	47,063,634	37,264	55,855,022
2021	1,012,039	8,909,081	34,335	51,464,896	40,553	60,785,458
2022	1,288,223	10,197,304	36,499	54,708,299	42,829	64,196,796
2023			40,641	60,917,710	48,058	72,035,450
2024			32,812	49,181,814	36,257	54,345,369
2025			26,490	39,706,858	28,059	42,058,252
2026			21,387	32,057,268	22,304	33,431,201
2027			17,267	25,881,384	18,208	27,291,648
2028			13,940	20,895,293	15,244	22,849,884
2029			11,255	16,869,781	13,058	19,573,319
2030			9,086	13,619,791	11,410	17,102,124
2031			7,336	10,995,916	10,136	15,192,627
2032			5,923	8,877,535	9,126	13,679,323
2033			4,782	7,167,264	8,306	12,449,361
2034			3,860	5,786,479	7,622	11,425,415
2035			3,117	4,671,704	7,041	10,554,162
2036			2,516	3,771,693	6,537	9,798,543
2037			2,032	3,045,070	6,093	9,132,542
2038			1,640	2,458,432	5,696	8,537,676
2039			1,324	1,984,811	5,338	8,000,621
2040			1,069	1,602,434	5,011	7,511,610

Appendix 3. Economic assessment

1. Incineration

Input values			
Generation			
	Incinerated waste	*Ton/yr*	1,869,482
	Generation	*kWh/yr*	968,906,977
	Electricity generation efficiency	*%*	25%
	Capacity Factor	*%*	80%
CAPEX			
	Incinerator Capacity	*MW*	111.00
	Cost-CAPEX	*USD/kW*	16,587
	Coefficient		0.82
	Total CAPEX	*USD*	227,474,483
OPEX			
	O&M	*%*	4%
	Annual O&M increase rate (inflation)	*%*	5.48%
	Insurance (as % age CAPEX)	*%/yr*	0.00%
	Land Lease	*USD/yr*	-
	Variable OPEX	*USD/MWh*	4.00
	Average fix OPEX	*USD/year*	9,098,979
Economic			
	Period of Analysis	*Year*	20
	Discount Rate	*%*	10.00%
	Internal use of electricity	*%*	20.00%
	Tax	*%*	20.00%
	Bank loan (80%)	*USD*	181,979,586.66
	Loan interest rate	*%*	6.00%
	Down payment	*%*	0.00%
	Depreciation rate with 20 years	*%*	5.00%
	Feed in tariff in 2022	*USD/kWh*	0.095
	Gate fee	*USD*	1.000

Calculations

			Year	0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
Generation																								
Waste input	ton	1,869,482	37,389,640	-	1,275,276	1,324,374	1,375,363	1,428,314	1,483,304	1,540,412	1,599,717	1,661,306	1,725,266	1,791,689	1,860,669	1,932,305	2,006,698	2,083,956	2,164,188	2,247,510	2,334,039	2,423,899	2,517,219	2,614,132
Electricity genration	kWh	968,906,977	19,378,139,541	-	660,944,736	686,391,011	712,817,176	740,260,584	768,760,581	798,357,757	829,094,444	861,014,471	894,163,407	928,588,816	964,339,504	1,001,466,522	1,040,022,910	1,080,063,701	1,121,646,170	1,164,829,584	1,209,675,449	1,256,248,014	1,304,613,517	1,354,841,185
Internal use electricity	kWh	193,781,395	3,875,627,908	-	132,188,947	137,278,202	142,563,435	148,052,117	153,752,116	159,671,551	165,818,889	172,202,894	178,832,681	185,717,763	192,867,901	200,293,304	208,004,582	216,012,740	224,329,234	232,965,917	241,935,090	251,249,603	260,922,703	270,968,237
Actual electricity sale to national grid	kWh	968,906,977	**23,291,157,089**	-	**528,755,789**	**549,112,809**	**570,253,741**	**592,208,468**	**615,008,465**	**638,686,206**	**663,275,556**	**688,811,577**	**715,330,726**	**742,871,053**	**771,471,604**	**801,173,218**	**832,018,328**	**864,050,961**	**897,316,936**	**931,863,667**	**967,740,360**	**1,004,998,411**	**1,043,690,813**	**1,083,872,948**
PV of electricity sale to national grid				-	**528,755,789**	**499,193,462**	**471,284,083**	**444,934,987**	**420,059,057**	**396,573,884**	**374,401,760**	**353,469,253**	**333,707,063**	**315,049,844**	**297,435,700**	**280,806,325**	**265,106,680**	**250,284,786**	**236,291,594**	**223,080,753**	**210,608,498**	**198,833,578**	**187,716,967**	**177,221,888**
Cumulative electricity sale				-	**528,755,789**	**1,027,949,251**	**1,499,233,334**	**1,944,168,321**	**2,364,227,378**	**2,760,801,262**	**3,135,203,021**	**3,488,672,274**	**3,822,379,337**	**4,137,429,181**	**4,434,864,881**	**4,715,671,206**	**4,980,777,886**	**5,231,062,672**	**5,467,354,266**	**5,690,435,019**	**5,901,043,517**	**6,099,877,095**	**6,287,594,062**	**6,464,815,950**
					660,944,736	**623,991,828**	**589,105,104**	**556,168,734**	**525,073,821**	**495,717,355**	**468,002,199**	**441,836,566**	**417,133,829**	**393,812,305**	**371,794,625**	**351,007,907**	**331,383,350**	**312,855,982**	**295,364,493**	**278,850,941**	**263,260,623**	**248,541,972**	**234,646,208**	**221,527,360**
Cash outflow																								
CAPEX																								
CAPEX (turn key)	USD	227,474,483	227,474,483	227,474,483	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
OPEX																								
Fix OPEX with inflation (5.48%)	USD	15,829,524	316,590,485	-	9,098,979	9,597,603	10,123,552	10,678,323	11,263,495	11,880,734	12,531,799	13,218,541	13,942,917	14,706,989	15,512,932	16,363,041	17,259,735	18,205,569	19,203,234	20,255,571	21,365,577	22,536,410	23,771,405	25,074,078
Variable OPEX	USD	4,080,768	81,615,367	-	2,643,779	2,896,021	3,007,518	3,123,307	3,243,555	3,368,431	3,498,115	3,632,792	3,772,654	3,917,902	4,068,741	4,225,388	4,388,065	4,557,005	4,732,450	4,914,649	5,103,863	5,300,362	5,504,425	5,716,346
Loan repayments	USD	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Insurance	USD	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Land Lease	USD	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Total OPEX	USD	**18,962,183**	**398,205,852**	-	**11,742,758**	**12,493,624**	**13,131,070**	**13,801,630**	**14,507,049**	**15,249,165**	**16,029,914**	**16,851,333**	**17,715,571**	**18,624,891**	**19,581,673**	**20,588,428**	**21,647,800**	**22,762,574**	**23,935,684**	**25,170,220**	**26,469,439**	**27,836,772**	**29,275,831**	**30,790,424**
PV of cash outlow				227,474,483	**11,742,758**	**11,357,840**	**10,852,124**	**10,369,369**	**9,908,510**	**9,468,532**	**9,048,468**	**8,647,399**	**8,264,445**	**7,898,772**	**7,549,583**	**7,216,119**	**6,897,656**	**6,593,507**	**6,303,014**	**6,025,551**	**5,760,521**	**5,507,357**	**5,265,515**	**5,034,480**
Cumulative cash outflow				**227,474,483**	**239,217,242**	**250,575,082**	**261,427,206**	**271,796,575**	**281,705,085**	**291,173,617**	**300,222,086**	**308,869,484**	**317,133,929**	**325,032,701**	**332,582,284**	**339,798,402**	**346,696,058**	**353,289,565**	**359,592,579**	**365,618,129**	**371,378,650**	**376,886,007**	**382,151,523**	**387,186,003**
Cash inflow																								
Gate fee	USD		37,389,640	-	1,275,276	1,324,374	1,375,363	1,428,314	1,483,304	1,540,412	1,599,717	1,661,306	1,725,266	1,791,689	1,860,669	1,932,305	2,006,698	2,083,956	2,164,188	2,247,510	2,334,039	2,423,899	2,517,219	2,614,132
Electricity sale to national grid	USD	193,781,395	1,472,738,605	-	50,231,800	52,165,717	54,174,105	56,259,804	58,425,804	60,675,190	63,011,178	65,437,100	67,956,419	70,572,750	73,289,802	76,111,456	79,041,741	82,084,841	85,245,109	88,527,048	91,935,334	95,474,849	99,150,627	102,967,930
Benefit from internal use electricity	USD		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cash inflow	USD	**71,910,869**	**1,510,128,245**	-	**51,507,076**	**53,490,091**	**55,549,468**	**57,688,119**	**59,909,109**	**62,215,601**	**64,610,895**	**67,098,406**	**69,681,685**	**72,364,439**	**75,150,472**	**78,043,761**	**81,048,440**	**84,168,797**	**87,409,297**	**90,774,558**	**94,269,373**	**97,898,748**	**101,667,847**	**105,582,063**
PV of cash inflow				-	**51,507,076**	**48,627,356**	**45,908,652**	**43,341,938**	**40,918,727**	**38,630,993**	**36,471,166**	**34,432,092**	**32,507,020**	**30,689,586**	**28,973,760**	**27,353,862**	**25,824,531**	**24,380,703**	**23,017,600**	**21,730,708**	**20,515,762**	**19,368,745**	**18,285,856**	**17,263,511**
Cumulative cash inflow					**51,507,076**	**100,134,432**	**146,043,084**	**189,385,021**	**230,303,749**	**268,934,742**	**305,405,908**	**339,837,999**	**372,345,020**	**403,034,606**	**432,008,366**	**459,362,228**	**485,186,759**	**509,567,461**	**532,585,061**	**554,315,768**	**574,831,531**	**594,200,276**	**612,486,132**	**629,749,643**
Depreciation and tax																								
Net cash flow (before tax cash flows)	USD	42,116,567	**884,447,909**	**(227,474,483)**	**39,764,318**	**40,996,467**	**42,418,398**	**43,886,489**	**45,402,059**	**46,966,436**	**48,580,981**	**50,247,072**	**51,966,114**	**53,739,548**	**55,568,798**	**57,455,332**	**59,400,640**	**61,406,224**	**63,473,614**	**65,604,338**	**67,799,934**	**70,061,977**	**72,392,016**	**74,791,638**
Depreciation on assets	USD	10,832,118	227,474,483	-	11,373,724	11,373,724	11,373,724	11,373,724	11,373,724	11,373,724	11,373,724	11,373,724	11,373,724	11,373,724	11,373,724	11,373,724	11,373,724	11,373,724	11,373,724	11,373,724	11,373,724	11,373,724	11,373,724	11,373,724
Taxable	USD	42,116,567	884,447,909	-	28,390,594	29,622,743	31,044,674	32,512,765	34,028,335	35,592,712	37,207,257	38,873,348	40,592,389	42,365,824	44,195,074	46,081,608	48,026,915	50,032,500	52,099,890	54,230,614	56,426,210	58,688,253	61,018,292	63,417,914
Tax	USD	8,423,313	176,889,582	-	5,678,119	5,924,549	6,208,935	6,502,553	6,805,667	7,118,542	7,441,451	7,774,670	8,118,478	8,473,165	8,839,015	9,216,322	9,605,383	10,006,500	10,419,978	10,846,123	11,285,242	11,737,651	12,203,658	12,683,583
Total depreciation and tax			**404,364,065**	-	**17,051,843**	**17,298,273**	**17,582,659**	**17,876,277**	**18,179,391**	**18,492,266**	**18,815,176**	**19,148,394**	**19,492,202**	**19,846,889**	**20,212,739**	**20,590,046**	**20,979,107**	**21,380,224**	**21,793,702**	**22,219,847**	**22,658,966**	**23,111,375**	**23,577,383**	**24,057,307**
PV of depreciation and tax	USD		179,218,839	-	**17,051,843**	**15,725,702**	**14,531,123**	**13,430,712**	**12,416,769**	**11,482,243**	**10,620,676**	**9,826,154**	**9,093,256**	**8,417,018**	**7,792,886**	**7,216,685**	**6,684,590**	**6,193,089**	**5,738,963**	**5,319,255**	**4,931,251**	**4,572,462**	**4,240,599**	**3,933,562**
Cumulative depreciation and tax	USD			-	**17,051,843**	**32,777,545**	**47,308,669**	**60,739,380**	**73,156,149**	**84,638,391**	**95,259,068**	**105,085,221**	**114,178,477**	**122,595,496**	**130,388,382**	**137,605,067**	**144,289,657**	**150,482,746**	**156,221,709**	**161,540,964**	**166,472,215**	**171,044,678**	**175,285,277**	**179,218,839**

				2023	2024	2025	2026	2027	2028	2029	2030	2031	2032	2033	2034	2035	2036	2037	2038	2039	2040	2041	2042
Financial analysis			Year	0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
Net income	USD	22,861,135	480,083,844	(227,474,483)	22,712,475	23,698,194	24,835,739	26,010,212	27,222,668	28,474,169	29,765,806	31,098,679	32,473,912	33,892,659	35,356,059	36,865,286	38,421,532	40,026,000	41,679,912	43,384,491	45,140,968	46,950,602	48,814,633	50,734,331
Cash flow	USD	33,693,254		(227,474,483)	34,086,199	35,071,918	36,209,463	37,383,936	38,596,392	39,847,893	41,139,530	42,472,403	43,847,636	45,266,383	46,729,783	48,239,011	49,795,257	51,399,724	53,053,636	54,758,215	56,514,692	58,324,326	60,188,358	62,108,055
Cumulative cash flow	USD			(227,474,483)	(193,388,284)	(158,316,366)	(122,106,902)	(84,722,966)	(46,126,574)	(6,278,681)	34,860,849	77,333,252	121,180,887	166,447,271	213,177,054	261,416,065	311,211,321	362,611,045	415,664,681	470,422,896	526,937,588	585,261,914	645,450,272	707,558,327
Present value of cash flow (at project discount ra	USD			**(227,474,483)**	**34,086,199**	**31,883,562**	**29,925,176**	**28,087,104**	**26,361,855**	**24,742,407**	**23,222,192**	**21,795,058**	**20,455,246**	**19,197,365**	**18,016,354**	**16,907,479**	**15,866,303**	**14,888,669**	**13,970,681**	**13,108,681**	**12,299,244**	**11,539,157**	**10,825,405**	**10,155,163**
Total cash flow	USD			**(227,474,483)**	**(193,388,284)**	**(161,504,722)**	**(131,579,546)**	**(103,492,441)**	**(77,130,586)**	**(52,388,179)**	**(29,165,987)**	**(7,370,929)**	**13,084,317**	**32,281,682**	**50,298,036**	**67,205,515**	**83,071,819**	**97,960,488**	**111,931,168**	**125,039,850**	**137,339,093**	**148,878,250**	**159,703,655**	**169,858,819**

NPV

		NPV
CAPEX	USD	206,794,985
OPEX	USD	131,992,992
Generation	kWh	7,346,381,761
NPV before tax	USD	**199,832,901**
NPV after tax	USD	169,858,819

LCOE

LCOE	USD/kWh	0.053

IRR

		IRR
IRR after tax	%	16.94%

		Payback period
Simple payback	Year	6
Payback period/Years to breakeven	Year	8.36

Result

Parameters	Unit	After tax
LCOE	USD/kWh	0.053
NPV	USD	169,858,819
IRR	%	16.94%
Payback period	Year	8.360344194
Annual average cash flow	USD	33,693,254
Total Life Cycle Cost	USD	387,186,003
ROI	%	43.87%

2. Landfill gas recovery

Input values

Generation

Landfilled waste	*ton/yr*	1,961,167
CH4 yield	*m3/yr*	111,261,826
Electricity generation	*kWh/yr*	197,893,066
Electricity generation efficiency	*%*	30.00%
Capacity Factor	*%*	85%

CAPEX

Incinerator Capacity	*MW*	23.00
Installing vertical gas extraction wells (C_v)	*USD*	12,750
Installing wellheads and pipes gathering (C_w)	*USD*	170,000
Installing knockout, blower, and flare system (C_{knock})	*USD*	526,988
Cost of engineering, permitting, and surveying (C_{eng})	*USD*	7,000
Cost of installation of reciprocating internal combustion engine	*USD*	31,000,000
Total Cost-CAPEX	*USD*	**31,716,738**

OPEX

Cost of landfill site operation and maintenance	*USD*	4,947,327
Cost of internal combustion engine	*USD*	31,100
Insurance (as % age CAPEX)	*%/yr*	0.00%
Land Lease	*USD/yr*	-
Fix OPEX cost	*USD*	4,978,427
Inflation rate	*%*	5.48%
Variable OPEX	*USD/MWh*	4.40

Economic

Period of Analysis	*Year*	15
Discount Rate	*%*	10%
Internal use of electricity	*%*	20%
Tax	*%*	20%
Bank loan (80%)	*USD*	25,373,390.73
Loan interest rate	*%*	8%
Down payment	*%*	20%
Depreciation rate with 20 years	*%*	5%
FIT from biomass plant in 2022	*USD/kWh*	0.095
Electricity sale in 2022	*USD/kWh*	-
Gate fee	*USD*	1.00

Calculations

Periods			Year	0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
Generation																				
Waste input	*ton*	2,133,264	31,998,958	-	1,615,958	1,678,172	1,742,781	1,809,878	1,879,558	1,951,922	2,027,070	2,105,112	2,186,159	2,270,326	2,357,734	2,448,506	2,542,774	2,640,671	2,742,337
LFG yield	*m3/yr*	111,261,826	1,668,927,395	-	64,904,264	73,373,795	81,023,955	88,042,945	94,584,054	100,772,412	106,710,421	112,482,175	118,157,026	123,792,463	129,436,447	135,129,292	140,905,188	146,793,451	152,819,507
Electricity generation	*kWh*	197,893,066	2,968,395,988	-	115,440,347	130,504,466	144,111,231	156,595,382	168,229,564	179,236,332	189,797,823	200,063,608	210,157,040	220,180,369	230,218,901	240,344,337	250,617,490	261,090,501	271,808,596
Internal use electricity	*kWh*	39,578,613	593,679,198	-	23,088,069	26,100,893	28,822,246	31,319,076	33,645,913	35,847,266	37,959,565	40,012,722	42,031,408	44,036,074	46,043,780	48,068,867	50,123,498	52,218,100	54,361,719
Actual electricity sale to national grid	*kWh*	158,314,453	2,374,716,790	-	**92,352,278**	**104,403,573**	**115,288,985**	**125,276,306**	**134,583,651**	**143,389,066**	**151,838,258**	**160,050,887**	**168,125,632**	**176,144,296**	**184,175,121**	**192,275,469**	**200,493,992**	**208,872,401**	**217,446,877**
PV of electricity sale to national grid				-	**92,352,278**	**94,912,339**	**95,280,153**	**94,121,943**	**91,922,445**	**89,033,328**	**85,708,738**	**82,131,412**	**78,431,848**	**74,702,376**	**71,007,482**	**67,391,379**	**63,883,565**	**60,502,894**	**57,260,559**
Cumulative electricity sale				-	**92,352,278**	**187,264,616**	**282,544,769**	**376,666,712**	**468,589,157**	**557,622,485**	**643,331,223**	**725,462,635**	**803,894,483**	**878,596,860**	**949,604,342**	**1,016,995,721**	**1,080,879,285**	**1,141,382,180**	**1,198,642,738**
Cash outflow																				
CAPEX																				
CAPEX (turn key)	*USD*	31,716,738	31,716,738	31,716,738	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
OPEX																				
Fix OPEX with inflation (5.48%)	*USD*	7,426,072	111,391,082	-	4,978,427	5,251,244	5,539,013	5,842,551	6,162,722	6,500,439	6,856,664	7,232,409	7,628,745	8,046,800	8,487,765	8,952,894	9,443,513	9,961,017	10,506,881
Variable OPEX	*USD*	916,590	13,748,847	-	507,938	605,687	668,838	726,778	780,774	831,857	880,874	928,519	975,364	1,021,884	1,068,474	1,115,467	1,163,146	1,211,752	1,261,496
Loan repayments	*USD*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Insurance	*USD*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Land Lease	*USD*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Total OPEX	*USD*	8,342,662	**125,139,929**	-	**5,486,364**	**5,856,931**	**6,207,850**	**6,569,328**	**6,943,496**	**7,332,297**	**7,737,538**	**8,160,928**	**8,604,109**	**9,068,683**	**9,556,238**	**10,068,361**	**10,606,658**	**11,172,769**	**11,768,377**
PV of cash outlow				31,716,738	**5,486,364**	**5,324,483**	**5,130,455**	**4,935,634**	**4,742,501**	**4,552,779**	**4,367,638**	**4,187,846**	**4,013,880**	**3,846,007**	**3,684,343**	**3,528,899**	**3,379,608**	**3,236,353**	**3,098,982**
Cumulative cash outflow				**31,716,738**	**37,203,103**	**42,527,586**	**47,658,040**	**52,593,674**	**57,336,175**	**61,888,955**	**66,256,593**	**70,444,439**	**74,458,320**	**78,304,327**	**81,988,670**	**85,517,569**	**88,897,178**	**92,133,531**	**95,232,512**
Cash inflow																				
Gate fee	*USD*		31,998,958	-	1,615,958	1,678,172	1,742,781	1,809,878	1,879,558	1,951,922	2,027,070	2,105,112	2,186,159	2,270,326	2,357,734	2,448,506	2,542,774	2,640,671	2,742,337
Electricity sale to national grid	*USD*	31,662,891	225,598,095	-	8,773,466	9,918,339	10,952,454	11,901,249	12,785,447	13,621,961	14,424,635	15,204,834	15,971,935	16,733,708	17,496,636	18,266,170	19,046,929	19,842,878	20,657,453
Benefit from internal use electricity	*USD*		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cash inflow	*USD*	**17,173,137**	**257,597,053**	-	**10,389,424**	**11,596,511**	**12,695,235**	**13,711,127**	**14,665,005**	**15,573,883**	**16,451,705**	**17,309,947**	**18,158,094**	**19,004,034**	**19,854,370**	**20,714,676**	**21,589,703**	**22,483,549**	**23,399,790**
PV of cash inflow				-	**10,389,424**	**10,542,283**	**10,491,929**	**10,301,373**	**10,016,396**	**9,670,156**	**9,286,559**	**8,882,740**	**8,470,885**	**8,059,566**	**7,654,719**	**7,260,368**	**6,879,145**	**6,512,683**	**6,161,896**
Cumulative cash inflow				-	**10,389,424**	**20,931,707**	**31,423,636**	**41,725,009**	**51,741,405**	**61,411,561**	**70,698,119**	**79,580,859**	**88,051,744**	**96,111,309**	**103,766,029**	**111,026,396**	**117,905,541**	**124,418,224**	**130,580,120**
Depreciation and tax																				
Net cash flow (before tax cash flows)	*USD*	8,830,475	**100,740,386**	**(31,716,738)**	**4,903,060**	**5,739,580**	**6,487,385**	**7,141,799**	**7,721,509**	**8,241,586**	**8,714,167**	**9,149,019**	**9,553,985**	**9,935,351**	**10,298,132**	**10,646,315**	**10,983,045**	**11,310,779**	**11,631,413**
Depreciation on assets	*USD*	2,114,449	31,716,738	-	2,114,449	2,114,449	2,114,449	2,114,449	2,114,449	2,114,449	2,114,449	2,114,449	2,114,449	2,114,449	2,114,449	2,114,449	2,114,449	2,114,449	2,114,449
Taxable	*USD*	6,716,026	100,740,386	-	2,788,610	3,625,131	4,372,935	5,027,349	5,607,060	6,127,137	6,599,718	7,034,570	7,439,536	7,820,902	8,183,683	8,531,866	8,868,596	9,196,330	9,516,964
Tax	*USD*	1,343,205	20,148,077	-	557,722	725,026	874,587	1,005,470	1,121,412	1,225,427	1,319,944	1,406,914	1,487,907	1,564,180	1,636,737	1,706,373	1,773,719	1,839,266	1,903,393
Total depreciation and tax			**51,864,975**	-	**2,672,171**	**2,839,475**	**2,989,036**	**3,119,919**	**3,235,861**	**3,339,877**	**3,434,393**	**3,521,363**	**3,602,356**	**3,678,630**	**3,751,186**	**3,820,822**	**3,888,168**	**3,953,715**	**4,017,842**
PV of depreciation and tax	*USD*		27,565,786	-	**2,672,171**	**2,581,341**	**2,470,278**	**2,344,041**	**2,210,137**	**2,073,801**	**1,938,625**	**1,807,016**	**1,680,526**	**1,560,098**	**1,446,245**	**1,339,175**	**1,238,890**	**1,145,250**	**1,058,023**
Cumulative depreciation and tax	*USD*			-	**15,708,184**	**33,333,186**	**52,641,857**	**73,394,884**	**95,405,313**	**118,527,590**	**142,648,885**	**167,682,215**	**193,560,976**	**220,234,585**	**247,665,017**	**275,824,018**	**304,690,885**	**334,250,676**	**364,492,759**

					2023	2024	2025	2026	2027	2028	2029	2030	2031	2032	2033	2034	2035	2036	2037
Financial analysis			Year	0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
Net income	*USD*	3,054,723	48,875,570	(31,716,738)	2,230,888	2,900,104	3,498,348	4,021,879	4,485,648	4,901,709	5,279,774	5,627,656	5,951,629	6,256,721	6,546,946	6,825,493	7,094,876	7,357,064	7,613,571
Cash flow	*USD*	5,037,019		(31,716,738)	4,345,338	5,014,554	5,612,797	6,136,329	6,600,097	7,016,159	7,394,223	7,742,105	8,066,078	8,371,171	8,661,396	8,939,942	9,209,326	9,471,513	9,728,020
Cumulative cash flow	*USD*			**(31,716,738)**	**(27,371,401)**	**(22,356,847)**	**(16,744,050)**	**(10,607,721)**	**(4,007,624)**	**3,008,535**	**10,402,758**	**18,144,863**	**26,210,941**	**34,582,112**	**43,243,508**	**52,183,450**	**61,392,775**	**70,864,289**	**80,592,309**
Present value of cash flow (at project discount ra	*USD*			**(31,716,738)**	**4,345,338**	**4,558,685**	**4,638,676**	**4,610,315**	**4,507,955**	**4,356,482**	**4,173,846**	**3,972,924**	**3,762,885**	**3,550,193**	**3,339,343**	**3,133,395**	**2,934,375**	**2,743,560**	**2,561,692**
Total cash flow	*USD*			**(31,716,738)**	**(27,371,401)**	**(22,812,716)**	**(18,174,040)**	**(13,563,725)**	**(9,055,770)**	**(4,699,288)**	**(525,441)**	**3,447,483**	**7,210,368**	**10,760,561**	**14,099,904**	**17,233,299**	**20,167,674**	**22,911,234**	**25,472,926**

NPV

		NPV
CAPEX	*USD*	28,833,399
OPEX	*USD*	52,492,375
Generation	*kWh*	1,362,094,021
NPV before tax	*USD*	**2,675,036**
NPV after tax	*USD*	25,472,926

LCOE

LCOE	*USD/kWh*	0.0699

IRR

		IRR
IRR before tax	*%*	11.83%
IRR after tax	*%*	18.53%

Payback period

		Payback period
Simple payback	*Year*	5
Payback period/Years to breakeven	*Year*	7.13

Result

Parameters	Unit	After tax
LCOE	USD/kWh	0.0699
NPV	USD	25,472,926
IRR	%	18.53%
Payback period	Year	7.132255578
Annual average cash flow	USD	5,037,019
Total Life Cycle Cost	*USD*	95,232,512
ROI	*%*	26.75%

3. Anaerobic digestion

Input

Generation

AD input waste	*ton/yr*	964,502
CH4 yield	*m3/yr*	95,215,126
Electricity generation	*kWh/yr*	238,347,263
Electricity generation efficiency	*%*	30%
Capacity Factor	*%*	85%

CAPEX

Internal combustion engines	*MW*	27.00
Cost-CAPEX	*USD/kW*	51827.082
Coefficient		0.55
Total CAPEX	*USD*	101,373,259

OPEX

O&M	*%*	3%
Variable operating and maintenance cost	*%*	5.48%
Insurance (as % age CAPEX)	*%/yr*	0.00%
Land Lease	*USD/yr*	-
Variable OPEX	*USD/MWh*	4.40

Economic

Period of Analysis	*Year*	20
Discount Rate	*%*	10%
Internal use of electricity	*%*	20%
Tax	*%*	20%
Bank loan (80%)	*USD*	-
Loan interest rate	*%*	8.00%
Down payment	*%*	20.00%
Depreciation rate with 20 years	*%*	5%
Electricity price in 2022	*USD*	0.095
Gate fee	*USD*	1.00

Calculations

				Year	0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
Periods																									
Generation																									
Waste input	Ton	964,502		-	657,940	683,270	709,576	736,895	765,265	794,728	825,325	857,100	890,098	924,367	959,955	996,913	1,035,294	1,075,153	1,116,546	1,159,534	1,204,175	1,250,536	1,298,682	1,348,681
LFG yield	m3/yr	95,215,126		-	64,951,474	67,452,096	70,049,013	72,745,894	75,546,608	78,455,142	81,475,656	84,612,458	87,870,026	91,253,033	94,766,277	98,414,774	102,203,735	106,138,570	110,224,907	114,468,569	118,875,602	123,452,318	128,205,228	133,141,134
Electricity generation	kWh	238,347,263		-	162,589,777	168,849,459	175,350,191	182,101,160	189,112,046	196,392,833	203,953,936	211,806,136	219,960,642	228,429,156	237,223,683	246,356,782	255,841,500	265,691,376	275,920,498	286,543,446	297,575,350	309,032,016	320,929,737	333,285,544
Internal use electricity	kWh	47,669,453		-	32,517,955	33,769,892	35,070,038	36,420,232	37,822,409	39,278,567	40,790,787	42,361,227	43,992,128	45,685,831	47,444,737	49,271,356	51,168,300	53,138,275	55,184,100	57,308,689	59,515,070	61,806,403	64,185,947	66,657,109
Actual electricity sale to national grid	kWh	190,677,811	**3,813,556,215**	-	**130,071,821**	**135,079,567**	**140,280,153**	**145,680,928**	**151,289,637**	**157,114,267**	**163,163,149**	**169,444,909**	**175,968,514**	**182,743,325**	**189,778,947**	**197,085,426**	**204,673,200**	**212,553,100**	**220,736,398**	**229,234,756**	**238,060,280**	**247,225,613**	**256,743,790**	**266,628,435**
PV of electricity sale to national grid			1,590,319,016	-	**130,071,821**	**122,799,607**	**115,934,010**	**109,452,237**	**103,332,858**	**97,555,598**	**92,101,344**	**86,952,031**	**82,090,610**	**77,501,009**	**73,167,999**	**69,077,239**	**65,215,189**	**61,569,062**	**58,126,793**	**54,876,978**	**51,808,853**	**48,912,269**	**46,177,627**	**43,595,880**
Cumulative electricity sale				-	**130,071,821**	**252,871,428**	**368,805,438**	**478,257,676**	**581,590,533**	**679,146,132**	**771,247,476**	**858,199,506**	**940,290,117**	**1,017,791,126**	**1,090,959,125**	**1,160,036,364**	**1,225,251,553**	**1,286,820,615**	**1,344,947,408**	**1,399,824,386**	**1,451,633,239**	**1,500,545,509**	**1,546,723,136**	**1,590,319,016**
Cash outflow																									
CAPEX																									
CAPEX (turn key)	USD	101,373,259	101,373,259	101,373,259	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
OPEX																									
Fix OPEX	USD	3,041,198	105,815,635	-	3,041,198	3,207,855	3,383,646	3,569,070	3,764,655	3,970,958	4,188,566	4,418,100	4,660,212	4,915,591	5,184,966	5,469,102	5,768,808	6,084,939	6,418,394	6,770,122	7,141,124	7,532,458	7,945,237	8,380,636
Variable OPEX	USD	1,048,728	22,084,761	-	715,395	783,651	813,821	845,153	877,692	911,483	946,575	983,018	1,020,864	1,060,167	1,100,984	1,143,371	1,187,391	1,233,106	1,280,580	1,329,883	1,381,083	1,434,255	1,489,473	1,546,818
Loan repayment		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Insurance	USD	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Land Lease	USD	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Total OPEX	USD	10,917,793	**229,273,655**	**101,373,259**	**3,756,593**	**3,991,506**	**4,197,467**	**4,414,223**	**4,642,346**	**4,882,440**	**5,135,141**	**5,401,117**	**5,681,075**	**5,975,758**	**6,285,949**	**6,612,473**	**6,956,200**	**7,318,045**	**7,698,974**	**8,100,004**	**8,522,207**	**8,966,713**	**9,434,710**	**9,927,454**
PV of cash outlow	USD		**152,593,851**	**101,373,259**	**3,756,593**	**3,628,642**	**3,468,981**	**3,316,471**	**3,170,785**	**3,031,611**	**2,898,653**	**2,771,627**	**2,650,264**	**2,534,305**	**2,423,505**	**2,317,631**	**2,216,460**	**2,119,777**	**2,027,380**	**1,939,077**	**1,854,681**	**1,774,016**	**1,696,916**	**1,623,218**
Cumulative cash outflow	USD			**101,373,259**	**105,129,851**	**108,758,493**	**112,227,474**	**115,543,945**	**118,714,730**	**121,746,342**	**124,644,995**	**127,416,622**	**130,066,886**	**132,601,190**	**135,024,696**	**137,342,327**	**139,558,787**	**141,678,564**	**143,705,944**	**145,645,021**	**147,499,702**	**149,273,718**	**150,970,633**	**152,593,851**
Cash inflow																									
Gate fee	USD		19,290,034	-	*657,940*	*683,270*	*709,576*	*736,895*	*765,265*	*794,728*	*825,325*	*857,100*	*890,098*	*924,367*	*959,955*	*996,913*	*1,035,294*	*1,075,153*	*1,116,546*	*1,159,534*	*1,204,175*	*1,250,536*	*1,298,682*	*1,348,681*
Income from electricity sale to grid	USD	4,766,945	362,287,840	-	*12,356,823*	*12,832,559*	*13,326,614*	*13,839,688*	*14,372,515*	*14,925,855*	*15,500,499*	*16,097,266*	*16,717,009*	*17,360,616*	*18,029,000*	*18,723,115*	*19,443,954*	*20,192,545*	*20,969,958*	*21,777,302*	*22,615,727*	*23,486,433*	*24,390,660*	*25,329,701*
Benefit from internal use electricity	USD		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cash inflow	USD	**17,181,096**	**381,577,874**		**13,014,763**	**13,515,829**	**14,036,191**	**14,576,583**	**15,137,781**	**15,720,583**	**16,325,824**	**16,954,366**	**17,607,107**	**18,284,983**	**18,988,955**	**19,720,029**	**20,479,248**	**21,267,698**	**22,086,504**	**22,936,835**	**23,819,902**	**24,736,970**	**25,689,342**	**26,678,383**
PV of cash inflow	USD		**159,124,585**	-	**13,014,763**	**12,287,117**	**11,600,158**	**10,951,603**	**10,339,308**	**9,761,245**	**9,215,502**	**8,700,271**	**8,213,845**	**7,754,618**	**7,321,064**	**6,911,750**	**6,525,320**	**6,160,494**	**5,816,067**	**5,490,896**	**5,183,905**	**4,894,078**	**4,620,454**	**4,362,129**
Cumulative cash inflow	USD			-	**13,014,763**	**25,301,880**	**36,902,038**	**47,853,640**	**58,192,948**	**67,954,194**	**77,169,696**	**85,869,966**	**94,083,812**	**101,838,429**	**109,159,493**	**116,071,243**	**122,596,563**	**128,757,057**	**134,573,124**	**140,064,020**	**145,247,925**	**150,142,002**	**154,762,456**	**159,124,585**
Depreciation and tax																									
Net cash flow (before tax cash flows)	USD	12,079,880	**253,677,478**	-	**9,258,170**	**9,524,323**	**9,838,724**	**10,162,360**	**10,495,434**	**10,838,143**	**11,190,683**	**11,553,249**	**11,926,032**	**12,309,224**	**12,703,006**	**13,107,556**	**13,523,049**	**13,949,653**	**14,387,530**	**14,836,831**	**15,297,695**	**15,770,257**	**16,254,632**	**16,750,929**
Depreciation on assets	USD	4,827,298	101,373,259	-	*5,068,663*	*5,068,663*	*5,068,663*	*5,068,663*	*5,068,663*	*5,068,663*	*5,068,663*	*5,068,663*	*5,068,663*	*5,068,663*	*5,068,663*	*5,068,663*	*5,068,663*	*5,068,663*	*5,068,663*	*5,068,663*	*5,068,663*	*5,068,663*	*5,068,663*	*5,068,663*	*5,068,663*
Taxable	USD	7,252,582	152,304,220	-	*4,189,507*	*4,455,660*	*4,770,061*	*5,093,697*	*5,426,771*	*5,769,480*	*6,122,020*	*6,484,586*	*6,857,369*	*7,240,562*	*7,634,343*	*8,038,893*	*8,454,386*	*8,880,990*	*9,318,867*	*9,768,168*	*10,229,032*	*10,701,594*	*11,185,969*	*11,682,266*
Tax	USD	1,450,516	30,460,844	-	*837,901*	*891,132*	*954,012*	*1,018,739*	*1,085,354*	*1,153,896*	*1,224,404*	*1,296,917*	*1,371,474*	*1,448,112*	*1,526,869*	*1,607,779*	*1,690,877*	*1,776,198*	*1,863,773*	*1,953,634*	*2,045,806*	*2,140,319*	*2,237,194*	*2,336,453*
Total depreciation and tax	USD	6,277,814	**131,834,102**	-	**5,906,564**	**5,959,795**	**6,022,675**	**6,087,402**	**6,154,017**	**6,222,559**	**6,293,067**	**6,365,580**	**6,440,137**	**6,516,775**	**6,595,532**	**6,676,441**	**6,759,540**	**6,844,861**	**6,932,436**	**7,022,297**	**7,114,469**	**7,208,982**	**7,305,857**	**7,405,116**
PV of depreciation and tax	USD		**59,554,897**	-	**5,906,564**	**5,417,995**	**4,977,417**	**4,573,555**	**4,203,277**	**3,863,720**	**3,552,272**	**3,266,549**	**3,004,371**	**2,763,749**	**2,542,863**	**2,340,052**	**2,153,798**	**1,982,712**	**1,825,527**	**1,681,082**	**1,548,316**	**1,426,259**	**1,314,023**	**1,210,796**
Cumulative depreciation and tax	USD			-	**5,906,564**	**11,324,560**	**16,301,977**	**20,875,533**	**25,078,809**	**28,942,529**	**32,494,801**	**35,761,350**	**38,765,721**	**41,529,470**	**44,072,333**	**46,412,385**	**48,566,183**	**50,548,895**	**52,374,422**	**54,055,504**	**55,603,820**	**57,030,079**	**58,344,101**	**59,554,897**
Investment analysis					**0**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**
Net income	USD				(101,373,259)	3,351,606	3,564,528	3,816,049	4,074,958	4,341,417	4,615,584	4,897,616	5,187,669	5,485,895	5,792,449	6,107,474	6,431,114	6,763,509	7,104,792	7,455,094	7,814,535	8,183,225	8,561,275	8,948,775	9,345,813
Cash flow	USD	5,802,066	121,843,376		**(101,373,259)**	**8,420,269**	**8,633,191**	**8,884,711**	**9,143,621**	**9,410,080**	**9,684,247**	**9,966,279**	**10,256,332**	**10,554,558**	**10,861,112**	**11,176,137**	**11,499,777**	**11,832,172**	**12,173,455**	**12,523,757**	**12,883,197**	**13,251,888**	**13,629,938**	**14,017,438**	**14,414,475**
Cumulative cash flow	USD		4,812,229		**(101,373,259)**	**(92,952,990)**	**(84,319,799)**	**(75,435,087)**	**(66,291,467)**	**(56,881,387)**	**(47,197,140)**	**(37,230,861)**	**(26,974,529)**	**(16,419,971)**	**(5,558,859)**	**5,617,278**	**17,117,055**	**28,949,227**	**41,122,682**	**53,646,438**	**66,529,636**	**79,781,524**	**93,411,462**	**107,428,900**	**121,843,376**
Present value of cash flow (at project discount ra	USD		(5,556,540)		**(101,373,259)**	**8,420,269**	**7,848,356**	**7,342,737**	**6,869,738**	**6,427,211**	**6,013,155**	**5,625,705**	**5,263,120**	**4,923,779**	**4,606,172**	**4,308,885**	**4,030,602**	**3,770,095**	**3,526,216**	**3,297,897**	**3,084,135**	**2,883,997**	**2,696,611**	**2,521,159**	**2,356,882**
Toal cash flow/Cumulative PV of cashflow	USD				**(101,373,259)**	**(92,952,990)**	**(85,104,634)**	**(77,761,898)**	**(70,892,160)**	**(64,464,949)**	**(58,451,794)**	**(52,826,089)**	**(47,562,969)**	**(42,639,190)**	**(38,033,018)**	**(33,724,133)**	**(29,693,532)**	**(25,923,437)**	**(22,397,221)**	**(19,099,324)**	**(16,015,189)**	**(13,131,192)**	**(10,434,582)**	**(7,913,422)**	**(5,556,540)**

NPV

		NPV
CAPEX	USD	92,157,508
OPEX	USD	134,488,576
Generation	kWh	1,642,891,545
NPV after tax	USD	**(5,556,540)**

LCOE

LCOE	USD/kWh	0.093

IRR

		IRR
IRR after tax	%	8.08%

Payback period

		Payback period
Simple payback	Year	10
Payback period/Years to breakeven	Year	> 20

Result

Parameters	Unit	Value
LCOE	USD/kWh	0.093
NPV	USD	(5,556,540)
IRR	%	8.08%
Payback period	Year	> 20
Annual average cash flow	USD	5,802,066
Total Life Cycle Cost	USD	152,593,851
ROI	%	-3.64%

Appendix 4. GHG emissions from WTE technology

1 Emissions from coal power plant

Technology for coal-fired power plant (CFPP)	Bituminous Subcritical	Sub-Bituminous Supercritical	Bituminous Supercritical	Anthracite Supercritical	Average (kgCO2-eq/kWh)
ASEAN Center for Energy 2021	871.43	922.47	824.32	1059.12	0.919335

2 LFG recovery

	2023	2024	2025	2026	2027	2028	2029	2030	2031	2032	2033	2034	2035	2036	2037	2038	2039	2040	2041	2042
E_{LFG}(ton/year)						9,743	11,014	12,162	13,216	14,198	15,127	16,018	16,885	17,736	18,582	19,429	20,284	21,151	22,035	22,939
GHG_{LFG}(tCO2-eq/year)						243,567	275,351	304,059	330,400	354,946	378,170	400,453	422,113	443,409	464,557	485,737	507,101	528,776	550,873	573,487
Emission avoidance (tCO2-eq/year)						106,128	119,977	132,486	143,964	154,659	164,778	174,488	183,925	193,205	202,420	211,648	220,957	230,401	240,030	249,883
Net GHG emissions (tCO2-eq/year)						**137,439**	**155,373**	**171,573**	**186,436**	**200,287**	**213,391**	**225,965**	**238,187**	**250,204**	**262,138**	**274,089**	**286,144**	**298,375**	**310,844**	**323,604**

3 Incineration 0.52 tCO2-eq/tMSW

	2023	2024	2025	2026	2027	2028	2029	2030	2031	2032	2033	2034	2035	2036	2037	2038	2039	2040	2041	2042
GHG emissions from incineration	665,479	691,100	717,707	745,339	774,034	803,834	834,782	866,921	900,297	934,959	970,955	1,008,336	1,047,157	1,087,473	1,129,341	1,172,820	1,217,974	1,264,866	1,313,563	1,364,135
Emission avoidance (tCO2-eq/year)	607,630	631,023	655,318	680,547	706,749	733,958	762,216	791,561	822,036	853,684	886,551	920,683	956,129	992,940	1,031,169	1,070,869	1,112,097	1,154,913	1,199,377	1,245,553
Net emission (tCO2-eq/year)	**57,849**	**60,076**	**62,389**	**64,791**	**67,286**	**69,876**	**72,566**	**75,360**	**78,262**	**81,275**	**84,404**	**87,653**	**91,028**	**94,532**	**98,172**	**101,952**	**105,877**	**109,953**	**114,186**	**118,582**

4 AD

	2023	2024	2025	2026	2027	2028	2029	2030	2031	2032	2033	2034	2035	2036	2037	2038	2039	2040	2041	2042
CH4 emission (tCH4/year)	2,166	2,250	2,336	2,426	2,519	2,616	2,717	2,822	2,930	3,043	3,160	3,282	3,408	3,540	3,676	3,818	3,965	4,117	4,276	4,440
GHG emission (tCO2-eq/year)	54,153	56,238	58,403	60,652	62,987	65,412	67,930	70,546	73,262	76,082	79,011	82,053	85,212	88,493	91,900	95,438	99,113	102,928	106,891	111,006
GHG avoidance (tCO-eq/year)	149,474	155,229	161,206	167,412	173,857	180,551	187,502	194,721	202,218	210,003	218,088	226,484	235,204	244,259	253,663	263,429	273,571	284,104	295,042	306,401
Net GHG emission (tCO2-eq/year)	**(95,321)**	**(98,991)**	**(102,802)**	**(106,760)**	**(110,870)**	**(115,139)**	**(119,572)**	**(124,175)**	**(128,956)**	**(133,921)**	**(139,077)**	**(144,431)**	**(149,992)**	**(155,766)**	**(161,763)**	**(167,991)**	**(174,459)**	**(181,176)**	**(188,151)**	**(195,395)**