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Environmental and economic impact assessment of single-use laboratory plastic waste: A case study

Yi Huan Tan [a], Hwai Chyuan Ong [b,c], Raja Affendi Raja Ali [a,d], Lai Ti Gew [a,*]

[a] Sir Jeffrey Cheah Sunway Medical School, Faculty of Medical and Life Sciences, Sunway University, Petaling Jaya, Selangor 47500, Malaysia
[b] School of Engineering, Faculty of Engineering and Technology, Sunway University, Sunway City, Malaysia
[c] School of Civil and Environmental Engineering, Faculty of Engineering and Information Technology, University of Technology Sydney, Sydney, Australia
[d] Gut Research Group, Faculty of Medicine, Universiti Kebangsaan Malaysia, Malaysia

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ABSTRACT

Research labs rely on disposable plastics for sterility, safety, and affordability, but their environmental and economic impact has been largely overlooked. While the impact of single-use plastics is well-known, laboratory waste poses an additional challenge due to necessary treatment processes. Hence, this study investigated both the environmental and economic impact associated with single-use plastic waste management in a university laboratory, serving as a case study for Southeast Asia. A retrospective analysis was conducted on laboratory plastic waste management between 2014 and 2023 at a Malaysian university. Three waste management methods, i.e. incineration, landfilling with microwave pre-treatment, and landfilling with ozone pre-treatment, were implemented at consecutive intervals. Environmental impact was assessed by quantifying the GHG emissions in terms of carbon dioxide equivalent (CO_2eq) emissions, while economic assessment was evaluated based on invoiced costs. A total of 29,180.11 kg of single-use plastic waste resulted in 46,420.08 kg CO_2eq emissions and a cumulative disposal cost of RM 84,890.49. Among the three methods, landfilling with ozone pre-treatment demonstrated the lowest environmental impact (1.55 kg CO_2eq/kg) and cost (RM 2.79/kg). Direct emission from the end-of-life stage was the main source of GHG emissions, while disposal fees represented the largest portion of total costs. In light of these findings, it is crucial for universities and research institutions to recognise the environmental and economic impact of laboratory single-use plastics. This case study will serve as a foundation for advancing pre-treatment technologies and future end-of-life solutions.

1. Introduction

According to Malaysian Qualification Agency (MQA), 67 academic institutions in Malaysia, including both public or private colleges and universities, offer courses in biological and related sciences. These fields, such as Biology, Medical Microbiology, Biomedical Science, Biotechnology, Biochemistry, and Medical Bioscience, heavily rely on laboratory experiments (Malaysian Agency Qualifications, 2024). The fast-paced environment of the teaching and research laboratories in these institutions undeniably contributes to a reliance on convenient, single-use plastics.

Plastic is one of the main materials to produce laboratory consumables due to its noncytotoxic characteristics (Bedell, 2018). Lab-grade plastic consumables require high level of purity to ensure the accuracy and reliability of laboratory experiments results. Hence, the laboratory plastic consumables are manufactured with minimal amount of additives, including plasticisers, fillers, stabilisers, and antioxidant. This optimisation decreases the risk of leaching of these additives into laboratory solutions (Bedell, 2018). In addition, the laboratory consumables are often pre-sterilised to prevent contamination and ensure immediate use. These consumables not only ensure sterility, but also save time and effort from additional decontamination procedures prior to use (Tan et al., 2022). They are often designed for one-time use and can be easily discarded after use to avoid cross-contamination (Aliouche, 2022).

Furthermore, price is also one of the factors to be considered for laboratory consumables procurement. Therefore, plastic laboratory consumables are preferentially sourced because of their lower price compared to glass laboratory consumables (Freeland et al., 2022). Additionally, being shatterproof and durable making minimal packaging

* Correspondence to: Department of Biological Sciences, School of Medical and Life Sciences, Sunway University, Petaling Jaya, Selangor 47500, Malaysia.
E-mail addresses: onghc@sunway.edu.my (H.C. Ong), affendi@sunway.edu.my (R.A. Raja Ali), janeg@sunway.edu.my (L.T. Gew).

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required to transport plastic laboratory consumables safely and thus reducing the waste in the laboratory (Bedell, 2018).

Research and teaching laboratories, especially in the life sciences and biomedical fields, utilise a vast variety of single-use plastic consumables such as pipette tips, falcon tubes, microcentrifuge tubes, culture flasks, well plates, petri dishes, and polymerase chain reaction (PCR) tubes (Banks et al., 2020). The commonly used plastics polymers to produce these laboratory consumables are polypropylene (PP) for tubes and pipette tips, polystyrene (PS) for culture consumables, and high-density polyethylene (HDPE) for caps.

The prevalence of single-use plastic in scientific research varies depending on the intensity of teaching and research activities (Table 1). The School of Pharmaceutical Sciences at Universiti Sains Malaysia reported that a postgraduate student produces approximately 10 kg of plastic waste in two weeks during periods of high productivity. Consequently, in a year, 240 kg to 260 kg of such laboratory plastic waste could have been produced by just a single student (Tan et al., 2022). Consistently, Howes (2019) also reported that a molecular biology postgraduate student may generate 230 g of plastic waste from consumables such as pipette tips and tubes daily. This adds up to 60 kg of plastics waste from the laboratory in a year (Howes, 2019). Furthermore, a study in University of Edinburgh also revealed that seven microbiologists may use nearly 2000 units of single-use plastic loops and spreaders, and 2200 Falcon and universal tubes, which is equivalent to 97 kg of waste, in a month (Alves et al., 2021). Moreover, a study of a bioscience department in the United Kingdom found that 280 scientists generated approximately 267 tonnes of plastic waste in 2014. By projecting this figure to the number of research laboratories worldwide, it is estimated that a total 5.5 million tonnes laboratory plastic waste could have been generated in that year, contributing to 1.8 % of global plastic waste (Urbina et al., 2015). While the contribution of this plastic waste may seems insignificant, in fact, it is equivalent to 83 % of the plastic recycled worldwide in 2012 (Urbina et al., 2015).

The widespread use of laboratory plastic and its high disposal rate pose a significant negative impact to marine life, marine ecosystems, and the environment (OECD, 2022). The laboratory plastic waste conventionally disposed of via landfilling and incineration (Freeland et al., 2022). Choice of disposal methods can significantly affect the carbon emission. Incineration has been identified as a major contributor, accounting for over half of the carbon emissions associated with protective wear and plastics in clinical settings (Ragazzi et al., 2023). By opting for recycling instead of incineration for recyclable plastic consumables, such as PP, PC, and PET, it is possible to reduce lifetime emissions up to 74 % (Ragazzi et al., 2023). Besides, laboratory consumables exposed to infectious substances must be decontaminated and sterilised before disposal (Freeland et al., 2022). This additional step further exacerbates the environmental burden of laboratory plastic waste. For example, autoclaving of infectious waste for decontamination purpose significantly increased the carbon emission by 2.3–4.5-fold (Rizan et al., 2021).

The literatures above clearly elucidated that the scientific research laboratories, particularly in the life sciences and biomedical fields, are significant contributors to plastic waste, but there is limited data to support this (Tan et al., 2022). Laboratory plastic waste management inevitably results in environmental impacts, primarily through the generation of GHG emissions, leading to global warming. These emissions arise from various stages of waste management, including transportation, pre-treatment, and final disposal. However, data specific to laboratory plastic waste generation and its associated environmental effects has been scarce and largely outdated. The estimate of waste generation rates in laboratory settings made by Urbina et al. (2015) has remained one of the key references in the field. While most studies on plastic pollution focus on household and municipal waste, the contribution of the scientific community and laboratories has been relatively overlooked (Tan et al., 2022). Despite a few universities includes plastics in the university carbon footprint assessment, only general procurement purchases data was assessed in which no detailed information on laboratory plastic is provided (Larsen et al., 2013; Ozawa-Meida et al., 2013; Thurston and Eckelman, 2011; Townsend and Barrett, 2015). There is only a study conducted by Farley and Nicolet (2022) to compare the carbon emission between single-use and reusable laboratory consumables. The findings revealed that single-use plastic tubes, which is incinerated at the end of life, generated 11.3 times more carbon dioxide equivalent (CO_2eq) than the reusable glass alternative, and up to 10.6 times more CO_2eq than reusing them (Farley and Nicolet, 2022). The findings indicate that the choice of end-of-life management and disposal methods can significantly influence overall emissions. This is demonstrated in a similar case in a hospital setting (Rizan et al., 2021). Recycling of hospital waste has the lowest environmental impact with 21–65 kg CO_2eq per tonne of waste, followed by low-temperature incineration (172–249 kg CO_2eq) and high-temperature incineration (1074 kg CO_2eq) (Rizan et al., 2021). These findings clearly highlight the importance of assessing emissions from different waste treatment and disposal methods. The variation in laboratory plastic waste generation across different types of laboratory and research activities highlights the need for customised studies to address this issue effectively.

Yet, while the environmental impact of plastic waste is widely discussed, there is limited attention to its economic burden, particularly in institutional contexts where sustainability budgeting is increasingly relevant. Waste disposal methods greatly influence both environmental and financial outcomes. Incineration, while effective in sterilization, is costly and carbon-intensive (Bernardo et al., 2016). Conversely, pre-treatment technologies such as microwave or ozone systems may reduce both GHG emissions and operational costs over time (Khannam et al., 2021). These choices not only determine the carbon footprint but also impose significant recurring financial commitments on laboratories and their institutions.

This study aims to bridge that gap by conducting a real-world environmental and economic assessment of laboratory plastic waste management to establish a baseline. By quantifying both the environmental and economic impacts of laboratory plastic waste management, this study provides critical insights into the sustainability performance of commonly used disposal methods. The findings offer a robust foundation for identifying and implementing more cost-effective and environmentally responsible alternatives, informing policy recommendations, supporting the transition to a circular economy for laboratory plastics, and contributing to Global Sustainability Goals (SDG). It is worth noting that methodology employed in this study provides a replicable framework for assessing the GHG emissions and cost implications associated with laboratory plastic waste. This approach can be readily adapted to laboratories worldwide, regardless of geographic location or institutional type.

Table 1
Estimation of laboratory plastic waste generation by a single lab user.

Year	Country	Estimated average amount of waste generated by a lab user			
		kg/day	kg/week[a]	kg/year[b]	Reference
2015	UK (England)	3.67	18.35	954.00	(Urbina et al., 2015)
2019	Germany	0.23	1.15	60.00	(Howes, 2019)
2021	UK (Scotland)	0.69	3.45	180.00	(Alves et al., 2021)
2022	Malaysia	1.00	5.00	260.00	(Tan et al., 2022)

[a] 5 working days = 1 week
[b] 52 weeks = 1 year

2. Methodology

The objective of this study is to quantify both the greenhouse gas (GHG) emissions and the financial costs associated with the management of single-use plastic waste generated in a university laboratory setting. The university selected is one of the top private universities located in Kuala Lumpur, Malaysia. This study aligns seamlessly with the mission of the university where their effort lies within becoming a sustainable campus through the incorporation of sustainable practices into all facets of the university experience, including research, work practices and resource consumption.

The laboratory was established in 2014. Across the 10 years of establishment, it has served over 750 undergraduate and postgraduate students of biological sciences related courses, including Biomedicine, Medical Sciences, and Biology. The high number of students utilising the laboratory for various biological sciences courses increases the potential for plastic waste generation. To accommodate high volume of students and research activities, there are a total of 11 laboratories dedicated to teaching and research across diverse biological sciences related fields such as microbiology, virology, immunology, molecular biology, neurobiology, etc. The diverse range of research areas often involves extensive use of plastic consumables, potentially contributing to a significant plastic waste footprint. Hence, understanding the environmental impact of this plastic waste is crucial.

2.1. Greenhouse gas (GHG) assessment

GHG Assessment aims to evaluate the GHG emissions generated from the management of single-use laboratory plastic waste between 2014 and 2023. This study scrutinises the end-life life cycle, including collection, transportation, and final disposal at designated facilities. As illustrated in Fig. 1, the laboratory employed three waste disposal methods sequentially during this period.:

a) Method A, incineration (January 2014 to March 2015);
b) Method B, landfilling with microwave pre-treatment (April 2015 to March 2018); and
c) Method C, landfilling with ozone pre-treatment (April 2018 to December 2023).

The pre-treatments in Methods B and C ensures the waste are inert prior landfilling to avoid potential contamination. Landfill site in both Methods B and C is integrated with the pre-treatment plant, hence, no additional transportation distance is reported.

The GHG emissions associated with the end-life management of laboratory plastic waste were estimated using the activity data (e.g., electricity consumption, milage) and the corresponding emission factor and Global Warming Potential (GWP), as described in Eq. (1) (IUCN, 2021). Emission factor is the average emission rate of a particular GHG per units of activity. Emission factor allows the conversion of the activity data into GHG emission in terms of carbon dioxide equivalent (CO_2eq). Different activity emits different kind of GHG such as methane (CH_4) and nitrous oxide (N_2O), and each kind of GHG has varying capacity to trap heat in the atmosphere (IUCN, 2021). Hence, GWP is used to standardised these emissions into a common unit, CO_2eq, to allow for comparison and summation of different GHG emissions. The GWP of the GHG in CO_2eq is considered over a period of 100 years (Table S1) (IPCC, 2015).

$$GHG\ emission = Activity\ data \times Emission\ factor \times GWP \tag{1}$$

Fundamentally, the GHG emission of the laboratory plastic waste management at the laboratory is the sum of the GHG emissions from the transportation of the plastic waste from the laboratory to the treatment plant (E_T), operation of the treatment plant (E_{OP}), and the end-life at the incinerator or landfill (E_{EOL}). The calculation of the net GHG emission of the laboratory plastic waste is summarised in Eq. (2). The assumptions made in this GHG assessment are as follows:

1. Diesel rigid trucks that are averagely laden were used to collect and transport the laboratory plastic waste.
2. Electricity generation mix in Malaysia for the years 2015 and 2016 is the same as year 2017 due to the availability of data.
3. Electricity generation mix in Malaysia for the year 2023 are the same as year 2022 due to the availability of data.
4. For microwave and ozone treatments that are claimed as zero-emission technologies, only electricity consumption during machine startup was considered in GHG assessment (Clean Waste Systems, n.d.-a; Tri Ecoedge Sdn Bhd, 2017).

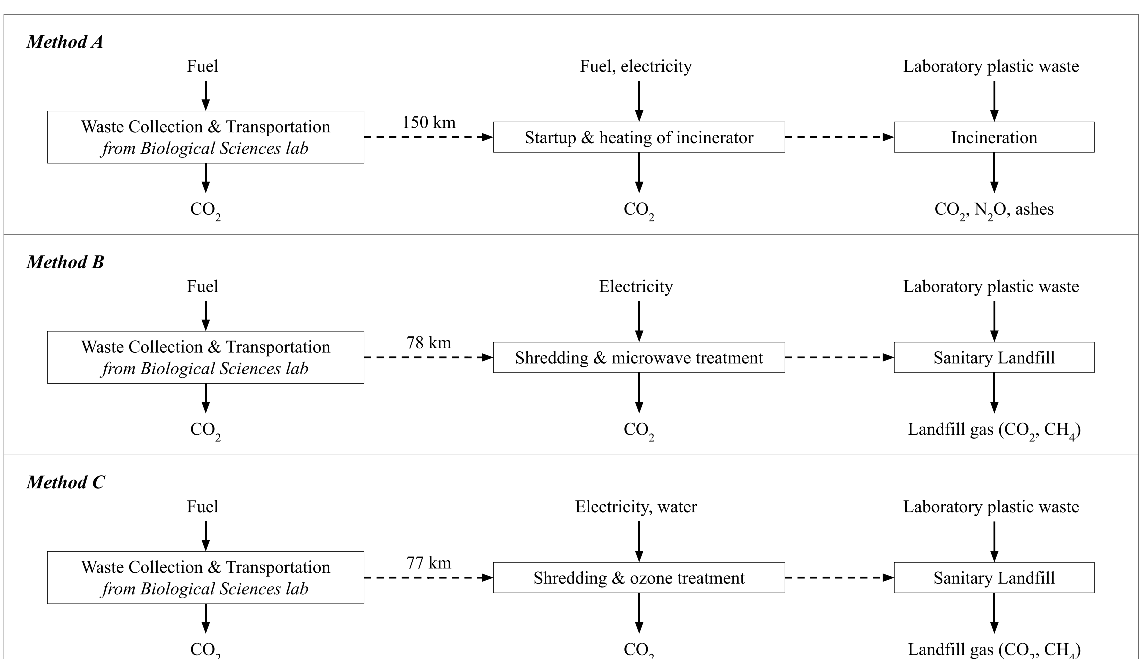

Fig. 1. Waste management methods of laboratory plastic waste generated from the laboratory from 2014 to 2023.

5. The transportation and environmental impact of ashes are not considered as ash content in plastic waste is less than 5 % (Chang, 2023; Chen et al., 2019; Zabaniotou and Vaskalis, 2023)
6. An on-site landfill is located adjacent to the microwave and ozone treatment plant. Hence, no further transportation needed from treatment plant to landfill.
7. Landfill gas calculations focus on CH_4 as it is the primary greenhouse gas from anaerobic degradation. Biogenic CO_2 is excluded, as it is considered carbon-neutral and accounted for under the Agriculture, Forestry, and Other Land Use (AFOLU) sector per IPCC guidelines (Eggleston et al., 2006).
8. GHG savings initiatives during the period of study is not reported.

$$GHG\ emission = E_{T,i} + E_{OP,i} + E_{EOL,i} \tag{2}$$

2.1.1. Transportation emission, E_T

The activity data is the transportation of the laboratory plastic wastes (PW) over a certain distance (d_i) by the waste collection truck to i treatment plant based on different methods, as denoted as T and outlined in Eq. (3). The transportation distance (d_i) from the laboratories to the incineration plant in Method A (A), microwave treatment plant in Method B (B), and ozone treatment plant in Method C (C) are 150 km, 78 km, and 77 km, respectively. During the transportation of the laboratory plastic wastes, the engine of the waste collection truck operates under varying weight of transported wastes. As the weight of transported wastes varies, the tension on the engine changes correspondingly. Heavier loads increase engine stress, leading to higher fuel consumption, while lighter loads reduce both (Baral et al., 2024).

$$T = PW \times d_i \tag{3}$$

The emission factor of goods transported in an averagely laden diesel rigid truck is used (Department for Energy Security and Net Zero, 2023). The total GHG emission from transportation of waste was concluded in Eq. (4).

$$E_T = T \times EF_T \tag{4}$$

$$E_{OP,i} = E_{energy,i} + E_{water,i} \tag{5}$$

2.1.2. Treatment plant operation emission, E_{OP}

To calculate the emission generated by the operation of i treatment plant ($E_{OP,i}$), three treatment facilities were considered: incineration plant in Method A (A), microwave treatment plant in Method B (B), and ozone treatment plant Method C (C). Two types of activity data were taken into account for operation emission, i.e., energy and water:

The type of energy consumption varies based on the energy demand of the treatment plant. Incineration plant operates using both diesel and electricity while the microwave and ozone treatment plants operate by only electricity. The information of the energy demand (e_i) was obtained from the technical data published on the official website of the i treatment plant provider. The total energy consumption ($energy_i$) was estimated by the operating hours (t_i) of i treatment plant and its corresponding energy demand (e_i), as summarised in Eq. (6). Operating hours (t_i) was estimated based on the minimum processing capacity of the i treatment plant (C_i) and the laboratory plastic waste being processed (PW), as described in Eq. (7). The total emission of the operation of i treatment plant was estimated based on the emission factor of the energy source of i treatment plant (EF_{energy}), as summarised in Eq. (8).

$$energy_i = t_i \times e_i \tag{6}$$

$$t_i = PW / C_i \tag{7}$$

$$E_{energy,i} = energy_i \times EF_{energy} \tag{8}$$

$$water_i = t_i \times w_i \tag{9}$$

$$E_{water,i} = water_i \times (EF_{water\ supply} + EF_{water\ treatment}) \tag{10}$$

Besides energy, water is also considered for the treatment plant operations. While incineration and microwave plants do not require water, the ozone treatment plant utilises water to generate humidified ozone for optimal decontamination. This process is claimed to produce no emissions, fumes, chemicals, steam, or heat. The water consumption ($water_i$) was estimated based on the by the operating hours (t_i) of the ozone treatment plant and its water demand (w_i) as published on the official website of the treatment plant provider.

The emission factor of both water supply and water treatment were taken into account to estimate the total GHG emission of water consumption.

2.1.3. End-of-life emission, $E_{EOL,i}$

2.1.3.1. Incineration emission, $E_{EOL,A}$. The primary greenhouse gases emitted during waste incineration are CO_2, CH_4 and N_2O. As the default oxidation factor of 1 for plastic waste incineration indicates complete combustion, consequently, CH_4 emissions are negligible under ideal conditions. As a result, the total incineration emissions of the laboratory plastic waste can be primarily attributed to CO_2 and N_2O (Eq. (11)) (Eggleston et al., 2006).

$$E_{EOL,A} = E_{CO_2,A} + E_{N_2O,A} \tag{11}$$

$$E_{CO_2,A} = PW \times dm \times CF \times FCF \times OF \times \frac{44}{12} \tag{12}$$

$$E_{N_2O,A} = PW \times EF_{N_2O,A} \times 10^{-6} \times GWP_{N_2O} \tag{13}$$

The estimation of CO_2 and N_2O emissions from incineration in the present study follows the IPCC guidelines (Eggleston et al., 2006). The estimation CO_2 emissions from incineration of the laboratory plastic waste is based on an estimate of the fossil carbon content (FCF) in the plastic waste incinerated, multiplied by the oxidation factor (OF), and converting the product to CO_2 by the conversion factor $\frac{44}{12}$. The activity data are the laboratory plastic waste inputs into the incinerator (PW), and the emission factors are based on the calculation in Eq. (12). IPCC guideline clearly outlines the default value of the dry matter content (dm), the total carbon content (CF), the fossil carbon fraction (FCF) and the oxidation factor (OF) of plastic waste (Eggleston et al., 2006).

The calculation of N_2O emissions is also based on the laboratory plastic waste input to the incinerator (PW) and a default emission factor provided by IPCC guideline (Eggleston et al., 2006). It is crucial to maintain the consistency of the amount and composition of the laboratory plastic waste with the activity data used for the calculation of CO_2 emissions from incineration in Eq. (12). The N_2O emission is converted into CO_2eq using the corresponding GWP value as reported by IPCC (IPCC, 2015).

2.1.3.2. Landfilling emission, $E_{EOL,L}$. After the pre-treatment at microwave and ozone treatment plant in Methods B and C, respectively, the waste was sent to landfill for final disposal. Landfilling of laboratory plastic waste resulted in the emission of CO_2 and CH_4. The activity is the amount of the laboratory plastic waste being landfilled (PW) while the emission factor of landfilling of plastic waste (EF_L) was calculated using IPCC default method (Penman et al., 2000). The estimation of the GHG emitted from landfilling of laboratory plastic waste was calculated using Eq. (14).

$$E_{EOL,BC} = \left[\left(PW \times MCF \times DOC \times DOC_f \times F \times \frac{16}{12}\right) - R\right] \bullet (1 - OX) \bullet GWP_{CH_4} \tag{14}$$

where MCF = methane correction factor, DOC = degradable organic carbon, DOC_f = fraction of DOC that can decompose, F = fraction of CH4 in landfill gas, R = recovered CH4, OX = oxidation factor, and GWP_{CH_4} = global warming potential of CH4.

2.2. Inventory data of GHG assessment

2.2.1. Method A: incineration (Jan 2014 to Mar 2015)

In Method A, the plastic waste from the laboratory were incinerated at an incinerator plant located at a distance of 150 km away. Waste collection was performed monthly between January 2014 and March 2015, with a total of 681.58 kg of laboratory plastic waste being collected and incinerated. The incinerator plant used a stepped hearth incineration system with a processing capacity of 600 kg of waste per hour. This incinerator plant uses 70 kWh of electricity to power the incinerator, with diesel fuel serving as an auxiliary energy source at a consumption rate of 40 L per hour. It was assumed that the incinerator allows complete combustion of the laboratory plastic waste, resulting in an oxidation factor (*OF*) is 1 (Eggleston et al., 2006). The default value of dry matter (*dm*) of the waste, total carbon content (*CF*) in the dry matter, fossil carbon content in the *CF* (*FCF*), and oxidation factor (*OF*) of laboratory plastic waste were obtained from the IPCC guideline (Eggleston et al., 2006). Inventory data related to the laboratory plastic waste management in Method A was presented in Table S2.

2.2.2. Method B: landfilling with microwave pre-treatment (Apr 2015 to Dec 2016)

Method B involves a waste management method that combines microwave pre-treatment with subsequent landfilling. The microwave treatment plant and the associated landfill was located 78 km from the laboratory. This disposal method was implemented for a period of 21 months, from April 2015 to December 2016, with monthly waste collection and disposal of a total of 2115.33 kg of laboratory plastic waste. The microwave pre-treatment aims to sterilise the waste prior to final disposal at landfill, thereby eliminating potential contamination risks. The treatment plant provider claimed a proven disinfection rate of 99.99 % (Ecosteryl, 2021). The microwave plant operated on electricity with power consumption of 60 kWh and required no water. It was designed to process at least 250 kg of waste per hour. The electricity consumption was calculated separately in each year due to the varying electricity generation mix and corresponding emission factors. Notably, the manufacturer asserted that the microwave treatment process generated no atmospheric emissions. Inventory data related to the laboratory plastic waste management in Method B was presented in Table S3.

2.2.3. Method C: landfilling with ozone pre-treatment (Jan 2017 to Dec 2023)

Method C involved ozone pre-treatment of laboratory plastic waste followed by landfilling at a facility located 77 km away. A total of 26,383.20 kg of laboratory plastic wastes underwent this waste management method. Waste collection was initially conducted on a monthly basis from January 2017 to July 2017, after which the frequency increased to at least bi-weekly intervals. The ozone treatment plant operated on both electricity and water. Averagely, the ozone treatment plant required electricity at a power consumption rate of 56 kWh. The electricity consumption was calculated separately in each year to account for variations in the electricity generation mix and corresponding emission factors. Additionally, the process required a water input of 1 gallon (equivalent to 0.0038 m^3) to generate humidified ozone for decontamination process. The ozone treatment plant was dedicated to process 450 kg of waste per hour. Inventory data related to the laboratory plastic waste management in Method C was presented in Table S4.

2.3. Economic assessment

Cost data for laboratory waste management were retrospectively collected from invoices issued by licensed waste facility operators to the university laboratory over a 10-year period (2014–2023). All costs were recorded in Malaysian Ringgit (RM) and compiled into a structured spreadsheet database. The invoices included detailed breakdowns of the total quantity of clinical waste collected (in kg), unit disposal costs, transportation charges per trip, number of collection trips, total transportation charges, and applicable Goods and Services Tax (GST). The number of collection trips each month was determined by the number of consignment notes issued, as each consignment note represented a single collection event. There were no rebate or discount agreements between the university and the waste contractors. According to the cost model charged by the waste facility operator, the invoiced charges are structured to ensure full cost recovery for the business operations. This includes both capital expenditure (CAPEX) such as investment in treatment technologies and infrastructure, and operational expenditure (OPEX) which covers all costs associated with daily waste handling operations, such as transportation, treatment, disposal, and labour costs related to the waste management workforce, alongside maintenance and logistical functions. Therefore, the invoiced costs comprehensively reflect the overall expense required to sustain the waste management system from point of collection to final disposal. No internal or indirect costs, such as laboratory labour or waste segregation time, were included in this analysis.

Over the 10-year study period, the three treatment and disposal methods previously described (Methods A, B, and C) were each managed by separate waste transporters and disposal facility operators, with cost structures varying across methods. For Method A, the waste disposal cost was RM 3.00/kg, with a transportation fee of RM 100 per trip and a total of 14 collection trips recorded. In Method B, the disposal cost remained at RM 3.00/kg, but the transportation fee was reduced to RM 50 per trip, and a 6 % Goods and Services Tax (GST) was applied. A total of 20 trips occurred during this period. Under Method C, the waste was initially charged at RM 3.00/kg, which was reduced to RM 2.60/kg beginning in August 2018. Transportation fees of RM 50 per trip were applied until March 2018, during which a total of 23 collection trips occurred. From April 2018 onward, transportation fees were waived. Additionally, a 6 % GST was applied until May 2018, after which the tax was lifted following a national policy change. A summary of the input economic data is provided in Table S5.

To facilitate comparative analysis across different time periods and treatment methods, the waste management costs were normalised by dividing the total invoiced cost by the corresponding total waste amount. By standardising costs on a per-kg basis, the analysis provides a consistent unit for comparing the economic performance of each waste management method over time.

3. Result and discussion

3.1. Climate change impact of laboratory single-use plastic waste

This study quantifies the GHG emissions associated with laboratory plastic waste generated by the Biological Sciences laboratory in a private university in Malaysia over a decade (2014–2023). The environmental impact was assessed across the entire laboratory plastic waste management lifecycle, including the waste collection and transportation to the treatment facility, processing and treatment of waste, and final disposal. GHG emissions were estimated in terms of CO_2eq through manual calculation.

Table 2 summarises the calculation results of the GHG emissions from all three methods as shown in Fig. 1. Across the ten years of the operation of the laboratory, a total of 46,420.08 kg CO_2eq was emitted from the management of laboratory plastic waste. This corresponds to 1.59 kg CO_2eq/kg of laboratory plastic waste being treated and

Table 2
Net GHG emission for laboratory plastic waste management at the laboratory in different method.

Eq.	Phases	Unit	Method A	Method B	Method C	Total
Plastic waste input, PW						
-	PW	kg	681.58	2115.33	26,383.20	29,180.11
Transportation emission, E_T						
Eq. (3)	T	tkm	102.24	165.00	2035.33	2302.57
Eq. (4)	$\boldsymbol{E_T}$	**kg CO_2eq**	**52.37**	**84.52**	**1042.66**	**1179.55**
Treatment plant operation emission, E_{OP}						
Eq. (7)	t	hours	1.14	8.46	65.43	75.02
Eq. (6)	*energy*	L or kWh	528.91	507.68	3725.01	4761.60
Eq. (8)	E_{energy}	kg CO_2eq	181.83	389.39	2887.91	3459.13
Eq. (9)	*water*	m^3	NA	NA	0.19	0.19
Eq. (10)	E_{water}	kg CO_2eq	NA	NA	0.07	0.07
Eq. (5)	$\boldsymbol{E_{OP}}$	**kg CO_2eq**	**181.83**	**389.39**	**2887.98**	**3459.20**
End-of-life emission, E_{EOL}						
Eq. (12)	$E_{CO_2,A}$	kg CO_2eq	1874.35	NA	NA	1874.35
Eq. (13)	$E_{N_2O,A}$	kg CO_2eq	9.03	NA	NA	9.03
Eq. (11)	$E_{EOL,A}$	kg CO_2eq	1883.38	NA	NA	1883.38
Eq. (14)	$E_{EOL,B/C}$	kg CO_2eq	NA	2961.46	36,936.48	39,897.94
Eq. (10)	$\boldsymbol{E_{EOL,i}}$	**kg CO_2eq**	**1883.38**	**2961.46**	**36,936.48**	**41,781.32**
Total GHG emission						
Eq. (2)	***GHG***	**kg CO_2eq**	**2117.58**	**3435.38**	**40,867.12**	**46,420.08**
Net GHG emission per 1 kg of laboratory plastic waste						
-	***Net GHG***	**kg CO_2eq**	**3.11**	**1.62**	**1.55**	**1.59**

disposed. Over the respective periods, Method A, which involved incineration, emitted 2117.58 kg CO2eq from 681.58 kg of laboratory plastic waste, while Method B, utilising microwave pre-treatment followed by landfilling, emitted 3435.38 kg CO2eq for 2115.33 kg of waste. Method C, which employed ozone pre-treatment followed by landfilling, resulted in the highest GHG emission at 40,867.12 kg CO2eq, managing 26,383.20 kg of waste.

A breakdown of the distribution of GHG emissions across different waste management stages for the three waste management methods (A, B, and C) and the overall waste management process revealed that direct emissions from the end-of-life disposal stage were the primary contributors across all methods (Fig. 2). Notably, the end-of-life direct emissions consistently constitute the largest contribution (>85 %) of GHG emissions in all three methods. This indicates that the final disposal method at the end of the life cycle of the laboratory plastic waste have a significant environmental impact. The result is exactly consistent with Ji et al. (2024) where direct emission accounts for more than 85 % of the total CO_2eq. By prioritising more sustainable end-of-life waste management practices, such as recycling recyclable plastic consumables like PP, PC, and PET instead of incineration, significant reductions in

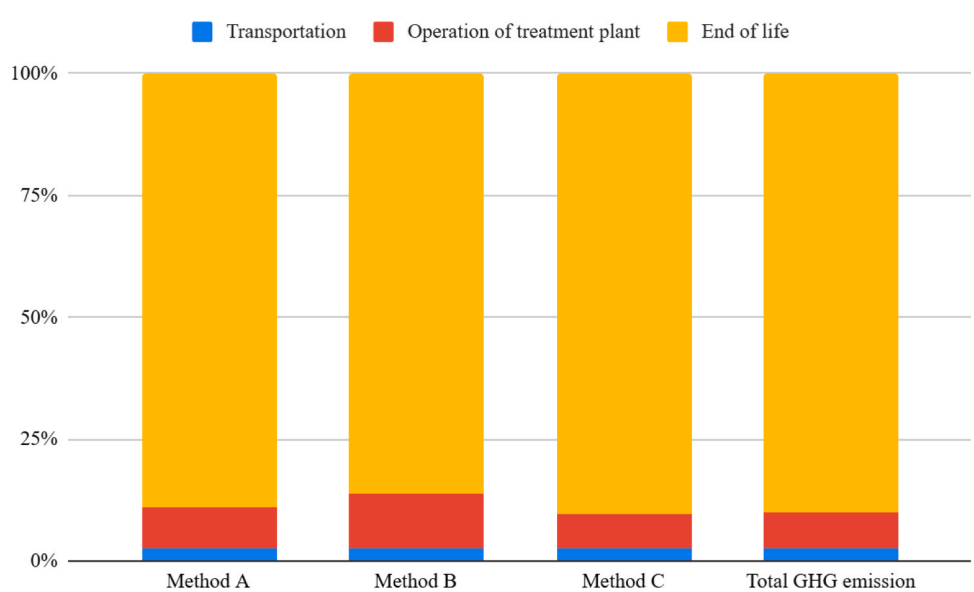

Fig. 2. Contributions of the GHG emissions from each stage of laboratory plastic waste management.

lifetime emissions, up to 74 %, are attainable (Ragazzi et al., 2023).

Incineration is a thermal treatment technology that burns the waste at very high temperature (800–1200°C) to eliminate pathogens and microbial contamination, subsequently resulting in residual ashes that will be disposed in landfill (Abhilash and Inamdar, 2022). In this study, Method A produced the highest average emissions at 3.11 kg CO_2eq/kg of plastic waste. This is in agreement with Ji et al. (2024) where incineration has the highest GHG emissions among eight analysed technologies. The emission result is slightly higher than the 2.4 kg CO_2eq/kg reported by Lee et al. (2022) for gate-to-grave incineration of plastic beverage cups without energy recovery and the 1.6 kg kg CO_2eq per 1 kg of medical waste incinerated reported by Ji et al. (2024). Direct emissions from incineration primarily result from the release of CO_2 and N_2O, generating 2.76 kg CO_2eq/kg of laboratory plastic incinerated. This result align with Hamilton et al. (2019), who reported incineration emissions of up to 2.9 kg CO_2eq per 1 kg of plastic packaging burned. Despite N_2O with a GWP 265 being significantly more potent as a GHG than CO_2, N_2O only accounted for less than 0.5 % of the total end-life emissions in incineration while CO_2 remains the primary contributor to overall emissions from incineration. Incineration also produces unburned residues or ashes as by-products. The ashes are discharged from the incinerator combustion chamber and stored in designated containers at the waste storage area. Once sufficient quantities are collected, the ashes are disposed of at a secured landfill (Tri Ecoedge Sdn Bhd, 2017). However, as the ash content in plastic waste is insignificant (< 5 %), it is not accounted in this study (Chang, 2023; Chen et al., 2019; Zabaniotou and Vaskalis, 2023).

Incineration has the potential to recover up to 3.5 kWh of electricity per kg of plastic waste, which could lead to a reduction of 3 kg CO_2eq/kg (Lee et al., 2022). However, in Malaysia, waste-to-energy (WTE) incineration technology has primarily been applied to municipal solid waste since the 2010s, and its use for laboratory waste remains underexplored (Yong et al., 2019). Although WTE has proven effective in industrial and municipal waste management, its feasibility for clinical, medical, and laboratory plastic waste remains an area for further research and development (Abhilash and Inamdar, 2022; Ahmad et al., 2019; Mudersbach et al., 2023). Thus, energy recovery was not considered in this study due to its limited implementation in Malaysia for laboratory waste.

Microwave pre-treatment is a sterilisation process that uses electromagnetic waves to trigger molecular vibrations via electromagnetic waves with wavelengths range of 1–1000 mm, increasing internal energy and maintaining the high temperatures needed for disinfection. Banana et al. (2013) revealed that 5 min of exposure to microwave irradiation sufficiently sterilise the clinical waste, suggesting that microwave technology is an efficient method for treating laboratory waste with the same infectious nature. The microwave machine studied operates at 2450 MHz for disinfection (Ecosteryl, 2021). This aligns with previous studies demonstrating the effectiveness of 2450 MHz microwaves in eliminating bacteria and fungi from clinical waste (Mahdi and Gomes, 2019; Wu and Yao, 2010). In this study, microwave pre-treatment with landfilling (Method B) showed a substantially lower GHG emission profile, averaging 1.62 kg CO_2eq/kg of waste. However, Ji et al. (2024) reported a lower result in which microwave sterilisation of medical waste and subsequent landfilling accounted for 0.308 kg CO_2eq per 1 kg of waste. This difference in emissions could be attributed to several factors, including variations in energy efficiency, waste composition, and landfill management practices. Nonetheless, Method B remains a more environmentally friendly option compared to Method A. This finding is consistent with the results of Zhao et al. (2021), further supporting the environmental benefits of microwave pre-treatment in clinical waste management.

Ozone pre-treatment followed by landfilling (Method C) demonstrated the lowest GHG emissions, at 1.55 kg CO_2eq/kg of waste. Ozone is a strong oxidising agent commonly used to eliminate pathogens, including bacteria and viruses, at a molecular level. Ozone penetrates materials to neutralise harmful microorganisms, making the waste safe for disposal (Ghozikali et al., 2024). The generation of ozone involves the dissociation of oxygen molecules (O_2) when subjected to electrical charge. The resulting oxygen atoms (O) readily bond with other oxygen molecules, yielding ozone (O_3). The highly reactive ozone decomposes back into oxygen after a short period, leaving no harmful residues (Clean Waste Systems, n.d.-b). This advanced technology is considered a key approach to reducing environmental pollution (Khannam et al., 2021). However, its environmental impact in waste management remains largely unexplored. The lack of existing literature on the environmental impact of ozone pre-treatment makes this analysis a novel contribution. While ozone pre-treatment appears to offer a low-emission alternative, further studies are essential to validate its sustainability and effectiveness compared to other sterilisation and treatment methods.

Both microwave and ozone pre-treatment methods (Method B and Method C respectively) rely on landfilling as the final disposal route, where the dominant GHG emitted is methane (CH_4). This finding is consistent with Ji et al. (2024), which reported methane as the predominant GHG emitted from landfilling scenarios. This reinforces the importance of advanced landfill gas capture systems to reduce methane leakage into the atmosphere. Captured CH_4 can be converted into energy sources like electricity or heat, thereby benefiting both economic and environmental impact (Lee et al., 2022).

3.1.1. Optimisation of waste management technology to minimise environmental impact

While end-of-life direct emissions are a major concern, it is important to note that the operation of the treatment plants also plays a crucial role in the GHG emission of laboratory plastic waste management. These emissions, which are directly linked to energy consumption, contribute to 8.6 %, 11.3 %, and 7.1 % of total GHG emissions in Methods A, B, and C, respectively. As energy consumption is a key factor influencing emissions, the technical specifications of treatment technology becomes critical. Technologies of treatment plants with lower energy consumption and higher processing capacity are generally preferable from a sustainability perspective. For instance, Method C offers a higher processing capacity, with the ozone treatment plant handling 450 kg/hr, compared to 250 kg/hr for the microwave treatment plant in Method B. This greater capacity allows Method C to handle larger waste volumes in shorter time frames, potentially reducing the energy consumed per kg of waste processed. Such operational efficiency can lead to lower overall GHG emissions, aligning with findings that emphasise the benefits of high-capacity, energy-efficient treatment systems in waste management (Shan et al., 2023).

In addition to processing capacity of the treatment plant, the choice of energy source plays a significant role in determining the environmental impact of waste treatment methods. Incinerator in Method A, while offering the highest processing capacity at 600 kg/hr, relies on both diesel and electricity. Notably, diesel use contributes to 66.5 % of the treatment plant's operational GHG emissions, making it a major source of environmental burden. In contrast, microwave treatment plant in Method B operates solely on electricity and, although it has a lower processing capacity (250 kg/hr), it avoids the high emissions associated with diesel combustion. In contrast, Method B operates solely on electricity, and although it has a lower processing capacity (250 kg/hr), it avoids the high emissions associated with diesel combustion. As a result, GHG emission of microwave treatment plant is approximately 1.45 times lower than those of incinerator. This comparison highlights the importance of transitioning to cleaner energy sources in waste treatment.

It is important to note that the responsibility for adopting and efficient waste treatment technologies lies not solely with the waste facilities but also with the laboratories and research institutions. They must proactively select and collaborate with waste facilities that prioritise and implement efficient waste treatment technologies. This shared responsibility ensures that sustainability is embedded throughout the entire waste management chain.

3.1.2. Effect of electricity on the environmental impact

Electricity consumption is also a key factor influencing the GHG emission from the treatment plant operation, as the power grid in Peninsular Malaysia primarily relies on fossil fuels, particularly coal (59.1 %) and natural gas (34.2 %) (Energy Commission, 2023). While a decline in coal usage by 6 % has led to a corresponding decrease in the emission factor from 0.821 (in year 2020) to 0.757 (in year 2021), the overall carbon intensity of the grid remains relatively high (Energy Commission, 2023, 2024). This is evident when compared to the power grid in Sabah with higher proportion of coal (85 %) and natural gas (6 %) (Energy Commission, 2024). As a result, Sabah has a relatively lower emission factor (0.524 in 2021) which could reduce the emissions from electricity by 31 % (Energy Commission, 2023).

These findings align with previous research highlighting the importance of energy sources in reducing the carbon footprint of plastic production processes. Using cleaner energy sources in the production of plastics polymers can significantly reduce the emissions (Ragazzi et al., 2023). As highlighted by Jeswani and Azapagic (2016), most of the environmental impacts associated with the national electricity mix would be reduced if electricity generation replaces coal or fuel oil within the UK grid. In addition, according to Rizan et al. (2021), PPE production in UK could achieve a 12 % reduction in emissions, with 75 % of this reduction attributed to the use of cleaner energy in the power grid of UK.

3.1.3. Effect of transportation distance on the environmental impact

Transportation emissions contribute minimally to the overall emissions, accounting for less than 3 % in all three waste management methods. This aligns with previous findings where transportation was identified as a minor contributor (< 6 %) to the cradle-to-gate life cycle GHG emissions of plastic laboratory consumables (Ragazzi et al., 2023).

Method A generated the highest GHG emission at 76.84 kg CO_2eq per 1 tonne of laboratory plastic waste disposed from the laboratory. Method B follows with 39.96 kg CO_2eq, and Method B with 39.52 kg CO_2eq per 1 tonne of laboratory plastic waste. Transportation emissions are primarily driven by the distance travelled. The transportation emissions in Method B and Method C are approximately 1.9 times lower than in Method A, which is consistent with the shorter transportation distances in Method B and Method C compared to Method A. Hence, the selection of an efficient transportation route is crucial for minimising unnecessary detours and excess mileage. he longer the transportation distance, the greater the fuel consumption, and consequently, the higher the GHG emissions. Therefore, optimising routes to reduce transportation distance directly reduces the GHG emissions, contributing to a more sustainable waste management process.

While Methods B and C incorporate treatment using microwave and ozone respectively, these processes do not constitute the final disposal step. Subsequent transportation to a landfill may extend the overall transportation distance and associated emissions. Integration of waste treatment and final disposal within a single location is important to minimise GHG emission. This integration may eliminate the need for additional transportation, resulting in a substantial decrease in the GHG emission of the waste management system (Rotthong et al., 2023).

3.2. Economic impact of laboratory single-use plastic waste

Over the 10-year period from 2014 to 2023, a total of 29,180.11 kg of laboratory single-use plastic clinical waste was managed, incurring a cumulative disposal cost of RM 84,890.49.

Waste treatment was carried out through three distinct technological methods, with each managed by different service providers. The distribution of waste and associated costs is summarised in Table 3.

A detailed cost breakdown indicates that Method A was the most expensive treatment approach, with an average cost of RM 5.05/kg. This high cost was driven primarily by an average transportation fee of RM 2.05/kg, which resulted from a flat RM 100 trip charge regardless of the waste volume collected. The disposal fee was RM 3.00/kg, and GST was not applied during this period.

Table 3
Summary of the result of economic assessment.

	Method A	Method B	Method C	Overall
Waste amount (kg)	681.58	2115.33	26,383.20	29,180.11
Total cost (RM)	3444.74	7786.75	73,659.00	84,890.55
Unit cost (RM/kg)	5.05	3.68	2.79	2.91

Cost efficiency also compromised by volume sensitivity. Monthly average cost of managing 1 kg of waste varied significantly, averaging RM 5.21 ± 0.59, with peaks observed during low-output months such as March 2014 (RM 6.26/kg, 30.68 kg collected). The fixed transport fee made low-output periods disproportionately expensive, while greater waste amount diluted these charges, resulting in lower unit costs. This inverse relationship between waste amount and monthly average cost indicates that incineration under Method A could benefit from batch scheduling or weight thresholds to achieve cost savings. This finding highlights an important operational insight for institutional waste planners.

Method B, which maintained the same disposal fee as Method A, offered moderate cost improvements, with an average cost of RM 3.68/kg. The primary cost reduction resulted from the reduction in the flat transport fee from RM 100 in Method A to RM 50 per trip under Method B. This change was driven by the reduction in transportation distance, which decreased from 150 km in Method A to 78 km in Method B, due to the different locations of the waste facilities associated with each method. This halving of transport costs significantly lowered the average transport fee RM 0.47/kg, especially when higher monthly waste amount was recorded. Compared to Method A, this change resulted in a 77 % savings in transportation costs. This outcome highlights the importance of selecting an optimised waste facility location, as it directly reduces transportation distance and, consequently, transport charges. In addition, starting from April 2015, the Malaysian government introduced a GST of 6 %, which applied to waste management services (ASEAN Briefing, 2015). As a result, GST added RM 0.21/kg to the overall cost. Despite this, the cost efficiency gained from lower transport charges offset the tax burden, where Method B still reflected a 27 % cost reduction per kg compared to Method A.

Method C was the most cost-efficient and scalable solution, handling over 90 % of the total waste amount (26,383.20 kg) at an average cost of RM 2.79/kg. Although initial costs were relatively high, a series of regulatory and contractual adjustments implemented in 2018 significantly enhanced the cost-efficiency of this method. Method C initially operated under the same cost model as Method B, with a disposal fee of RM 3.00/kg, a transportation charge of RM 50 per trip, and a 6 % GST. However, significant cost reductions began in April 2018, when the transportation fee was fully waived, followed by a reduced disposal fee of RM 2.60/kg in August 2018. The abolition of GST in June 2018 further reduced the financial burden on waste generators (PwC, 2018). The reduced disposal fee of RM 2.60/kg reflected the adoption of a more energy and cost-efficient treatment technology, i.e. ozone treatment. Khannam et al. (2021) reported that ozone treatment system costs five times lower than incineration. This financial advantage is further supported by operational data indicating that ozone technology consumes 10.4 times less energy and requires minimal staffing where a single operator can manage two machines simultaneously (Clean Waste Systems, n.d.-a). These factors collectively reduce both operational expenditures, allowing waste facility operators to offer lower service rates. In total, Method C achieved a 45 % cost reduction compared to Method A and a 24 % reduction compared to Method B.

The cost disparity observed across methods also aligns with broader national waste management trends. Incineration method, as used in

Method A, are known for higher financial burdens. According to the Malaysia's Ministry of Housing and Local Government, the gate fee for incineration is RM 250 per tonne of waste, while landfilling costs significantly less at RM 48/tonne (KPKT, 2015). Phuang et al. (2023) and Yong et al. (2019) attributed these disparities to the high CAPEX and OPEX of incineration, which are largely driven by complex pollution control technologies and specialised labour requirements. Data suggested that two-thirds of incineration CAPEX is dedicated to air pollution control systems (Yong et al., 2019). In Malaysia, CAPEX for incinerators can be 10 times higher than that of sanitary landfills, and monthly OPEX may be three times higher and may exceed RM 300,000 per year (Sulaiman et al., 2007; Yong et al., 2019).

This analysis reveals that cost-effective waste management cannot be achieved through technology choice alone. For instance, Method C offered a significant cost reduction compared to Method A due to its financial benefits of energy-efficient technologies and optimised logistics, combined with lower CAPEX and OPEX. Moving forward, a comprehensive economic assessment should incorporate both CAPEX and OPEX to provide a more accurate and holistic financial evaluation. Institutional stakeholders can further optimise cost by scheduling waste collection in larger batches, negotiating weight-based pricing models, leveraging technology with lower energy and labour requirements, and advocating for policy reforms that support sustainable waste management practices.

3.3. Temporal analysis of environmental and economic impact

Fig. 3 presented a temporal perspective on the interplay between plastic waste generation, environmental impact, and disposal costs from 2014 to 2023. Several critical inflection points emerge across the decade, revealing the implications of shifting waste management strategies and broader institutional or external influences. Notably, the period from 2014 to 2015 marks a key transition, where GHG emissions dropped sharply despite a modest increase in plastic waste generation. This suggests a decisive switch to a more environmentally friendly treatment method, i.e. landfilling with microwave pre-treatment (Method B), which successfully reduced emissions per unit of waste.

From 2017 onwards, laboratory plastic waste generation and disposal costs increased sharply, while the rise in GHG emissions was relatively moderate. This trend can be attributed to the shift towards an even more sustainable waste management method, i.e. landfilling with zone pre-treatment (Method C) in 2017. The reduced emissions intensity during this period reflects the effectiveness of this advanced treatment method in mitigating environmental impact despite the rising waste volumes.

In 2019, notably, disposal costs dropped despite continued growth in plastic waste generation. This trend aligns with the policy and contractual adjustments, including the waiver of transportation fees, the abolition of the GST, and a reduction in the unit price of waste collection fee. These changes collectively contributed to lower overall disposal expenditures.

There is a significant reduction across the plastic waste generation, GHG emissions, and cost in 2020. This inflection coincided with the onset of the COVID-19 pandemic, which likely caused a sharp decline in laboratory activity and a shift in operational behaviours. After 2020, the environmental and economic impact continued to decrease proportionally with the reduced waste generation. This trend highlights the critical role of waste minimisation in achieving both sustainability and cost-efficiency. Reducing waste generation not only lowers GHG emissions and disposal costs but also alleviates the pressure on waste management systems. It reinforces the idea that minimising waste at the source remains one of the most effective strategies for sustainable laboratory operations.

3.4. Integrating environmental and economic perspectives for sustainable laboratory waste management

This study presented an integrated environmental-economic evaluation of laboratory single-use plastic waste management, offering a multi-dimensional understanding of sustainability performance (Fig. 4). Notably, the method with the lowest environmental impact, landfilling

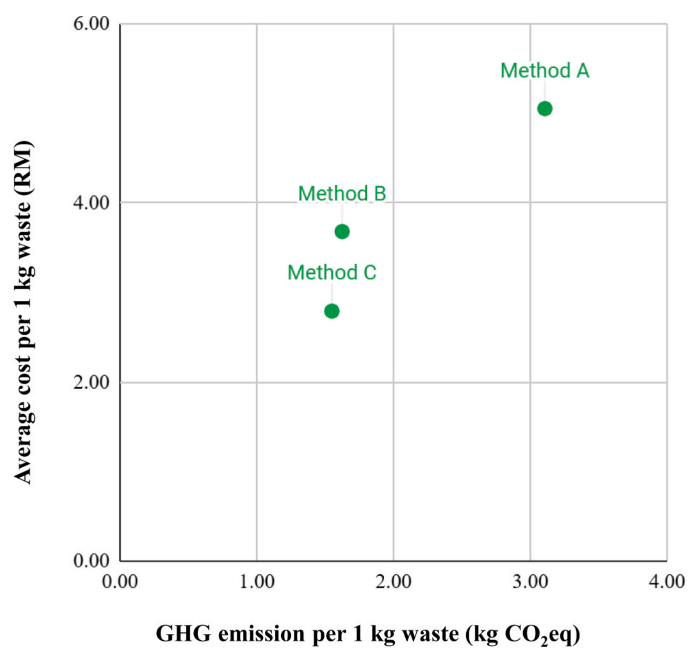

Fig. 4. Comparative analysis of the three plastic waste management methods based on unit GHG emissions and cost.

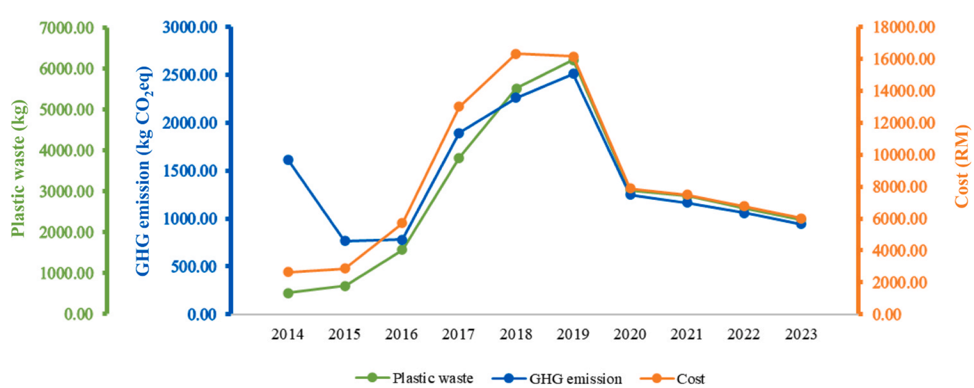

Fig. 3. Yearly GHG emission from the plastic waste management in the laboratory from 2014 to 2023.

with ozone pre-treatment (Method C), also demonstrated the highest cost-efficiency, with GHG emissions of 1.55 kg CO_2eq/kg of waste and a unit cost of RM 2.79/kg. This finding demonstrated that sustainable waste management does not necessarily entail higher operational costs. In contrast, incineration (Method A) exhibited the highest emissions (3.11 kg CO_2eq/kg) and the highest cost (RM 5.05/kg), demonstrating the environmental and economic drawbacks of traditional waste treatment systems. These findings revealed that adopting cleaner technologies not only reduce their environmental footprint but also manage long-term costs.

From a policy and procurement perspective, these findings supported the strategic selection of waste treatment technologies based on integrated sustainability metrics. Universities and research institutions should adopt green procurement strategies that consider both carbon emissions and lifecycle costs when engaging waste service providers. In a broader context, the integration of environmental and economic performance indicators reflects the evolving landscape of sustainability governance, where environmental performance must be evaluated alongside economic viability. Embedding both environmental and economic metric into institutional planning supports a more holistic transition toward low-carbon and cost-effective laboratory operations while contributes to national climate goals and the United Nations SDGs.

4. Conclusion

Malaysia's academic institutions, particularly those offering Biological Sciences programmes, are significant consumers of single-use plastic laboratory consumables such as pipette tips, Petri dishes, and centrifuge tubes. While these items are essential for maintaining sterility and operational efficiency in research and teaching, they also contribute substantially to laboratory waste. This study provides one of the first in-depth assessments that quantifies not only the environmental impact but also the economic costs of managing laboratory plastic waste over a 10-year period.

A total of 46,420.08 kg CO_2eq was emitted by 29,180.11 kg of laboratory waste generated in the 10-year period. To offset these emissions, approximately 54 trees would need to be planted and allowed to grow until 35 years old, assuming each tree sequesters 25 kg of CO_2 per year, accumulating a total of 875 kg carbon sequestration over its lifetime (Ecotree, 2024). While planting 54 trees may seem straightforward, maintaining their growth over 35 years is a significant challenge. Furthermore, as waste generation continues annually, additional trees would be required each year to offset the ongoing emissions.

Among the three waste management methods analysed, landfilling with ozone pre-treatment (Method C) emerged as both the most environmentally sustainable and cost-effective solution, producing the lowest GHG emissions (1.55 kg CO_2eq/kg) and incurring the lowest cost (RM 2.79/kg). In contrast, incineration (Method A) resulted in the highest emissions and cost, highlighting the environmental and financial drawbacks of conventional waste treatment approaches. These findings reinforce the value of integrating environmental and economic considerations into institutional decision-making. Selecting waste treatment technologies that minimise emissions while controlling costs can significantly improve the sustainability of laboratory operations.

Future studies should analyse the plastic composition of laboratory waste to accurately calculate carbon emissions and global warming potential for different disposal methods, thereby subsequently strengthen the environmental analysis. Given the limited availability of alternatives, laboratory plastic consumables are unlikely to be totally eliminated. Hence, in order to further reduce its environmental impact, a cradle-to-gate environmental analysis is necessary to optimise the entire life cycle of laboratory plastic consumables through improved design and supply chain management. While polylactic acid (PLA) is a potential alternative, its effectiveness in reducing the environmental impact needs further evaluation (Freeland et al., 2022).

Furthermore, future research should also explore and advance alternative waste management methods such as pyrolysis or recycling. Pyrolysis offers potential advantages in terms of pollution reduction and carbon footprint compared to other thermal treatments (Aragaw and Mekonnen, 2021). The growing interest in pyrolysis in Malaysia aligns with this opportunity (PETRONAS Chemicals Group Berhad, 2023). Until greener alternatives are developed, recycling laboratory plastics remains the most practical approach (Tan et al., 2022).

Finally, economic assessments should accompany environmental assessment to support decision-making in policy and procurement. This includes evaluating life-cycle costs, cost-benefit ratios of emerging technologies, return on investment (ROI) from sustainable procurement, and the potential for scaling cost-efficient systems through government incentives or public-private partnerships. Collectively, these environmental and financial insights will be essential in driving practical, sustainable, and economically viable solutions in laboratory plastic waste management.

CRediT authorship contribution statement

Lai Ti Gew: Writing – review & editing, Validation, Supervision, Formal analysis, Data curation, Conceptualization. **Yi Huan Tan:** Writing – original draft, Formal analysis, Data curation. **Raja Affendi Raja Ali:** Writing – review & editing, Supervision, Conceptualization. **Hwai Chyuan Ong:** Writing – review & editing, Validation, Formal analysis, Data curation.

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Declaration of Competing Interest

Authors declared no conflict of interest.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.clwas.2025.100389.

Data availability

The authors confirm that the data supporting the findings of this study are available within the article and the supplementary materials.

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