Assessment of open waste burning activity and its mitigation strategy to control environmental pollution

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Introduction

1.1. Background

Proper municipal solid waste (MSW) management is a critical issue in many developing countries globally. Community awareness, habits, household collection services, and other related factors are becoming essential to its management system (Ramaswami et al., 2016). Open burning, waste dumping on waterways, and other uncontrolled waste management practices are still problems when waste services are not provided (Triassi et al., 2015). Those problems are strategically discussed in the local community to fulfill the objectives of sustainable development goals (SDGs). However, the local government and community's lack of discipline and commitment implements many strategic actions going slowly and ineffective (Kristanto and Koven, 2019).

Activities and incidents inventories of open waste burning (OWB) are necessary to produce a proper strategy for decreasing the number of wastes burning in a specific region. Chemical species and predicted emission of OWB have also been evaluated in several urban and rural locations in the world. Most of the research are coming from developing countries such as India (Kumar et al., 2018; Kumari et al., 2019; Lal et al., 2016; Nagpure et al., 2015; Sharma et al., 2019), Nigeria (Adesina et al., 2020; Daffi et al., 2020a; Oguntoke, 2019; Okedere et al., 2019a; Pasquini and Alexander, 2004; Rim-Rukeh, 2014), Nepal (Das et al., 2018b), Ghana (Boadi and Kuitunen, 2005), Ethiopia (Bulto, 2020); Mexico (Gullett et al., 2010; Reyna-Bensusan et al., 2019, 2018; Zhang et al., 2011), China (Li et al., 2019; Lundin et al., 2013; Wang et al., 2017; Xu et al., 2009), Thailand (Pansuk et al., 2018), Jordan (Haddad and Moqbel, 2018), Lebanon (Baalbaki et al., 2018; Mouganie et al., 2020), Kenya (Shih et al., 2016), Indonesia (Bastian et al., 2013), and Brazil (Krecl et al., 2021); and a few research come from developed countries such as Korea (Hwang et al., 2017; Park et al., 2013), Hungary (Hoffer et al., 2020) and United States (Wiedinmyer et al., 2014).

Some researchers have been reviewed the potential emission and health risks of OWB. Akagi et al. (2011) have comprehensively reviewed global biomass burning, including OWB for atmospheric models input. Wiedinmyer et al. (2014) estimated global emission from OWB using greenhouse gas (GHG) inventories methods of IPCC guidelines. Some emission factor (EF) was also reviewed in the study. Chen et al. (2017) also reviewed biomass burning emission and air quality impact in China. While comparative assessment of the robust method to predict waste burning emission in the field is needed, knowing the exact emission and methods of OWB incidents will reference policymakers to make appropriate mitigation actions for reducing emission from OWB (Cheng et al., 2020a). From a government perspective, reducing OWB incidents will contribute to accomplishing 3, 11, 13, and 14 SDGs agenda (Krecl et al., 2021). The researchers systematically reviewed the toxicological risks associated with OWB for public health and the surrounding environment (Proietti and Mantovani, 2017; Renan and Iino, 2010; Velis and Cook, 2021).

Some techniques and methods are presented in the previous literature to estimate OWB incidents, activities, and emissions. IPCC default methods (tier 1) are the well-known methods since they can give a simple model and projection of waste burning emission in a municipality, typically using business as usual (BAU) scenario. However, IPCC tier 1 cannot predict an apparent OWB emission since there is the uncertainty of the source composition (Reyna-Bensusan et al., 2019). Then, some approaches such as transect walk (Nagpure et al., 2015), which is combined with household interview survey (Das et al., 2018) and fixed (static/plume sampling)-mobile field monitoring (Krecl et al., 2021; Zhang et al., 2011) for defining the activities and estimating the actual field emission are well developed for achieving higher tier levels. Determining the appropriate techniques for emission inventory will be necessary for mitigating OWB practices in the field.

1.2. Objectives and Scopes

1.2.1. Research Objectives

As there are gaps in the activity and inventory assessment of MSW open burning, later known as open waste burning (OWB) or domestic OWB, this study has several aims or objectives to accomplish.

a. Analyzing the status of open burning at the global and regional levels, the environmental and health impact of OWB, and factors affecting OWB
b. Analyzing the temporal pattern of OWB in Semarang City
c. Assessing the environmental and health risk of OWB practices
d. Evaluating the appropriate waste collection site reallocation for reducing OWB practices
e. Examining the appropriate policy recommendation for reducing open waste burning in SEA countries

1.2.2. Research Scope

For the first chapter, systematic literature network analysis (SLNA) was conducted to understand the study's first aim. The literature review task was done by analyzing status, environmental and health impact, and factors affecting OWB practices were searched and analyzed through Google Scholar and Scopus databases. For the second to the fourth chapter of this study, the study area is limited to Semarang City, which consists of assessing the temporal activity of OWB, evaluating environmental and health risks, and countermeasures to reduce the OWB. Semarang City became the pilot study area, representing a city in SEA countries, especially Indonesia. Indonesia has two classifications of cities: city and regency; a typical regency has a less dense population and a larger area than a city. Semarang is the capital city of the Central Java Province, which explains the availability of an efficient waste collection system covering almost all sub-districts. The temporal pattern evaluation of OWB activities was conducted in 1 year (during rainy and dry seasons). For the third chapter, the environmental risks did not include the possibility of dioxin emitted during waste burning. In the spatial analysis, which is determined as one example of the countermeasures to reduce OWB, the waste collection route was ignored and only focused on the reallocation of waste collection point (e.g., distribution, reallocation, and reevaluation of the existing waste collection site). The fifth chapter assesses the priority of policy recommendations on the Southeast Asian (SEA) countries.

1.3. Methodological Framework

To achieve the research goals, several steps have been taken. First, the literature review was done from a global perspective and SEA countries. The literature review employed systematic literature network analysis (SLNA), which combines bibliometric analysis (BA) and qualitative content analysis. The second methodology used is the transect walk method, which employed field surveyors to find any OWB activities in the sampling site. This method successfully evaluated the OWB activities in dry and wet seasons. Several waste samples from the sampling site were taken and assessed in the laboratory during the transect walk methods. Chemical speciation for determining the fly ash and bottom ash was used to evaluate the risk of OWB to humans. Therefore, environmental impact was estimated based on the number of OWB activities multiplied by several emission factors available from the literature. Spatial analysis, which includes kernel density, average nearest neighbor, and incremental spatial autocorrelation, combined with multi-criteria decision analysis (MCDA) and location-allocation analysis, was done to evaluate the placement of existing and reallocated new waste collection site (WCS) in Semarang City. Lastly, strength-weakness-opportunity-threat (SWOT) and quantitative strategic planning matrix (QSPM) were conducted to assess the questionnaire sent to experts on waste management. This analysis was used to prioritize several recommendations taken from the literature survey. Detailed information regarding each research method is described in the following chapters. The research framework can be seen in Figure 1.1.

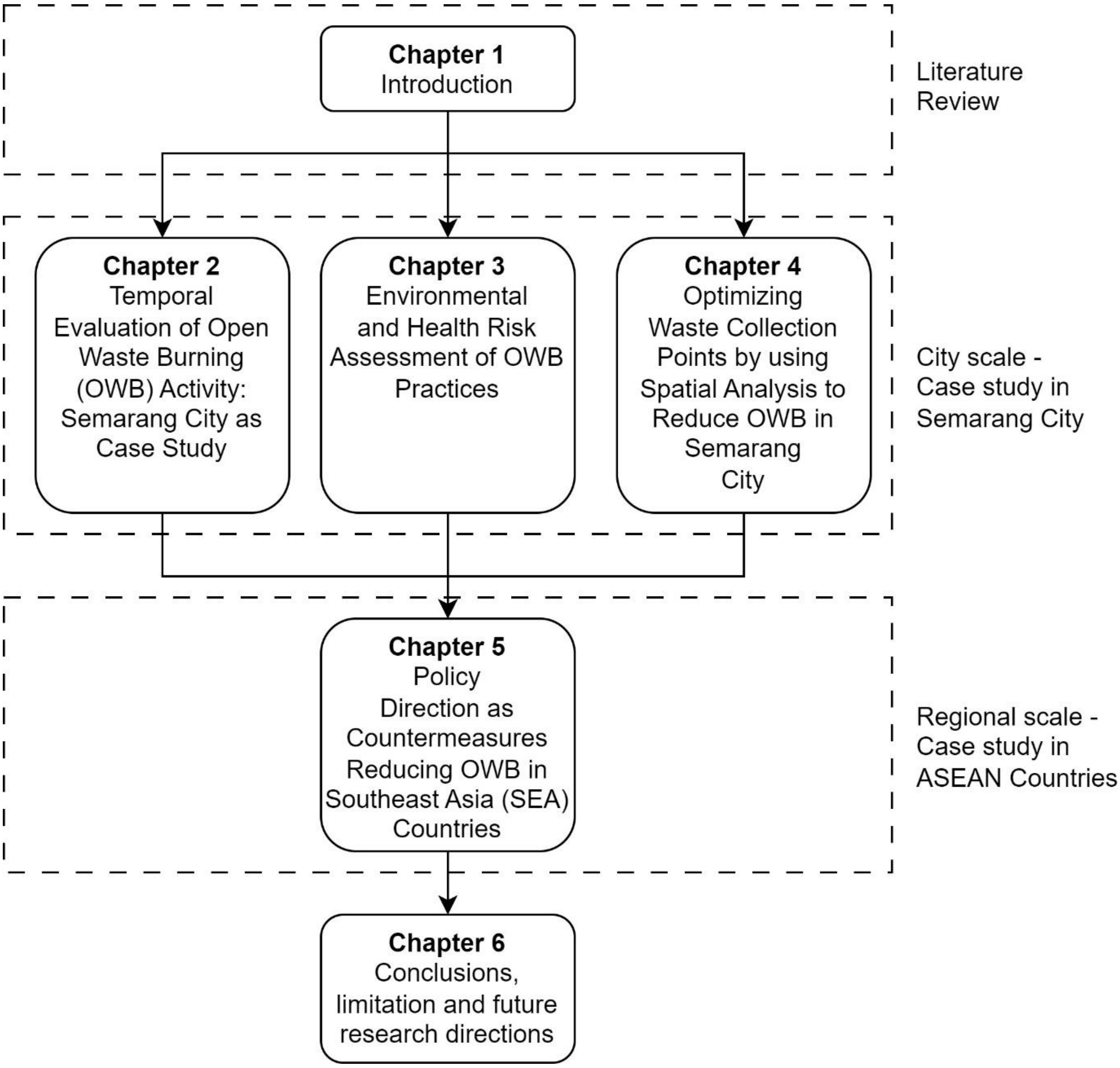

Figure 1.1 Research Framework

1.4. Current Status of OWB Research

1.4.1. Global Status of OWB

Domestic-OWB practices occurred mainly in the developing countries where waste collection is becoming a significant problem of waste management systems in the local authorities (Hoffer et al., 2020). OWB refers to burning any unused materials consisting of agricultural residues, construction scraps, backyard waste, and municipal waste (Li et al., 2019). Around 20% of air pollution in Mumbai, India, is contributed by OWB (Lal et al., 2016). Typically, suburbs and the peripheral area have many open spaces that make OWB possible without disturbing their neighbor (Daffi et al., 2020). OWB emission in the low socioeconomic status is also significantly higher than other places where this may give more risk to the rural neighbor. People in those areas also have a large backyard to dig pits for their temporary waste disposal (Mihai et al., 2019). Municipal solid waste is then disposed of in the pit and collected until complete before it is being burned (Remigios, 2013). It is reported that OWB incidents were found in more than 30% in rural areas and 13% in urban areas. Another disposal practice that most people do is collect in the formal waste collection system, open dumping, burying, and dispose of waste to the water bodies (See Table 1.1). This fact is considered as mismanagement of domestic waste that people in developing countries are still doing.

Table 1.1 Typical waste disposal practices worldwide

Area	**Dispose of in the waste collection / disposal site**	**Composted**	**Segregated for sale and animal feed**	**Open waste burning (in and outside backyard)**	**Dumping randomly (in and outside backyard, ditches, abandoned land)**	**Bury (in and outside backyard)**	**Dispose of in the river, canal, swamps**	**Others**	**References**
Urban Areas									
Kampala City, Uganda	87.00%	-	-	13.00%	-	-	-	-	(Kulabako et al., 2010)
Huejutla City, Mexico	24.6%	-	14.8%	22.4%	-	2.1%		38.5%[a]	(Reyna-Bensusan et al., 2018)
Peri Urban									
Kano Metropolis, Nigeria	16.25%	-	3.75%	3.75%	66.25%	-	-	13.25%	(Nabegu, 2010)
Rural Areas									
Rural Thailand	23.70%	0.40%	10.72%	53.70%	5.20%	1.10%	0.50%	4.70%	(Pansuk et al., 2018)
Rural Southwest China	35.00%	8.00%	-	30.00%	27.00%	-	0.00%	-	(Han et al., 2015)
Rural Iran	12.55%	4.30%	8.55%	47.50%	14.85%	6.10%	6.15%	-	(Vahidi et al., 2017)

[a] Waste from commercial, schools, and hospitals which are disposed of by their technologies

Pansuk et al., (2018) reported that more than 50% of domestic waste produced by rural communities in Thailand is burned. The average waste composition that is being burned mainly consists of organic waste (62.71%), followed by plastics (31.68%), and others such as paper and cardboard, glass, metals, leather, wood textiles, and rubber (LWTR). This composition indicates that some primary hazardous aerosols which occur during plastic burning, such as carbonaceous compounds, acidic gases, and smoke, will be released into the surrounding environment (Forbid et al., 2011). Garden waste such as wood, leaves, and other pruning residues may also emit many pollutants, which can have a significant and harmful impact on the environment (Alves et al., 2019). Since Thailand generally has only two seasons (wet and dry), the domestic waste that is usually burnt may be different. Nagpure et al., (2015) explored the differences in burned waste between summer and winter. They found that people tend to burn more organic waste (compostable waste) than other waste in the summer season. In the winter season, wood and paper waste are more favorable to burn. Therefore, this phenomenon may be different, which is dependent on the socioeconomic status of the community. Each area has different characteristics, e.g., rural, urban, and peri-urban (transition) areas, leading to the difference of pollutants emitted (Park et al., 2013). Table 1.2 shows the domestic waste composition from the published literature.

Table 1.2 World domestic waste composition in rural, peri-urban, and urban areas

No	City	Paper and Cardboard	Glass	Metals	Plastics	Leather, wood, textiles, and rubber	Organic, garden, and food waste	Other	References
Urban Areas of Developed Countries									
1	Czech Republic	25.70%	11.20%	1.70%	16.80%	15.10%	15.60%	13.90%	(Doležalová et al., 2013)
2	Five Norwegian cities	31.50%	5.30%	5.50%	13.20%	3.60%	28.90%	12.00%	(Slagstad and Brattebø, 2013)
3	Danish Municipalities	15.80%	2.10%	2.30%	12.60%	17.60%	45.70%	3.90%	(Edjabou et al., 2015)
4	Bucharest and Timisoara City, Romania	8.62%	3.89%	3.16%	14.41%	5.10%	50.61%	14.21%	(Ciuta et al., 2015)
5	Brussels City, Belgium	12.40%	2.54%	12.81%	6.80%	5.77%	12.98%	46.70%	(Zeller et al., 2019)
Urban Areas of Developing Countries									
1	Makurdi Urban Area, Nigeria	5.16%	1.88%	1.77%	7.31%	2.64%	39.96%	41.28%	(Sha'Ato et al., 2007)
2	Four Mexican Cities	10.75%	4.48%	1.90%	8.70%	-	53.67%	20.50%	(Gomez et al., 2008)
3	Sangamner City, India	5.83%	1.91%	4.78%	5.73%	13.85%	40.97%	26.93%	(Thitame et al., 2010)
4	Kano Metropolis, Nigeria	2.10%	2.75%	2.20%	11.30%	3.80%	53.57%	24.28%	(Nabegu, 2010)
5	Ahvaz City, Iran	11.40%	1.20%	1.18%	7.09%	1.27%	76.86%	1.00%	(Monavari et al., 2012)
6	Gaza Strip, Palestine	11.00%	3.00%	3.00%	13.00%	-	53.00%	18.00%	(AbdAlqader and Hamad, 2012)

No	City	Paper and Cardboard	Glass	Metals	Plastics	Leather, wood, textiles, and rubber	Organic, garden, and food waste	Other	References
7	Surabaya City, Indonesia	8.70%	1.00%	0.70%	13.90%	3.6%	60.40%	11.90%[c]	(Bastian et al., 2013)
8	Abuja, Nigeria	9.70%	2.60%	3.20%	8.70%	1.60%	63.60%	10.60%	(Ogwueleka, 2013)
9	Muscat City, Oman (Winter)	25.20%	8.00%	2.80%	16.40%	7.20%	33.80%	6.60%	(Palanivel and Sulaiman, 2014)
10	Dehradun City, India	8.00%	1.00%	7.00%	-	-	80.00%	4.00%	(Suthar and Singh, 2015)
11	Kumasi Metropolis and Accra Metropolis, Ghana	12.00%	-	5.10%	28.70%	-	50.50%	3.70%	(Boateng et al., 2016)
12	Kosrowshah City, Iran	6.31%	1.13%	0.86%	13.94%	11.07%	50.98%	11.28%	(Taghipour et al., 2016)
13	Muscat City, Oman (Average)	19.38%	2.89%	2.59%	31.32%	15.75%	14.27%	13.80%	(Baawain et al., 2017)
14	Xiamen City, China	9.92%	3.60%	1.04%	13.05%	5.60%	65.70%	1.09%	(Xiao et al., 2017)
15	Urban China	9.31%	2.03%	4.29%	1.70%	8.52%	72.96%	2.06%[b]	(Han et al., 2017)
16	Sapele City, Nigeria	6.35%	3.52%	2.53%	10.23%	-	75.22%	2.15%	(Orhorhoro et al., 2017)
17	Kerbala City, Iraq	15.00%	2.40%	3.60%	14.60%	-	57.90%	6.50%	(Abdulredha et al., 2017)
18	Thu Dau Mot City, Vietnam	11.03%	2.41%	-	11.67%	-	67.88%	7.01%	(Trang et al., 2017)
19	Thailand City Municipalities	10.82%	2.49%	1.10%	22.22%	2.80%	57.75%	2.82%	(Pansuk et al., 2018a)
20	Seven cities in Taiwan	42.37%	1.82%	0.63%	17.94%	-	33.22%	4.02%	(Chen, 2018)
21	Medan City, Indonesia	4.14%	3.70%	1.71%	5.43%	1.56%	79.16%	4.30%	(Khair et al., 2018)
22	Depok City, Indonesia	7.07%	1.25%	1.37%	3.57%	3.65%	76.61%	6.48%	(Kristanto and Koven, 2019)
Peri- / Semi-Urban / Transition / Mixed Areas of Developed Countries									
1	Czech Republic	22.60%	7.80%	2.10%	17.60%	16.40%	21.60%	11.90%	(Doležalová et al., 2013)
Peri- / Semi-Urban / Transition / Mixed Areas of Developing Countries									
1	Kano Metropolis, Nigeria	17.20%	20.55%	9.49%	18.50%	9.30%	17.50%	7.46%	(Nabegu, 2010)
2	Kampala City, Uganda	11.00%	2.00%	0.40%	12.00%	1.00%	73.20%	0.40%	(Kulabako et al., 2010)
3	Thailand Sub-district Municipalities	10.36%	1.98%	1.69%	26.36%	4.81%	52.44%	2.36%	(Pansuk et al., 2018a)
4	Medan City, Indonesia	1.75%	1.60%	1.65%	4.91%	1.61%	86.29%	2.19%	(Khair et al., 2018)
Rural Areas of Developed Countries									
1	Czech Republic	7.80%	4.90%	2.60%	9.70%	11.70%	11.70%	51.60%	(Doležalová et al., 2013)
Rural Areas of Developing Countries									

No	City	Paper and Cardboard	Glass	Metals	Plastics	Leather, wood, textiles, and rubber	Organic, garden, and food waste	Other	References
1	Bangkok, Thailand	12.10%	6.60%	3.50%	10.90%	7.30%	43.00%	16.60%	(Chiemchaisri et al., 2007)
2	Kano Metropolis, Nigeria	2.90%	3.96%	2.60%	11.70%	4.60%	53.84%	20.40%	(Nabegu, 2010)
3	San Quintin and Vicente Guerrero, Mexico	9.53%	3.95%	2.05%	14.70%	7.48%	34.71%	27.58%	(Taboada-González et al., 2011)
4	Southwest China	10.55%	5.19%	0.66%	13.96%	9.00%	40.64%	20.00%	(Han et al., 2015)
5	Sercaia, Brasov County, Romania	8.08%	1.64%	2.34%	5.16%	2.54%	55.36%	24.88%	(Ciuta et al., 2015)
6	Atwima Nwabiagya and Dangbe West District, Ghana	-	-	-	36.40%	-	63.60%	-	(Boateng et al., 2016)
7	12 Villages in Iran	6.07%	2.09%	0.47%	13.58%	0.14%	50.98%	12.27%	(Taghipour et al., 2016)
8	Rural China	7.77%	2.40%	1.28%	8.78%	4.90%	43.58%	31.23%[b]	(Han et al., 2017)
9	Chaharmahal and Bakhtiari and Yazd Province, Iran	9.60%	6.50%	13.45%	15.30%	9.70%	40.15%	5.30%	(Vahidi et al., 2017)
10	5 villages in India	6.63%	0.42%	0.41%	3.35%	2.90%	81.03%	5.26%	(Mandawat, 2017)
11	59 rural areas in China	7.77%	2.45%	1.28%	8.78%	2.75%	45.73%	31.24%	(Han et al., 2017)
12	Mae Salong Nok Sub-district, Thailand	9.93%	11.28%	1.66%	30.67%	2.18%	42.79%	1.49%	(Suma et al., 2019)

[a] construction waste, particular waste, inert, ash waste, hazardous waste, fines
[b] including ash waste
[c] including diapers

1.4.2. OWB Practices on Southeast Asia (SEA) Countries

As reported in the previous sub-section, the data on the open burning of MSW is limited in the literature. On the global scale, the number of OWB is reported to be more than 10% of total waste generated (See Table 1.1). Therefore, knowing each country's economic situation, waste generation, and waste collection efficiency is essential to predict the possibility of burning practices at the household level. Looking up to the regional level, the number of wastes burning at the household level can be estimated based on waste collection efficiency. As can be seen in Table 1.3, the highest population is coming from Indonesia, the Philippines, and Vietnam. Therefore, the highest GDP per capita is shown in Singapore which is interestingly in line with the average waste generation per capita, the total CO_2 emission, and the waste collection efficiency. Collection efficiency is also important data to know how much waste may be burned (JICA, 2021). On average, 57% of waste is not collected in SEA countries subjected to improper disposal. However, this data still needs some country-level validations since the data taken from the literature review have different assumptions, methodologies, and time.

Table 1.3 Regional outlook of waste management in SEA Countries

Country	**Population**[a]	**GDP**[a] **(billion USD)**	**GDP per Capita**[a] **(USD)**	**CO_2 Emissions**[a] **(t per capita)**	**Total GHG Emissions**[a] **(kt $CO_{2\text{-eq}}$)**	**Avg Waste Generation (kg/cap/day)**	**Waste Collection Efficiency (%)**
Singapore	5,453,566	360.90	66,176.4	8.3	67,230	1.10[b]; 0.94[c]	100[g]
Brunei Darussalam	441,532	13.21	29,927	16.1	9,300	0.66[b]; 1.40[c]	50-70[h]
Malaysia	32,776,195	354,88	10,827.3	7.9	313,020	0.81[b]; 0.90[c]	66-90[h]
Thailand	69,950,844	438.62	6,270.4	3.8	422,090	0.64[b,c]	59[n]
Indonesia	276,361,788	1,070.00	3,855.8	2.3	1,002,370	0.76[b]; 0.49[c]	65[i]
Vietnam	98,168,829	331.13	3,373.1	3.5	450,150	0.61[b]; 0.41[c]	60[e]
Philippines	111,046,910	378.96	3,412.6	1.3	234,280	0.52[b]; 0.40[f]	65[f]
Papua New Guinea*	9,119,005	24,21	2,655.2	0.9	22,410	0.41[d]	n/a
Laos	7,379,358	19.05	2,582.2	2.6	29,280	0.55[b]; 0.64[c]	40-50[j]
East Timor*	1,343,875	2.19	1,626.4	0.5	5,910	0.45[e]	55[k]
Cambodia	16,946,446	23.72	1,399.8	1.0	40,060	0.52[b]	72[l]
Myanmar	54,806,014	70.81	1,292.1	0.7	133,250	0.45[b]; 0.44[m]	53-84[m]

[a]data.worldbank.org accessed Nov 26, 2022; GDP is calculated using constant USD from 2015; [b]Nguyen Ngoc et al. (2009); [c]Kawai and Tasaki (2016); [d]Karak et al. (2012); [e]Woodruff (2014); [f]Premakumara et al. (2018); [g]Jerin et al. (2022); [h]Shams et al. (2014); [i]Kementerian Lingkungan Hidup dan Kehutanan (2021); [j]JICA (2021); [k]Ximenes and Maryono (2021); [l]Pheakdey et al. (2022); [m]The World Bank Infographic (2019); [m]Pansuk et al. (2018); *Observer countries

The lower waste collection efficiency (less than 60%) is found in the SEA countries. Therefore, it is estimated that around 15% of waste is burned based on Thailand and the Philippines' experience (See Table 1.4), while based to Wiedienmeyer et al. (2014), around 40% of waste is burned in the world. This number reveals that less estimated waste burning intensity is found in the SEA countries compared to the world estimation. The total municipal waste burning is less than 25% at the city level. Generally, the highest composition of burned waste is food and garden waste (Menikpura et al., 2022; Ramadan et al., 2022), whereas plastic waste is reported to give the most elevated portion in the Thailand case study (Pansuk et al., 2018). Plastic waste can emit a higher black carbon emission than other types of waste. It is also known that around 40% of waste is burned worldwide (Wiedinmyer et al., 2014). Plastic burning, especially PET and polystyrene (PS), emits black carbon more than other waste burning (Reyna-Bensusan et al., 2019).

Table 1.4 Open burning of municipal waste in ASEAN countries and cities

Location	Average waste generation (Mt/year)	Total waste burning (kt/year)	Fraction of open burning (%)	Composition (%)							References
				Food waste	Garden waste	Plastic	Paper	Metal and Glass	Textile and Rubber	Others/ Inert	
Vientiane City, Laos	0.23	35.18	15	34.0	30.0	12.0	7.0	8.0	8.0	1.0	Babel and Vilaysouk, 2016
Luangprabang City, Laos	0.03	2.64	9	39.0	31.0	8.0	6.0	2.0	5.0	9.0	Vilaysouk and Babel, 2017
Depok City, Indonesia	0.41	25.55	6.3	73.0	3.7	3.6	7.1	2.6	3.6	6.4	Kristanto and Koven, 2019
Semarang City, Indonesia	0.61	58.80	9.7	53.9 *0.2*	- *73.4*	21.5 *17.5*	10.9 *4.3*	8.7 *0.3*	- *3.3*	5.0 *1.0*	Ramadan et al., 2022*
Thailand	26.20	3,430	13	- *10.3*	48.0 *17.4*	15.0 *36.3*	15.0 *0.9*	10.0 *4.7*	- *18.1*	14.0 *12.2*	Pansuk et al., 2018*
Philippines	14.86	2,602	17.5	52		28.0			20.0 (include special waste like e-waste, healthcare, and bulky waste)		Premakumara et al., 2018
Myanmar	20.48	4.38	21.0	7.9	54.1	18.7	7.8	3.8	3.9	4.2	Jeske et al., 2020 (Report)
Steung Saen Municipality, Cambodia	12.96	2.74	21.17	34.0 *16.1*	16.0 *27.4*	23.0 *18.2*	8.0 *26.8*	4.0 -	6.0 *0.22*	9.0 *11.22*	Menikpura et al., 2022 (Report)
Padang City, Indonesia	241.08	27.67	11.48	45.2 *8.8*	6.7 *46.7*	28.1 *31.4*	7.5 *5.4*	1.8 -	1.8 *0.8*	8.9 *6.9*	Menikpura et al., 2022 (Report)
Bago City, Myanmar	35.95	0.75	2.10	45.0 -	10.0 87.3	25.0 *10.0*	4.0 *0.9*	3.0 -	8.0 *0.2*	6.0 *1.6*	Menikpura et al., 2022 (Report)

*The italic number represents the composition of burned waste pile (below) which is different with the municipal waste composition (above)

According to the newest published report, the BC emission from open waste burning is in line with the number of populations (See Figure 1.2). However, the number of wastes burning in Bago City, Myanmar, is lower than others even though the population is higher than Steung Saen, Cambodia and Luangprabang City, Laos. A higher waste collection rate is reported there, which could be why the burning intensity is lower than in other cities. The lesson learned is that when the waste collection services are higher, the possibility of burning practices will decrease. Thus, promoting and establishing waste infrastructure helps reduce the burning practices (Pansuk et al., 2018b; Ramadan et al., 2022).

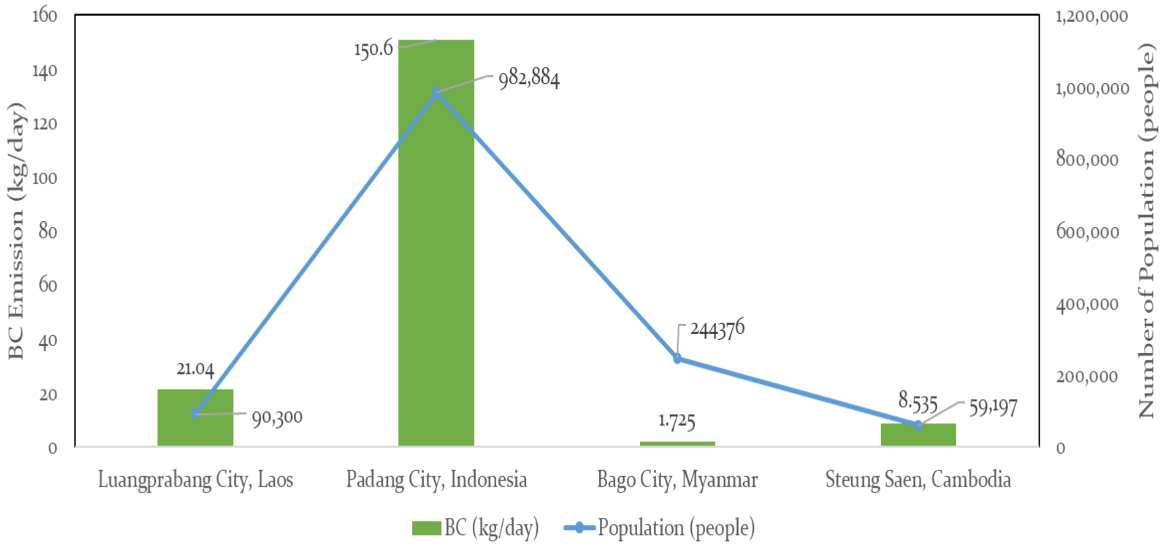

Figure 1.2 Reported BC emission of open waste burning at ASEAN Cities

1.5. Impact of OWB Practices

1.5.1. Potential Health Impact of OWB

Most of the respondents exposed to OWB feel that they are disturbed by the smoke and foul odor. Besides their educational status, people also notice the negative impact of OWB on the health issue (Oguntoke, 2019). OWB is also considered an inefficient combustion process due to a lack of oxygen supply and temperature control. Thus, the level of toxicity is higher than controlled incineration (Krecl et al., 2021a). The high level of exposure to PM can cause respiratory and cardiovascular disease and lead to cancer and adverse birth (Das et al., 2018b). Open burning of waste could also produce other gaseous and particle-bound compounds which have mutagenic nature, including polycyclic aromatic hydrocarbons (PAHs), polychlorinated dibenzo-p-dioxins (PCDDs), polychlorinated dibenzofurans (PCDFs), polychlorinated biphenyls (PCBs), and other harmful compounds (Hsu et al., 2016; Lundin et al., 2013). In the landfill site, PAHs are found to be the major contributor to air pollution. PAHs are also found in the soil, coming from residual ashes of OWB in the landfill site. The level of PAH concentration in the soil and ambient could be higher than the permissible limit (Adesina et al., 2020). Electrical equipment burning such as insulated wires, cables, circuit boards is also the highest source of dioxins, followed by plastic waste, garden waste, rubber waste, and mixed household waste (Zhang et al., 2017). Those pollutants are considered and classified as human carcinogens (Triassi et al., 2015). Therefore, OWB emission has a higher carcinogenic potential than wood combustion (Hoffer et al., 2020). OWB could emit particle-bound metals, which increases the possibility of cancer risk for people around the OWB incidents. The higher level of Zn, Pb, Ti, P, and Ba is found during waste burning in the landfill site (Baalbaki et al., 2018). Wang et al. (2017) stated that Zn (17.70 mg/kg) is produced at the highest level of emission at the landfill site in China, followed by Cu (3.78 mg/kg), As (1.30 mg/kg), and Pb (0.96 mg/kg). At those levels, short-term effects such as physical symptoms and physiological stress and long-term effects such as cancer, respiratory and cardiovascular symptoms may be experienced by people around waste burning incidents.

As mentioned earlier, biomass burning is the most topic found in the collected literature and is divided into 2 subtopics: forest fire and crop residue (See Table 1.5). This indicates the problem's urgency and impact on the residents, especially in Thailand and Vietnam. Besides, even though the data is limited, it was also found that municipal and e-waste burning have similar adverse health effects on human beings. Most of the literature reported the potential cancer risk from the metals and hydrocarbons bound particulate from a forest fire and crop residue burning (Pham et al., 2019; Sirithian et al., 2017; Wiriya et al., 2013). Besides, people inhaling the burned biomass waste smoke can receive several adverse respiratory infections such as ischemic heart diseases (IHD) and chronic obstructive pulmonary diseases (COPD) (Ha Chi and Kim Oanh, 2021; Punsompong and Chantara, 2018). Moreover, a few studies on burning municipal waste reported the possibility of cancer risk from dioxin exposure. According to Chi et al. (2022), dioxin contained in $PM_{2.5}$ can be released from municipal waste burning and contaminating the environment through ingestion pathway. The burning of e-waste can also emit flame retardant, which can affect brain development, fasten the growth of cancer cells, protein denaturation, and membrane cell malfunction. The metals bound particulate from the e-waste burning can also accumulate and give several adverse health effects to humans.

Table 1.5 Health impact of open burning.

Waste burned	Spesific sources / chemical characteristics	Pathway	Health impact	Study location	References
Biomass (forest and crop residue)	PM10-bound PAHs	Inhalation	Cancer risk from PAHs exposure	Thailand	Wiriya et al., 2013
Crop residue (maize)	PM10-bound PAHs	Inhalation	Cancer risk from PAHs exposure	Thailand	Sirithian et al., 2017
1. Forest fire 2. Crop residue	Black carbon	Inhalation	The same risk with passively smoked cigarette	Thailand	Pani et al., 2019
Forest fire	PM2.5	Inhalation	Cancer risk from PAHs exposure	Thailand	Chantara et al., 2020
Crop residue	PM2.5	Inhalation	Lower respiratory infections, ischemic heart diseases (IHD) = Long-term mortality / non-accidental deaths	Thailand	Ha Chi and Kim Oanh, 2021
Biomass (forest, crop residue, and grassland)	PM2.5	Inhalation	**Stroke burden,** ischemic heart disease (IHD), lung cancer (LC), and chronic obstructive pulmonary disease (COPD) = premature death	Thailand	Punsompong et al., 2021
1. Forest fire 2. Crop residue	n.d.	Inhalation	Respiratory disease, such as COPD and lung cancer	Thailand	Kaewrat et al., 2022
1. Forest fire 2. Crop residue	PM-bound PAHs during haze events	Inhalation	Respiratory health risks	Thailand	Insian et al., 2022
Crop (rice straw) residue	PM2.5-bound PAHs	Inhalation	Cancer risk from PAHs exposure	Vietnam	Pham et al., 2019
Open biomass burning (forest and crop residue)	PM2.5	Inhalation	Respiratory morbidity, ischemic heart disease (IHD), lung cancer (LC), chronic obstructive pulmonary disease (COPD), cardiovascular disease	Laos, Myanmar, Cambodia, Thailand, Vietnam	Thao et al., 2022
1. Municipal waste 2. Biomass (as fuel)	PM2.5-bound PCDD/Fs (dioxin)	Ingestion, diet	Cancer risk from dioxin exposure	Thailand, Vietnam, Taiwan	Chi et al., 2022
1. E-waste 2. Municipal waste	Flame retardant additives (for plastic or electronic additives)	Ingestion, inhalation, dermal	Autism, affect brain development, promote the growth of cancer cells, protein denaturation, membrane cells malfunction	n.d.	Chean-Yiing et al., 2022
E-waste (cables and wires for metal recovery)	Dioxin (PAHs), flame retardant, and metals	Inhalation, ingestion, diet	Non-cancer risk (bioaccumulation) caused by metals contamination. Other adverse effect related to the emitted pollutant.	Vietnam	Hoang et al., 2022

1.5.2. Potential Environmental Impact of OWB

Biomass burning is reported to have a significant contribution to global warming since it emits a lot of carbon dioxides (CO_2) and SLCPs such as BC and methane (CH_4) to the ambient air (See Table 1.6). The sequence of emission factor (EF) from the highest to the lowest for CO_2 emission is reported as follows: forest burning > sugarcane biomass > MSW > maize > crop residue burning. Therefore, the EF for the BC can be seen as follows: MSW (plastic) > rice straw > forest burning. In the mainland of SEA, Myanmar contributed the highest CO_2 and BC emissions, followed by Cambodia, Vietnam, Thailand, and Laos. However, based on the older reported data (2007), Indonesia can emit as much as Myanmar because of biomass burning (Permadi and Kim Oanh, 2013). It is estimated that the SEA countries contribute to the emission of CO_2 and BC around 172 Mt and 74 kt annually (Kim Oanh et al., 2018). Based on the literature's comparison, the BC from MSW burning in the Philippines has been reported to have a similar potential impact to BC emitted from rice straw residue burning in Vietnam. The number of emissions from MSW and e-waste burning has yet to be fully understood because the number of wastes burned has yet to be estimated.

Table 1.6 Environmental impact of open burning.

Study location	**Fuel Type**	**Emission Factor (dry biomass)**		**Total Annual Emission Estimation (Mt)**		**References**
		CO_2 (g kg^{-1})	**BC (g kg^{-1})**	**CO_2 (Mt)**	**BC (kt)**	
ASEAN	Crop residue (RS = rice straw, M = maize, S = sugarcane, OCR = another crop residue)	RS 1,177 M 1,350 S 1,130 OCR 1,130	RS 3.1 M 2.2 S 3.3 OCR 0.7	172	74	Kim Oanh et al., 2018
	Forest	-	-	655	220	
Thailand	Sugarcane biomass (pre- and post-harvesting)	1,515	-	Pre-harvest 9.80 Post-harvest 12.7	-	Sompoon et al., 2014
Indonesia	Rice straw	-	0.939 ± 0.417	-	-	Hafidawati et al., 2017
Thailand	Rice straw	1,177 ± 140	0.53	5.34	2 1 ± 1	Junpen et al., 2018
Vietnam	Rice straw	1,177	0.51	3.82	1.6	Le et al., 2020
Thailand	Rice straw	1,247 ± 190	-	8.23	-	Hong Phuong et al., 2022
Indonesia	Savanna / shrub land	1,613	0.48	*Total emission =* 57.28	*Total emission =* 0.24	Permadi et al., 2013 (2007 data)
	Peat land / mangrove forest	1,703	0.57			
	MSW	1,453	0.65			
	Tropical forest	1,580	0.66			

Study location	Fuel Type	Emission Factor (dry biomass)		Total Annual Emission Estimation (Mt)		References
		CO_2 (g kg^{-1})	BC (g kg^{-1})	CO_2 (Mt)	BC (kt)	
Thailand	Cropland	1,585	0.75	*Total emission* = Myanmar 64 ± 12 Cambodia 45 ± 8 Laos 13 ± 2 Thailand 27 ± 9 Vietnam 30 ± 12	*Total emission* = Myanmar 20 ± 12 Cambodia 13 ± 7 Laos 4 ± 3 Thailand 10 ± 5 Vietnam 13 ± 6	Junpen et al., 2020 (2015 data)
	Forestland	1,643	0.52			
	Shrubland / savanna	1,686	0.37			
Vientianne, Laos	MSW	1,453 (wet basis)	-	0.027	-	Babel and Vilaysouk, 2016
Luangprabang, Laos	MSW	1,453 (wet basis)	5.5	0.005	0.007	Vilaysouk and Babel, 2017
Depok City, Indonesia	MSW	801.2	-	0.26	-	Kristanto and Koven, 2019
Thailand	MSW	n/a (IPCC calculation)		0.499	-	Pansuk et al., 2018
Philippines	MSW	n/a (IPCC calculation)	0.65	944.69 (uncollected waste)	1.63	Premakumara et al., 2018

1.6. Factors Affecting OWB Practices

Looking up to the risks that may harm the urban health, there are some reasons that motivate people in developing countries to still burn their waste, including an erratic and unsorted waste collection services; a quick and inexpensive methods to clear their dump sites; a lack of environmental health awareness, attitude and practices of the OWB practices (Krecl et al., 2021; Remigios, 2013); a lack of motivation to sort their waste; no local regulation or policy which makes the people do not mind to doing OWB; organic decomposition which creates nuisance smell and attracts insects; a lack of space for waste dumping in the backyard; animals scavenging and disturbances; a lack of resources/time to transport the household waste to the waste collection services (Krecl et al., 2021); cost avoidance (Laghari et al., 2015; Lemieux, 1998) and an exceeding volume of waste due to some specific event (tree pruning, marriage, religious ceremony, or other events) (Papadakis et al., 2015). The environmental knowledge level of the people may be the most critical factor that affects the number of OWB incidents than other factors (Pasukphun, 2018).

There are also some constraints of the local government of developing countries to enhance their waste management services, including economic development and gross domestic product (financial standing); an inadequate transportation infrastructure: unpaved roads, potholes, old trucks, many more; distance between landfill/disposal site to the service area; and local people rejection of the establishment of waste collection temporary site (Lingan et al., 2014). Although waste collection services are provided in urban or suburban areas, people are still burning their waste due to their habit and impatience in waiting for the collection services. Therefore, raising public awareness through local campaigns, environmental education, environmental incentives, and regular inspection reinforcement are needed for lowering the waste burning events (Krecl et al., 2021). More robust policies and law reinforcement (such as OWB ban and 3R awareness endorsement) for a better solid waste management system are needed (Das et al., 2018).

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Chapter 2

Temporal Evaluation of Open Waste Burning (OWB) Activity: Semarang City as Case Study

2.1. Introduction

Proper waste management is becoming a primary concern for many municipalities in developing countries. Inadequate waste management systems lead to traditional open burning, burying, and random disposal (Velis and Cook, 2021), which are carried out at relatively higher levels in rural areas where waste collection services are unavailable (Remigios, 2013). In most rural areas of developing countries, open burning is most commonly practiced, instead of random dumping or disposal, recycling, and burying practice, by the local population (Han et al., 2015). In fact, this pattern was found in the rural part of Huejutla City, Mexico, where at least 22.4% of the waste is burned (Reyna-Bensusan et al., 2018). In the rural areas of Thailand (Pansuk et al., 2018), Southwest China (Han et al., 2015), and Iran (Vahidi et al., 2017), open burning is the dominant waste management practice, accounting for more than 30.0% of all practices. However, open burning of waste is also performed in urban areas in many developing countries as it is an easier option for eliminating waste. For instance, in the urban area of Kampala City, at least 13% of the population burns their waste (Kulabako et al., 2010). Reducing the number of dumping and burning practices is part of the international strategic objectives that must be achieved to meet the sustainable development goals (SDGs) by 2030; thus, reducing these practices is an important task.

Open-burning municipal solid waste (MSW) processes are inefficient owing to limited oxygen supply and poorly controlled temperature. This incomplete combustion results in toxin emission, such as particulate matter (PM), carbon monoxide (CO), and other gases, into the atmosphere without any air pollution control (Krecl et al., 2021). Occasionally, the open burning of MSW contains considerable plastic waste, which is the most significant source of dioxins and other halogenated compounds (Nagpure et al., 2015). Pansuk and his team reported that plastic waste is the second-highest waste in rural Thailand (31.7%). Some primary toxic aerosols, such as smoke and carbonaceous compounds, are released, thereby polluting the environment and harming human health (Pansuk et al., 2018). Open burning may thus significantly contribute to air pollution compared to emissions from the transportation and industrial sectors (Wiedinmyer et al., 2014). An emission inventory is needed to identify suitable methods to control pollution and better understand the negative effects of open waste burning. However, an open burning activity data inventory, which may divert from the evaluation system and enable the implementation of laws and policies related to reducing open burning practices, is lacking (Nagpure et al., 2015; Permadi and Kim Oanh, 2013).

Most of the mass estimation for open burning is derived from questionnaire-based survey and literature-based assumptions, which either results in an overestimation or underestimation of the open burning incident itself. Therefore, some researchers have employed another approach to derive the best results for burned MSW mass estimation. A team of researchers led by Nagpure, Raj, and Ramaswami used transect walks to determine the number of active burning piles, and the social and infrastructural factors affecting open burning, as well as estimate the number of illegal dumping of MSW and its physical characteristics in India (Lal et al., 2016; Nagpure et al., 2015; Nagpure, 2019; Ramaswami et al., 2016). Das et al. (2018) employed a different approach by combining household survey and the transect walk method to validate the P_{frac} value of the IPCC calculation method (fraction of people burning waste in a household). In a recent study, Krecl et al. (2021) used a transect walk survey principle to identify fire spots in specific areas. Overall, more field estimation studies regarding open waste burning are required to assemble an appropriate emission inventory for a specific country. In this study, the amount of unmanaged waste in Semarang City, Indonesia, was determined. Due to the lack of high-level (tier) data inventory, especially in open waste burning, waste pile composition and characteristic analyses in two different seasons were conducted in this study. The information presented in this study will be essential for evaluating policy and law interventions, and other potential future research benefits related to open waste burning (OWB).

2.2. Materials and Methods

This study sought to estimate burning activities, incidents, and emissions in the selected sub-district area of Semarang City. The transect walk survey methods were modified from previous methods employed in India, Mexico, and Nepal (Das et al., 2018; Nagpure et al., 2015; Reyna-Bensusan et al., 2019). The laboratory test used to determine waste composition was carried out according to Nagpure et al. (2015).

2.2.1. Study Area

Semarang City, the capital city of Central Java Province, is considered a metropolitan city as it was one of the top six cities with the highest gross domestic product (GDP) in Indonesia in 2019. The GDP per capita of Semarang City reached 105.59 million rupiahs and is constantly increasing by approximately 7% each year (Syafrudin et al., 2021). Semarang City is also considered an urban coastal city as it is located south of the Java Sea. Semarang City consists of 16 sub-districts divided into 177 sub-districts, with Wonolopo as the largest sub-district (area = 1,459.53 ha), and Sukorejo as the smallest sub-district (area = 15 ha). Based on the following background, Semarang City might generate more waste than other cities. Waste generation is reported to increase by 2%–4% each year and Semarang City is estimated to produce 606,728 tons of waste annually. This waste is dominated by organic waste (53.86%), followed by plastic (21.52%), paper (10.97%), metals (8.72%), and other products (4.93%) (Hadiwidodo et al., 2019). Most of the waste in the city is generated from households (76%), market (14%), industry (4%), and others (6%) (Pertiwi et al., 2018). It is estimated that 4.54% of the waste is recycled through informal actors in Semarang City. Plastic is becoming the most recovered and recycled waste (53%–56%) compared to paper, metals, glass, and others (Pertiwi et al., 2018; Syafrudin et al., 2021). According to the Semarang City Government, 77.75% of municipal waste is processed at the landfill site, 17.65% of waste is processed at the source in material recovery facilities available in some districts, and 4.60% of waste is burned, buried, and disposed directly into the environment. The amount of waste collected in 2019 was estimated to be 390,915 ton/year. The researchers used k-means cluster methods to obtain four different clusters with similar characteristics. Each cluster was identified and named using the definition of urban area classification, such as rural, outer peri-urban, inner peri-urban, and urban, by Hanna Karg and her team (Karg et al., 2019). Figure 2.1 describes the position of each selected sub-district (transect area) on the Semarang City Map.

2.2.2. Online Questionnaire Survey

An online questionnaire survey was conducted to determine the current waste disposal practices and open burning potential in Semarang. The questionnaire comprised nine questions, including the name of the respondent (secured as privacy), sub-district where they live, number of family members, number of family members who burned their waste, burning frequency (daily), common waste disposal practices, availability of waste collection services, availability of door-to-door collection vehicles, and frequency of waste collection. According to Semarang City Statistical Agency, the total population of Semarang City is 1,656,564. Thus, using the formula shown by Hu et al. (2019), the sample size at a 95% confidence level and margin of error of 5% can be determined as much as 385, which later becomes the minimum data amount. Therefore, the questionnaires were distributed to 408 citizens. However, after data cleaning, answers from only 344 respondents were selected for analysis because of completeness and validity. Descriptive analysis was applied to the questionnaire data. The number of respondents (r_{a-d}) from rural, outer peri-urban, inner peri-urban, and urban areas was 86, 85, 89, and 84, respectively. To determine the average number of family members (FM_{OB}) who burned their waste and occurrence possibility of a burning waste event (BE_{a-d}) in each cluster, Eq. (1) and (2) were used.

$$\sum_{a-d} FM_{OB} = \frac{FM_{OB_{a-d}}}{r_{a-d}} \quad (1)$$

$$BE_{a-d} = 1 - \left(\frac{\left(\frac{\sum_{a-d} BF}{r_{a-d}} \right)}{90} \right) \quad (2)$$

In equation (2), BF is burning frequency, which was defined daily, and 90 represents the maximum day of burning frequency. If the respondents reported no burning frequency in a specific area, BF was considered 90.

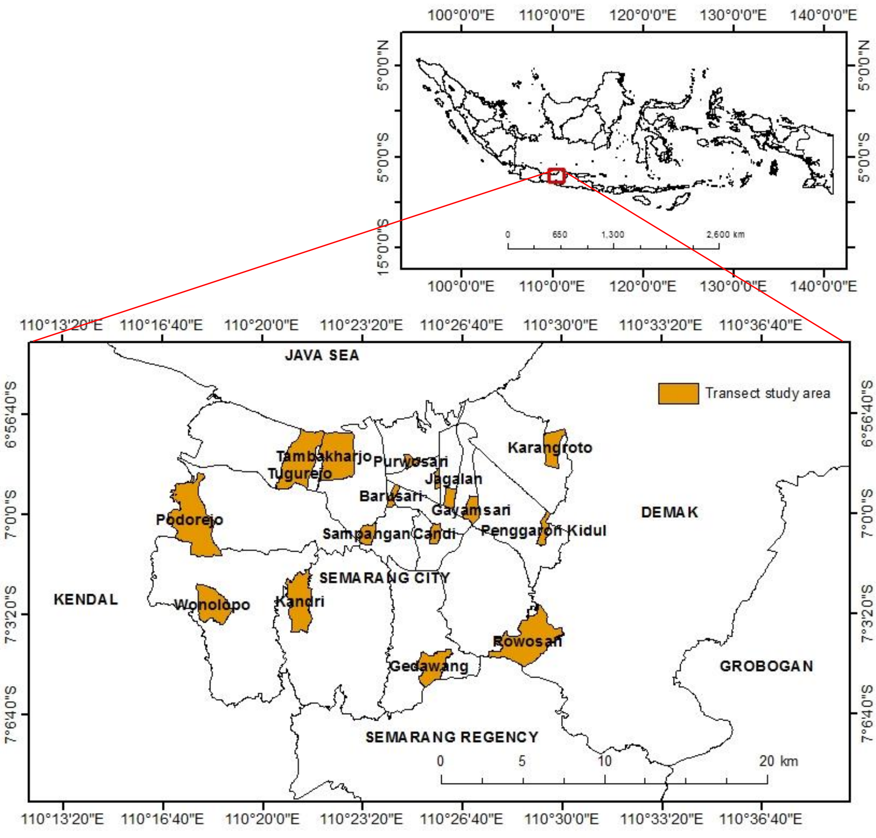


Figure 2.1 Semarang City maps and transect study areas

2.2.3. Transect Walk Survey

The transect walk survey methods follow those employed in a previous successful study by Das et al. and Nagpure et al. (Das et al., 2018; Nagpure et al., 2015). The transect walk routes were determined randomly for each sub-district belonging to the four clusters mentioned above. Each route was approximately 10 km long and could either be a neighborhood loop or a straight line. The survey was conducted in the rainy season from mid-January to mid-February 2021 and in the dry season from May to July 2021, both during the semi-lockdown policy for COVID-19 in Semarang City. Preliminary surveys were also conducted to ensure the performance of open burning at the household and landfill sites. The surveyors were well prepared and equipped with a mask, gloves, handheld GPS, and a camera. The surveyors asked the local people about their burning practice frequency once during the transect. This field-based experiment was carried out in the morning and afternoon on two different days (four-time surveys). The total number of piles was the sum of the piles found from the first to the fourth survey. During the transect walk survey, the surveyor recorded the waste pile coordinates, dimensions (estimated width, length, and height using measure tape and stick), distance from road/place perpendicular to the road, photos, and conditions (currently burn, burned, half-burned, or not burned). Waste piles that were not burned were categorized as potentially unmanaged waste, buried, fed to animals, or other potential waste practices. Landfills were not considered as burning sources, as there were no reported waste burning incidents.

The transect results (in volume) were converted into a weight basis after the specific density of the waste piles was determined. Each route's estimated weight was multiplied by the transect area to determine the pile density (ton/km^2). The details of equation are as follows.

$$\sum M_a = \sum V_a \ x \ \rho_a \quad (3)$$

$$TrA_a = TrL_a \ x \ SS \quad (4)$$

$$M_b = TrA_a \ x \sum M_a \quad (5)$$

where M_a and V_a are the weight (kg) and volume (m^3) of the waste burned in each district, respectively; ρ_a is the compaction density of the piles (kg/m^3); TrA_a and TrL_a are the transect area (m^2) and transect line (m) of the specific surveyed area (in each sub-district), respectively; SS is the maximum sightseeing (m); and M_b is the weight estimation of the burned waste in each sub-district (ton/day). Each cluster's average pile density was scaled up to the city level by multiplying the pile weight (M_b) by the total area of each cluster-covered area. The average pile density of each cluster was also multiplied by the population density (P_D) to determine the coarse estimation of burned waste per capita. Semarang City waste generation was estimated by assuming 3.74 l/person/day of waste per capita, 245 g/l of waste density (Budihardjo and

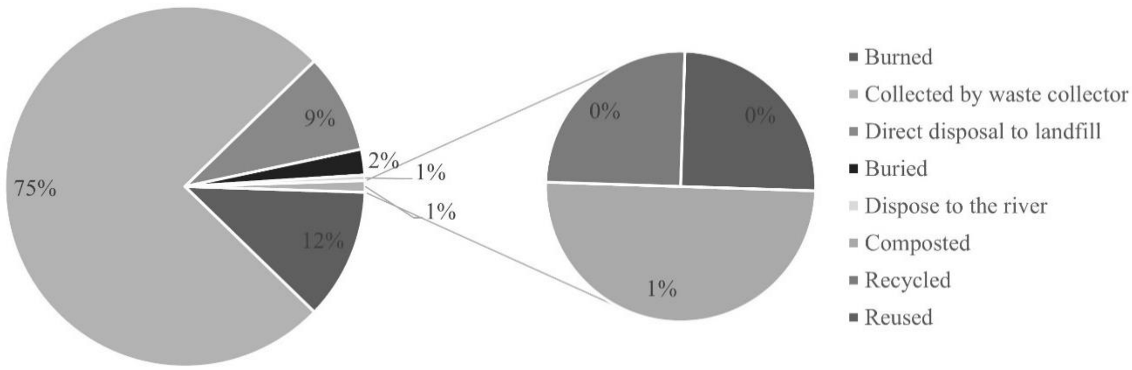
Wahyuningrum, 2018), and 1,814,110 persons of the Semarang City population in 2019. Information regarding the collected waste sent to the landfill was obtained from the Environmental Services Government of Semarang City.

2.3. Results and Discussion

2.3.1. General Information of OWB Practices in Semarang City

From our randomized questionnaire survey, OWB was found to be the second most common waste disposal practice. As shown in Figure 2.2(a), other improper practices, such as burying and direct disposal in the river, also exist. The proportion of composting, recycling, and reuse was small (2% in total), indicating the presence of a linear or conventional system (collection, transport, and disposal) in the city. The door-to-door waste collection in Semarang is managed by each neighborhood unit (NU) or association (NA), which gathers waste from households and brings it to the waste collection site (WCS). The municipal government manages the transportation of waste from the waste collection sites to landfills. From areas near the landfill, the door-to-door waste collection vehicle directly brings the collected waste to the landfill. Each NU/NA has a different waste collection system, as shown in Figure 2.2(b); three-wheeled motorcycles are the major household waste collection vehicles in Semarang. Waste from some areas is not collected by vehicles and burning or direct disposal of waste into the environment are the common practices in such areas. Respondents from the rural and outer peri-urban areas are most likely to burn their waste rather than bringing it to the nearest WCS.

(a)



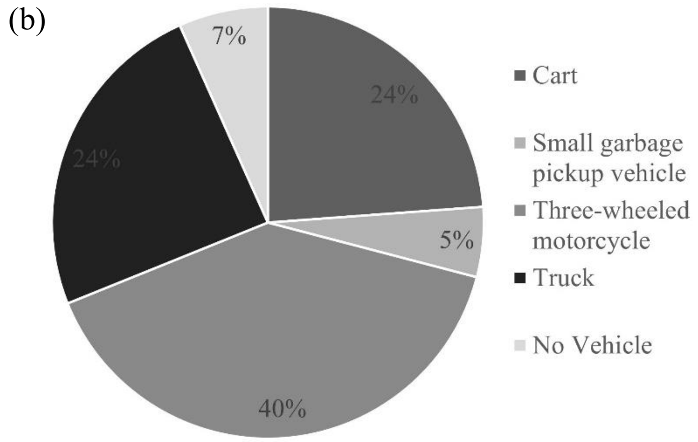


Figure 2.2 (a) Common waste disposal practices and (b) door-to-door waste collection service vehicles in Semarang.

Figure 2.3 presents some interesting findings regarding waste burning practices in Semarang. The proportion of people in families who burned waste was higher in rural areas than in other areas. This may be attributed to the lower waste collection frequency and availability of larger backyards in rural areas. Therefore, the higher the frequency of waste collection, the lower the possibility of waste burning or other improper waste disposal practices. A higher proportion of family members burning their waste implies that the practice has already become a habit for residents in rural areas. However, the present survey was based on an online questionnaire that was open to random citizens in the city, and the possibility of bias may therefore be high. Next, the transect walk survey was undertaken to precisely identify and model waste burning events in Semarang.

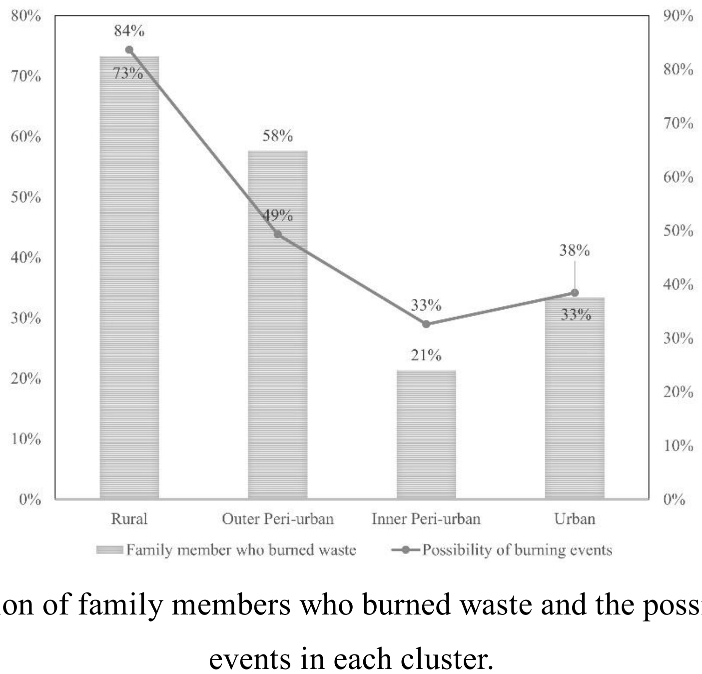


Figure 2.3 Fraction of family members who burned waste and the possibility of burning events in each cluster.

2.3.2. OWB Practices in the Different Seasons

During the transect walk, visual observation was conducted to determine the composition of the waste burned (see Table 2.1). In Semarang City, backyard waste consisting of branches, twigs, and leaves is the main waste burned, accounting for 62.7% and 73.61% in rainy and dry season of the total burned waste. Plastic waste was always the second-largest contributor to waste burned after backyard waste in all areas and all seasons. Inert materials, such as glass and metals, were detected in outer peri-urban area, or the highest percentage compared to other region, indicating a lower burning efficiency in this region. People in rural areas are more likely to burn organic matter, wood, or branches than those in other areas. The highest burning of backyard waste was noted in rural areas. This may be attributed to the availability of larger backyards in rural areas than in other areas. However, the proportion of plastic in burned waste tended to be higher in outer and inner peri-urban areas.

Then, the greatest contributor to plastic waste burning was the inner peri-urban area. Interestingly, outer peri-urban area showed a greater diversity of waste being burned since almost all the waste composition except glass and miscellaneous exceed 10% of total composition. In the dry season, nonetheless, among burned waste, organic waste (leaves > yard waste > food waste) and LWTR (wood > textiles > leather > rubber) accounted for the highest share, followed by plastic, paper, and other waste/miscellaneous. Polyethylene terephthalate (PET, 5.4%) contributed to the highest burned plastic fraction in Semarang, followed by low-density polyethylene (LDPE, 4.9%), polyvinyl chloride (PVC, 3.5%), high-density polyethylene (HDPE, 1.4%), polypropylene (PP, 0.2%), polystyrene (PS, 0.1%), and other plastic waste that does not belong to or is a combination of the other six categories of plastic waste, such as bisphenol A polycarbonate (PC), or bioplastics (0.2%). The average compaction density was 223 and 90 kg/m^3 in rainy and dry season, and the average moisture content was 43.13%, respectively. Although all sub-districts were in a similar city, a significant variation in the geographical boundary, socioeconomic activities, and lifestyle was found, which led to different densities and compositions. Therefore, the burned waste was assumed to have a relatively high combustible fraction value (average - 0.72).

Table 2.1 Burned waste composition

Waste Composition	**Cluster 1 (Rural)**		**Cluster 2 (Outer Peri-urban)**		**Cluster 3 (Inner Peri-urban)**		**Cluster 4 (Urban Core)**		**Semarang City**	
	Dry	**Rainy**	**Dry**	**Rainy**	**Dry**	**Rainy**	**Dry**	**Rainy**	**Dry**	**Rainy**
Paper	0.4%	9.5%	17.0%	3.0%	1.7%	2.2%	0.7%	2.7%	4.9%	4.3%
Plastic	5.8%	4.3%	16.3%	7.5%	25.0%	42.1%	15.8%	15.9%	15.7%	17.5%

Waste Composition	Cluster 1 (Rural)		Cluster 2 (Outer Peri-urban)		Cluster 3 (Inner Peri-urban)		Cluster 4 (Urban Core)		Semarang City	
	Dry	**Rainy**	**Dry**	**Rainy**	**Dry**	**Rainy**	**Dry**	**Rainy**	**Dry**	**Rainy**
Organic	61.8%	19.8%	17.9%	5.1%	35.5%	42.2%	49.7%	46.6%	41.2%	28.4%
LWTR	30.9%	66.4%	17.7%	82.6%	4.5%	11.4%	32.7%	34.1%	21.5%	48.6%
Glass	0.0%	0.0%	7.9%	0.0%	0.0%	0.0%	0.0%	0.0%	2.0%	0.0%
Metals	1.3%	0.0%	14.9%	0.6%	7.1%	0.1%	0.0%	0.3%	5.8%	0.3%
Miscellaneous	0.0%	0.0%	8.2%	1.2%	26.3%	2.0%	1.1%	0.5%	8.9%	0.9%

A total of 154 (dry) and 171 (rainy) piles were identified during the transect walk survey at the household level. As shown in Table 2.2, the highest number of piles was found in the Karangroto sub-district (inner peri-urban area) in the rainy season and Podorejo sub-district (rural area) in the dry season, while the lowest pile number and pile density were found in the Jagalan and Barusari sub-districts (urban area) and Candi sub-district (urban area) in the rainy and dry season, respectively. Notably, the total piles in each cluster showed a different pattern, with the rural area displaying the highest number of piles. The average pile density shows a sequential order from the highest to the lowest pile density from rural to urban areas. Interestingly, the inner or outer peri-urban area, also called the transition area, had the highest number of open burning, thereby differing from the results of previous research (Reyna-Bensusan et al., 2018). Only 19.33% and 11.38% of the total waste piles in the rainy and dry season in the transect areas were not burned during visual inspection. Therefore, the highest burning intensity was found in the inner peri-urban area, which aligns with a previous finding that peri-urban areas contribute the most to open burning in Semarang City.

Table 2.2 Physical profile of waste piles found in the transect study area

Sampling site		Sub-district	Coarse estimate of volume (m^3)	Coarse estimate of the weight (kg)	Transect area (km^2)	Total piles		Piles density (ton/km^2)	Average piles density (ton/km^2)	Percentage of burning incidents (%)	
Cluster 1 (Rural)	Rainy	Wonolopo	0.64	19.22	0.2100	21	65	0.09	0.76	71	80.65
		Podorejo	1.02	61.41	0.1450	11		0.42		100	
		Rowosari	1.52	91.12	0.0500	21		1.82		76	
		Tugurejo	0.31	11.43	0.0160	12		0.78		75	
	Dry	Wonolopo	0.22	56.88	0.0500	18	58	1.14	1.17	78	79.35
		Podorejo	0.23	57.76	0.0700	22		0.83		95	
		Rowosari	0.09	17.20	0.0800	7		0.21		71	
		Tugurejo	0.25	74.65	0.0300	11		2.49		73	
Cluster 2 (Outer periurban)	Rainy	Penggaron Kidul	0.29	58.20	0.0471	10	44	1.24	0.66	80	68.08
		Kadri	1.07	214.69	0.3000	18		0.72		100	
		Tambakharjo	0.02	2.79	0.0156	3		0.18		0	
		Gedawang	0.60	29.99	0.0570	13		0.53		92	
	Dry	Penggaron Kidul	0.05	18.68	0.0200	4	35	0.93	0.74	100	96.26
		Kadri	0.35	13.30	0.0500	17		0.27		94	
		Tambakharjo	0.20	27.45	0.0200	3		1.37		100	

Sampling site		Sub-district	Coarse estimate of volume (m^3)	Coarse estimate of the weight (kg)	Transect area (km^2)	Total piles		Piles density (ton/km^2)	Average piles density (ton/km^2)	Percentage of burning incidents (%)	
		Gedawang	0.10	24.40	0.0600	11		0.41		91	
Cluster 3 (Inner periurban)	Rainy	Gayamsari	0.07	7.92	0.0254	7	50	0.31	0.62	100	90.63
		Karangroto	0.45	57.81	0.0450	24		1.30		88	
		Karang Tempel	0.29	17.38	0.0250	15		0.70		100	
		Sampangan	0.06	3.09	0.0170	4		0.22		75	
	Dry	Gayamsari	0.09	28.11	0.1000	7	33	0.28	0.59	86	78.88
		Karangroto	0.09	33.61	0.0700	9		0.48		89	
		Karang Tempel	0.11	30.88	0.0200	6		1.54		50	
		Sampangan	0.02	1.63	0.0300	11		0.05		91	
Cluster 4 (Urban Core)	Rainy	Jagalan	0.00	0.17	0.0145	1	12	0.01	0.09	100	83.33
		Barusari	0.02	1.75	0.1280	1		0.01		100	
		Candi	0.02	2.15	0.0250	6		0.09		83	
		Purwosari	0.06	4.89	0.0195	4		0.25		50	
	Dry	Jagalan	0.02	3.98	0.0200	7	28	0.20	0.21	100	100
		Barusari	0.05	14.92	0.0500	10		0.30		100	
		Candi	0.01	0.38	0.0150	4		0.03		100	
		Purwosari	0.05	12.71	0.0400	7		0.32		100	

In the per-capita context, rural areas were found to have the highest burning incidents compared to other areas (see Table 2.3). Each person can be estimated to burn 0.539 – 1.098 kg of waste per day; however, a lower number was found in urban areas. Therefore, this estimated result aligns with the collection points available in each cluster. For instance, a lower waste collection efficiency in the rural cluster has been reported, enabling a higher possibility of open burning practice. In the peri-urban area, the number of burning incidents per capita was lower than that in rural areas, indicating that an appropriate number of waste collection units and services is provided in the area. Therefore, a high level of waste collection services, population density, and environmental awareness in urban areas may reduce the possibility of burning incidents.

Table 2.3 Estimation of burning intensity per capita in each sub-district and cluster

Sampling		Sub-district	Population (capita)	Areas (km^2)	Coarse estimation of burning intensity (kg waste/capita/day)			
					Non-burning incidents	Average	Burning incidents	Average
Cluster 1 (Rural)	Rainy	Wonolopo	9,864	14.60	0.026	0.129	0.109	0.539
		Podorejo	9,376	9.72	0.085		0.354	
		Rowosari	12,381	8.70	0.248		1.033	
		Tugurejo	7,550	8.63	0.158		0.659	
	Dry	Wonolopo	9,864	14.60	0.348	0.286	1.336	1.098
		Podorejo	9,376	9.72	0.177		0.679	
		Rowosari	12,381	8.70	0.031		0.120	
		Tugurejo	7,550	8.63	0.587		2.256	

Sampling		Sub-district	Population (capita)	Areas (km²)	Coarse estimation of burning intensity (kg waste/capita/day)			
					Non-burning incidents	Average	Burning incidents	Average
Cluster 2 (Outer periurban)	Rainy	Penggaron Kidul	7,202	2.53	0.138	0.083	0.295	0.176
		Kandri	4,827	2.45	0.116		0.248	
		Tambakharjo	3,297	1.67	0.029		0.062	
		Gedawang	9,598	2.70	0.047		0.101	
	Dry	Penggaron Kidul	7,202	2.53	0.012	0.012	0.315	0.306
		Kandri	4,827	2.45	0.005		0.130	
		Tambakharjo	3,297	1.67	0.026		0.669	
		Gedawang	9,598	2.70	0.004		0.110	
Cluster 3 (Inner periurban)	Rainy	Gayamsari	12,385	0.93	0.002	0.009	0.021	0.088
		Karangroto	14,015	2.06	0.018		0.171	
		Karang Tempel	3,942	0.92	0.015		0.147	
		Sampangan	10,623	0.97	0.002		0.015	
	Dry	Gayamsari	12,385	0.93	0.004	0.024	0.017	0.090
		Karangroto	14,015	2.06	0.015		0.056	
		Karang Tempel	3,942	0.92	0.076		0.284	
		Sampangan	10,623	0.97	0.001		0.004	
Cluster 4 (Urban Core)	Rainy	Jagalan	5,811	0.27	0.000	0.001	0.000	0.004
		Barusari	6,151	0.40	0.000		0.001	
		Candi	11,595	0.59	0.001		0.004	
		Purwosari	8,898	0.48	0.002		0.011	
	Dry	Jagalan	5,811	0.27	-	-	0.009	0.012
		Barusari	6,151	0.40	-		0.019	
		Candi	11,595	0.59	-		0.001	
		Purwosari	8,898	0.48	-		0.017	

2.3.3. Scaling up the Transect Walk Results

Data on the amount of burned waste and its composition are essential to provide scientific evidence and establish appropriate waste management systems and policies (Haywood et al., 2019). After the amount of waste burned in each cluster was estimated, the average waste burned density in the cluster was multiplied by the total area of each cluster in Semarang City. The outer peri-urban area was the largest contributor to open burning, with 50.82% of the total waste burned in Semarang City. The lowest estimate for waste burning was found in the urban core, with only 2.74% of the total waste burned or 0.27% of the total waste generated in Semarang City. As shown in Table 2.4, the estimation number may align with that of other previous studies, such as studies in Nepal and India, where the city core was found to only contribute a maximum of 2% of the total waste generated in the city (Das et al., 2018; Nagpure et al., 2015). The urban areas of Semarang City present a lower waste burning percentage (9.7%) compared to other cities such as Vientiane City, Laos (15%), Steung Saen Municipality, Cambodia (21.2%), Padang City, Indonesia (11.5%), and Agra, India (24.2%)

(Babel and Vilaysouk, 2016; Menikpura et al., 2022); Nagpure et al., 2015. In the present study, the amount of waste burned per capita in the urban areas of Semarang was the same (0.012 kg.day^{-1}) as that reported in the Kathmandu Valley. Meanwhile, the amount of waste generated per capita in the Kathmandu Valley (0.40 kg.day^{-1}) was half of that generated in Semarang (Das et al., 2018a). Conversely, despite the similar amount of waste generated per capita (0.85 kg.day^{-1}), the amount of waste burned per capita was higher in urban area of Mexico (0.048 kg.day^{-1}) than in Semarang. However, the amount of waste burned per capita in rural areas of Mexico (0.280 kg.day^{-1}) was lower than that in the rural areas of Semarang (1.098 kg.day^{-1}) (Reyna-Bensusan et al., 2018). This finding is interesting because rural areas represent a higher burning intensity, thereby acting as a hotspot of open fires in the city.

Table 2.4 Comparative estimation of open waste burning incidents with other municipal scale studies

Municipal	**Population (person)**	**Amount of waste (ton.day^{-1})**	**Estimated amount of waste burning (ton.day^{-1})**	**Avg Burning Percentage (%)**	**Waste burning per capita (kg.day^{-1})**	**References**
Semarang City, Indonesia	1,656,564	1,662	Urban = 4.41 - 11.26 Inner peri-urban = 39.50 - 41.39 Outer peri-urban = 81.90 - 100.91 Rural = 35.35 - 59.38 Overall = 161.17 - 212.94	9.7 (rainy) 12.81 (dry)	Urban = 0.004 - 0.012 Inner peri-urban = 0.088 - 0.090 Outer peri-urban = 0.176 - 0.306 Rural = 0.539 - 1.098 Overall = 0.202 - 0.376	This study
Kathmandu valley municipalities, Nepal (2016)	1,751,114	2,060	20	3.0	Urban = 0.003 – 0.014 Peri-urban = 0.008 – 0.027 Overall = 0.012	Das et al. (2018)
Municipality of Huejutla, Mexico	122,905	64	Urban = 0.163 – 0.488 Peri-urban = 0.929 – 1.895 Rural = 23.243 Overall = 23.263	36.3	Urban = 0.048 Peri-urban = 0.063 Rural = 0.280 Overall = 0.189	Reyna-bensusan et al. (2018)
Depok City, Indonesia	2,484,000	1,120	70	6.3	0.028	Kristanto and Koven (2019)
Vientiane City, Laos	731,118	637	95.55	15.0	0.131	Babel and Vilaysouk (2016)
Luangprabang City, Laos	90,300	57	5.13	9.0	0.057	Vilaysouk and Babel (2017)
Bago City, Myanmar	244,376	99	2.07	2.1	0.008	Menikpura et al. (2022)

Municipal	Population (person)	Amount of waste ($ton.day^{-1}$)	Estimated amount of waste burning ($ton.day^{-1}$)	Avg Burning Percentage (%)	Waste burning per capita ($kg.day^{-1}$)	References
Steung Saen Municipality, Cambodia	59,197	35.5	7.54	21.2	0.127	Menikpura et al. (2022)
Padang City, Indonesia	105,577	661	71.8	11.5	0.680	Menikpura et al. (2022)
Delhi, India	16,700,000	8,390	190 - 246	2.9	0.014	Nagpure et al., 2015
Agra, India	1,960,000	923	223	24.2	0.113	
Agra, India	1,960,000	1,136	261.46	23.0	0.130	Lal et al., 2016

OWB practices are dominant in rural and outer peri-urban areas of Semarang because of the lack of waste collection services. Reyna-Bensusan et al. (2018) stated that regular waste collection and availability of waste collection facilities can reduce the intensity of OWB in urban and peri-urban areas. These speculations are consistent with the reports of Nagpure et al., (2015), who recorded the highest number of burning incidents in areas with a low socioeconomic status (SES) in India. Low-SES areas are similar to rural or peri-urban areas, which have a larger area but a lower population density. Typically, waste collection in such areas is extremely limited, and larger backyards are available at the household level. These contrasting socioeconomic profiles result in different burning profiles in selected study areas (Ramadan et al., 2022b). For instance, in rural areas of Mexico, over 65% of the total generated waste is burned, which is comparable to the amount of waste burned in rural areas of Semarang; meanwhile, <10% of waste in urban areas is burned (Reyna-Bensusan et al., 2018c). In addition, differences in lifestyle, income, and resources result in diverse waste disposal patterns (Mihai et al., 2021). In the present study, rural and outer peri-urban areas were the largest contributors to open burning, where 10% of the total generated waste was burned, assuming that the total waste generation of 1,662 tons per day in Semarang according to the calculation of Ramadan et al., (2022b). Furthermore, intensive OWB may be driven by the lack of law enforcement and environmental knowledge among the residents. Residents are often unaware of the legal consequences of OWB (Mihai et al., 2021). In fact, OWB is a common waste disposal practice following waste collection by local authorities, such as in Indonesia (Ramadan et al., 2022b), South Africa (Haywood et al., 2019), Mexico (Reyna-Bensusan et al., 2018c), India (Nagpure et al., 2015a), Nepal (Das et al., 2018), Eswatini, and Ghana (Nxumalo et al., 2020). Residents tend to burn their uncollected waste rather than burying or disposing it off into water streams because (1) they do not have any other option to manage the generated waste and (2) it is easy to eliminate waste from their sight. Therefore, realizing that OWB is dangerous and damage their property may be one of the motives to prevent OWB (Nxumalo et al., 2020; Ramadan et al., 2022a).

The composition of burned waste in outer peri-urban areas differed between the dry and rainy seasons. In the rainy season, the proportion of plastic waste was the lowest in burned waste. Overall, however, seasons did not significantly change the composition of waste being burned (Ramadan et al., 2022b). In addition, waste composition is an appropriate tool for estimating GHG and particulate emissions and predicting the potential risks to citizens. For instance, burning of HDPE and other types of plastics may emit CO_2, CO, NO_2, SO_2, and PM (Nxumalo et al., 2020). However, the composition of burned waste shapes the extent of risks and amount of contaminants released. Therefore, the inventories of emissions differ across cities or countries (Park et al., 2013; Reyna-Bensusan et al., 2019).

2.4. Summary

According to the findings of this study, approximately 161.17 – 212.94 tons/day of municipal waste is burned in Semarang City, ultimately accounting for 9.70% - 12.81% of the total waste generated in the city. The outer peri-urban area cluster had the highest contribution to open burning, representing 47.39 - 50.82% of the total open burning incidents. Further, branches, twigs, and leaves were identified as the most numerous burned components, followed by plastic both in the two seasons, which pose significant risks to human health. Interestingly, the inner peri-urban and urban areas were found to have more plastic waste for burning, despite having a significantly lower number of piles than the outer peri-urban area. Based on coarse estimation per capita, the highest burning incidents per capita were found in the rural areas of Semarang City, followed by the outer peri-urban, inner peri-urban, and urban areas. Approximately 80.67 – 88.62% of the piles were burned while 11.38 – 19.33% were unburned. The unburned pile can be assumed to be buried, dumped, or disposed directly into the environment. This finding aligns with that of previous research where rural areas were found to have more per capita waste burning incidents than urban areas. In addition, the number of mismanaged wastes was 3-fold higher than the local government estimates.

2.5. References

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Chapter 3

Environmental and Health Risk Assessment of OWB Practices

3.1. Introduction

Open waste burning (OWB) is a potential source of emissions in major cities of many low- and middle-income countries (Das et al., 2018; Lal et al., 2016; Nagpure et al., 2015). OWB releases many hazardous compounds that may pose risks to the public and environment around the burning areas (Powrie et al., 2021). OWB is an adverse practice to several sustainable development goals (SDGs), such as goal numbers 3 (good health and well-being), 6 (clean water and sanitation), 11 (sustainable cities and communities), and 12 (responsible consumption and production) (Mihai et al., 2021). This practice is commonly undertaken in areas that are not covered by waste collection services, along with other disposal practices, such as burial or dumping on the open ground or water surface (Reyna-Bensusan et al., 2019). Some open dumpsites with diverse waste characteristics may be burnt uncontrollably. For instance, a recent study reported that open burning at a Nigerian landfill site contributes to the emission of greenhouse gases (GHGs) into the atmosphere (Daffi et al., 2020). Additionally, according to Sharma et al. (2022), the largest contributor of particulate matter (PM) emissions in India may be OWB by 2035 in the case of lack of appropriate political intervention. Therefore, these severe threats should be treated appropriately to reduce the possibility of other accidents.

According to Ramadan et al. (2022a), the environmental and health risks of OWB exposure have attracted much research attention. OWB emits greenhouse and trace gases, PM, black carbon (BC), and other bound compounds (Wiedinmyer et al., 2014). The IPCC 2006 methodologies have been extensively used to calculate the environmental impact of OWB practices. However, BC is not considered in these calculations and must be quantified using a separate procedure (Premakumara et al., 2018). BC produces a greater environmental impact than carbon dioxide or methane (Reyna-Bensusan et al., 2019). Furthermore, OWB emissions are more dangerous because of their potential to emit hydrocarbons and metal-bound particulates (Chi et al., 2022). A few polycyclic aromatic hydrocarbons (PAHs) can be released during the burning process because of the presence of plastic waste that is burned together with other domestic waste (Hoffer et al., 2020). PAHs can be released through volatilization and can bind PM (Hubai et al., 2022; Velis and Cook, 2021). Importantly, some typical heavy metals with carcinogenic risk, such as Pb, Ni, and Cd, may be bound to fly ash (FA) generated from

0.01 to 14.16 mg·kg^{-1} of burned waste (Park et al., 2013), posing carcinogenic and chronic health risks.

Several studies have reported the health effects of the OWB. Velis and Cook (2021) has been specifically reviewed about the health risks of open burning of plastic waste. In that study, dioxins, and related compounds (DRCs), bisphenol A (BPA), and PAHs were identified during the OWB incidents and informal recyclers are susceptible to high risk from direct inhalation and ingestion. Shih et al. (2015) estimated that OWB at landfill site increase PCDD/F concentrations in the environmental media. Several cancer deaths reported in Nairobi can be related to the dioxin emissions from OWB. However, estimation of cancer posed by dioxin differs among the populations, uncertainty occurred in the estimation. Kodros et al. (2016) made estimation related to the global mortalities to the ambient $PM_{2.5}$ emissions from OWB. The results interestingly showed that 9% of mortalities from $PM_{2.5}$ emissions are due to biomass burning. Since it is a coarse model estimation, smaller scale of study in regional, national, or city scale is needed to reduce the uncertainties of the estimated model.

In the previous studies, Park et al. (2013) and Hoffer et al. (2020) estimated the potential of smoke, heavy metal, and PAH-bound PM emission factor from different type of OWB. In another study, Reyna-Bensusan et al. (2019) measured BC emissions from uncontrolled waste burning and estimated their effects on global warming potential (GWP). In addition, the emission pattern and contribution of OWB practices have been estimated at the national level (Cheng et al., 2020; Pansuk et al., 2018; Sharma et al., 2022) and city level in some countries (Das et al., 2018; Lal et al., 2016; Nagpure et al., 2015; Reyna-Bensusan et al., 2018). Meanwhile, according to Chaudhary et al., (2022), waste burning may be legalized through the use of portable clean air devices as substitutes for conventional OWB systems. Recently, Ramadan et al., (2022b) conducted transect walk surveys in Semarang during semi-lockdown in rainy seasons. The authors studied CO, CO_2, HC, NO_x, and total particulate matter (TPM) emissions from OWB. The combination of transect walk and questionnaire survey is better for making a robust inventory of emissions especially for OWB incidents (Ramadan et al., 2022a). In another study, Wiedinmyer et al., (2014) have summarized typical emissions from OWB; however, the authors estimated emission factors using old data, due perhaps to the lack of availability of data on current OWB practices. Previously, the contribution of OWB to GWP has only been evaluated once by Reyna-Bensusan et al., (2019), and the health risks associated with FA or bottom ash (BA) from OWB remain largely unknown. Most previous studies focused on the emission profiles of biomass burning, and limited scientific evidence is available regarding the impact of OWB exposure on the environment and human health

(Powrie et al., 2021). Thus, this chapter explains the environmental risk of OWB by multiplying the total waste burning in Semarang with the potential BC and other GHG emissions factors reported in literature. Finally, FA and BA residues from the open burning of household waste were characterized in terms of their chemical speciation and potential health risks. This study focused on the FA/BA residues or particulate matter (PM) and the exhaust gases. Our findings can fill the gaps in the high-level data inventories of OWB and support appropriate policy and decision making aimed at reducing emissions from the waste sector.

3.2. Materials and Methods

The study involved four sub-activities: laboratory tests, metals and hydrocarbons analysis, environmental risk assessment, and health risk estimation. Detailed information on each sub-activity is provided in the following sub-sections.

3.2.1. Laboratory Tests

Of the 16 routes determined, unburned waste was randomly collected (approximately 3-5 kg) from each route to assess its characteristics, composition, raw weight, and specific density. The unburned waste was divided into 11 categories: food waste, branches and twigs, paper and cardboard, plastic, metal, textile, rubber, glass, leaves, hazardous waste, and other waste. Thereafter, the 16 waste compositions were grouped and averaged as a defined cluster; these compositions were essential for determining waste composition for the mimicking of open burning practices and controlled combustion tests.

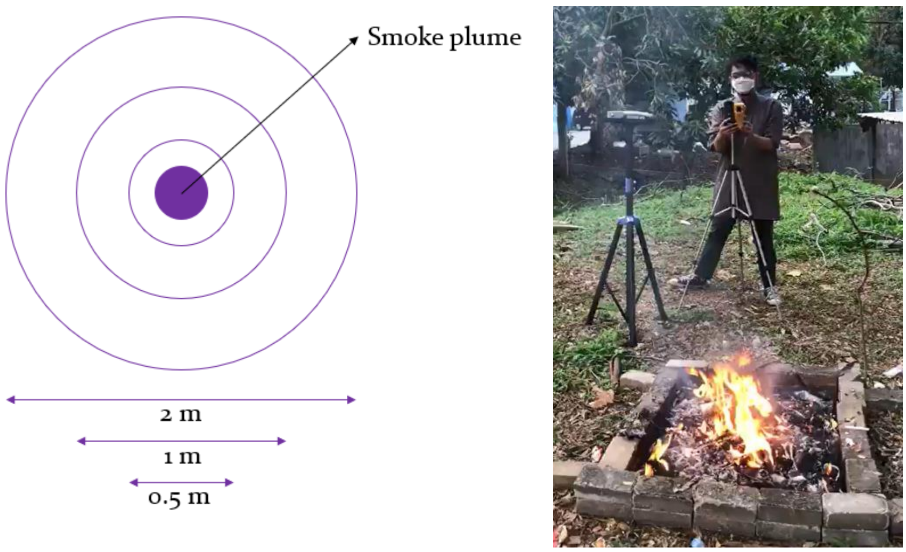


Figure 3.1 (a) Illustration of the distance for the field open burning measurement and **(b)** documentation for the measurement campaign

The first laboratory test was designed to mimic on-site open burning practices, involving the collection and burning of waste in a controlled manner. This methodology was inspired by the study of Alves et al. (2019) on on-site biomass burning and of Vreeland et al. (2016) for the exposure distance. Three piles were burned according to the general composition of burned waste, specific weight, volume, and moisture content of the Semarang City (Ramadan et al. (2022b)). The emissions produced during burning were then analyzed to identify the presence of various substances, including fine PM (PM_{10} and $PM_{2.5}$) (Alves et al. (2019)). The PM monitoring equipment Aeroqual 500® and handheld CO meter® were positioned approximately 0.5, 1, and 2 m from the pile and 1.2 m above the ground to obtain representative air quality from the smoke plume. Wind directions were determined during the burning event. Before the burning events, the air quality at the exact locations was measured to obtain the background concentrations. The burning time was defined as exposure time (ET) which is 30 minutes. The burning frequency per week was treated as exposure frequency (EF) as considered based on the short interviews with residents who practiced open burning. The burning and ambient temperature were measured directly using thermogun®. Figure 3.1(a) and (b) illustrate the details of laboratory test.

Figure 3.2 Laboratory test incinerator

The design of the second laboratory test (controlled combustion test) and the burning procedure followed that of Park et al. (2013), as shown in Figure 3.2. Approximately 2-4 kg of backyard waste was found to be burned. The initial suction blower discharge was approximately 8 m^3/min, and the average flow rate of the dust collection was 5.5 m^3/min. The waste was burned to completion. The average time taken to obtain wholly burned waste was approximately 25-30 min. The temperature of the burning chamber was approximately 400 – 500 °C. Fly ash was taken from the cyclone output, and bottom ash was taken from the bottom of the combustion chamber. The fly ash and bottom ash were weighed to determine the TPM emission factors and further chemical speciation analysis.

3.2.2. Statistical Analysis

The difference in the potential exposure level for each distance considered in the first laboratory test was measured using the Kruskal-Wallis's test. This test is a non-parametric statistic used to determine any significant differences between the medians of two or more groups (Mugica-Álvarez et al., 2018). This was used because the exposure data were not normally distributed. Instead, it uses ranks to determine whether groups have significant differences. The test was conducted by ranking all data from lowest to highest followed by comparison of the ranks of the data from different groups (Dubravská et al., 2020).

3.2.3. Metal and Hydrocarbons Speciation

As it is mentioned in the previous section, the second burning test followed the description by Ramadan et al., (2022b) and Park et al., (2013). The test was considered complete when fresh waste was completely burned. FA was collected using an isokinetic cyclone separator during the burning test. TPM and BA were further analyzed by identifying metal- and PAH-bound particulates. TPM and BA (0.1 g) were analyzed at the Advanced Chemistry Research Center, National Research and Innovation Agency, Indonesia. Metal-bound particulates were analyzed using inductively coupled plasma-optical emission spectrometry (ICP-OES). PAHs were analyzed by preparing 10 g of FA and BA samples and extracting them using 50 mL of dichloromethane while shaking for 6 h. The extract was concentrated to 2 mL using a rotary evaporator and then transferred to amber glass vials for gas chromatography-mass spectrometry (GC-MS) analysis.

Particulate matter collected in the second laboratory test was digested using nitric acid and hydrochloric acid mixture at a high temperature (180 °C) and power (1200 watts) for a specific time (25 min). The digested sample was diluted with 1 M nitric acid to a specific volume (50 mL) for analysis by Inductively coupled plasma optical emission spectrometry (ICP-OES) to determine the concentrations of certain metals, with the intensity of replication to each sample analysis was three times. The intensity and concentration of metals in the sample were determined by plotting the intensity of the signal against a calibration curve for the standard solution.

3.2.4. Health Risk Estimation

Potential health risks were evaluated by considering the cancer risk (CR) following human exposure to metals and PAHs, specifically among people who burn the waste. Since municipal waste burning is mostly conducted in the backyard, many people surrounding the house may have the same possibility of being exposed to FA. The average exposure doses of metal- and PAH-bound particulates from FA and BA were estimated using Eq. (1-3), presented by Keshavarzi et al. (2015), Liang et al. (2019), and Khan et al. (2020).

$$D_{ing} = \frac{C\ x\ IR_{ing}\ x\ FE\ x\ ED\ x\ CoF}{BW\ x\ AT} \quad (1)$$

$$D_{derm} = \frac{C\ x\ SA\ x\ AF\ x\ ABS\ x\ FE\ x\ ED\ x\ CoF}{BW\ x\ AT} \quad (2)$$

$$D_{inh} = \frac{C\ x\ IR_{inh}\ x\ FE\ x\ ED}{PEF\ x\ BW\ x\ AT} \quad (3)$$

D represents the exposure dose, which involves three main pathways, namely ingestion (D_{ing}), dermal contact (D_{derm}), and inhalation (D_{inh}). C is the total concentration of soil PAHs and metals (mg·kg^{-1}). IR_{ing} and IR_{inh} are the ingestion and inhalation rates, respectively (mg·day^{-1}). FE represents the frequency of exposure (days·year^{-1}). ED indicates the duration of exposure (year). BW is the average body weight (kg). AT is the lifespan (d). In equation for the dermal contact exposure dose, SA represents the surface area of the skin exposed to contaminants (cm^2), AF is the dermal adherence factor (mg·cm^{-2}), and ABS is the factor of absorption. PEF is particle emission factor (m^3·kg^{-1}) in the inhalation exposure dose calculation (D_{inh}).

CR of hydrocarbon-bound particulates was estimated using Eq. (4-7) (Liang et al., 2019), and chronic risk exposure caused by heavy metals from each pathway was determined using Eq. (8). The hazard index (HI) was determined to estimate the overall chronic risk (Eq. 9). The CR of only Cd, Pb, and Ni was considered since these metals are carcinogenic. Cd, Pb, and Ni contamination occurs through inhalation; therefore, CR caused by these metals was estimated by multiplying the inhalation exposure dose D_{inh} with CSF_{inh} and accounted for 6.3, 9.8, and 0.042 for Cd, Pb, and Ni, respectively. Pb can also be ingested, resulting in the values of 0.0085 mg·kgd^{-1} of CSF_{ing}. The human threshold set by the US EPA (2001) for CR is > 10^{-6}. The higher the CR value, the greater the carcinogenic risk to humans (Khan et al., 2020).

$$CR_{ing} = D_{ing}\ x\ CSF_{ing} \quad (4)$$

$$CR_{derm} = D_{derm}\ x \frac{CSF_{derm}}{GIABS} \quad (5)$$

$$CR_{inh} = D_{inh}\ x\ CSF_{inh} \quad (6)$$

$$CR_{total} = CR_{ing} + CR_{derm} + CR_{inh} \quad (7)$$

$$HQ_i = \frac{D_i}{RfD_i} \quad (8)$$

$$HI = \sum_{i=1}^{n} HQ_i \quad (9)$$

CSF represents the ingestion (CSF_{ing}), dermal (CSF_{derm}), and inhalation (CSF_{inh}) cancer slope factors (mg·kgd^{-1}), GIABS is the contaminant fraction absorbed in the gastrointestinal tract, and CR is the cancer risk of each exposure method (Keshavarzi et al., 2015). HQ represents hazard quotient, i represents the exposure pathways which are ingestion, dermal, or inhalation, and HI represents hazard index. RfD represents the specific reference dose for each pathway (mg·kg^{-1}·d^{-1}). Some RfD values were derived from Khan et al. (2020) and Liang et al. (2019), and the RfD of arsenic was derived from Nikolaidis et al., (2013). The reference data for each exposure factor are presented in Table 3.1.

Table 3.1 Reference data for exposure factors

Exposure variable	Child	Adult	Unit	Reference
Ingestion rate (IR_{ing})	200	100	mg.d^{-1}	US EPA, 2011
Inhalation rate (IR_{inh})	7.6	20	m^3.d^{-1}	US EPA, 2011
Frequency of Exposure (FE)	180	180	d.y^{-1}	Ferreira-Baptista and De Miguel, 2005
Exposure Duration (ED)	6	30	y	US EPA, 2011
Average body weight (BW)	16.2	61.8	kg	US EPA, 2011
Average life span (AT)	2,190	10,950	d	Keshavarzi et al., 2015
Skin exposed area (SA)	2800	5700	cm^2.d^{-1}	US EPA, 2011
Skin adherence factor (AF)	0.7	0.07	mg.cm^{-2}d^{-1}	US EPA, 2011
Skin absorption fraction (ABS)	0.001	0.1	unitless	US EPA, 2011; Man et al., 2010
Particle emission factor (PEF)	1.36×10^9	1.36×10^9	m^3.kg^{-1}	US EPA, 2011
Gastrointestinal absorption factor (GIABS)	1	1	unitless	US EPA, 2011
Ingestion cancer slope factor (CSF_{ing})	7.3 for hydrocarbon 0.0085 for Pb		mg.kg^{-1}d^{-1}	Knafla et al., 2006 Khan et al., 2020
Inhalation cancer slope factor (CSF_{inh})	3.85 for hydrocarbon 6.3, 9.8, and 0.042 for Cd, Pb, and Ni		mg.kg^{-1}d^{-1}	Wang, 2007 Khan et al., 2020
Skin cancer slope factor (CSF_{der})	25 for hydrocarbon		mg.kg^{-1}d^{-1}	US EPA, 1994

3.2.5. Environmental Risk Estimation

The oxygen concentration and flue gas, including HC, CO_2, CO, and NO_x, were measured using a QROTech (QRO-402) gas analyzer. The second burning test was repeated three times to improve the data accuracy. The flue gas concentration was counted 12 times in

24 min. The emission factor of the TPM was calculated using the following equation proposed by Park et al. (2013):

$$EF = \frac{S \ x \left(\frac{Q}{Q_p}\right)}{M} \tag{10}$$

where S is the mass of the fly ash collected in the cyclone, $\frac{Q}{Q_p}$ is the fraction of flow rate in the dust collection divided by the flue gas flow rate, and M is the total burned mass of the waste. The burning efficiency can be calculated by dividing the mass burned to completion by the raw/initial weight of the waste. Some emission parameters were estimated using the references' emission factors. The total emissions of municipal waste burning were calculated using the equation of Das et al. (2018):

$$Em = M_i \ x \ EF_i \tag{11}$$

where M_i is the total burned mass of waste, EF_i is the emission factor of the parameters, and Em is the total emission of the pollutant.

As it is informed above, the environmental risk caused by CO_2, CH_4, and N_2O emissions was calculated using the equation derived from literature e.g. the 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Volume 5 (Waste) (Beltran-Siñani and Gil, 2021). BC emissions were calculated separately because they are not included in the IPCC inventory. Therefore, BC emissions from OWB must be quantified separately, as this component is categorized as a short-lived climate pollutant (SCLP) and presents a higher GWP than CO_2 or CH_4 (Reyna-Bensusan et al., 2019). Eq. (12-15) were used to estimate CO_2, CH_4, N_2O, and BC emissions from open burning incidents in Semarang.

$$CO_2 \text{Emissions} = \sum_j \left(M_{wbj} \ x \ dm_j \ x \ CF_j \ x \ FCF_j \ x \ CE_j\right) \ x \frac{44}{12} \tag{12}$$

$$CH_4 \text{Emissions} = \sum_j \left(M_{wbj} \ x \ CH_4 \ EF_j\right) \ x \ CoF \tag{13}$$

$$N_2O \ \text{Emissions} = \sum_j \left(M_{wbj} \ x \ N_2O \ EF_j\right) \ x \ CoF \tag{14}$$

$$BC \ \text{Emissions} = \sum_j \left(M_{wbj} \ x \ BC \ EF_j\right) \ x \ CoF \tag{15}$$

$$\text{Total GWP} = \sum_j \ Em \ x \ GWP_k \tag{16}$$

M_{wb} represents the wet weight of waste burned in the city ($t \cdot y^{-1}$), dm is the dry matter fraction of the burned waste, CF is the fraction of carbon in the dry matter, FCF is the fraction of fossil carbon in the total carbon, CE is the combustion efficiency, CoF is a conversion factor of 10^{-6} $kg \cdot mg^{-1}$, and j represents the type of waste being burned. Some parameters, such as dm, CF, and FCF, were derived from default data in the IPCC inventories. From recent studies, the emission factors (EFs) for CH_4, N_2O, and BC were set at respectively 4, 0.24, and 4.7 g of pollutant per kilogram of burned wet waste. All EFs are based on Tier 1 or global emission default where assumed the waste contains 25-50% of DOC and 2% of N in dry matter and 60% of moisture content (Beltran-Siñani and Gil, 2021; Sharma et al., 2019b). All emissions were converted to ton year^{-1}. Then, GWP was calculated by summarizing the number of equivalencies for each pollutant (CH_4, N_2O, and BC) to CO_2 (Em). The values of 100-year GWP or $CO_{2\text{-eq}}$ (GWP_k) for CH_4, N_2O, and BC were 34, 298 (Hawthorne et al., 2017), and 1,100 (Bond et al., 2011), respectively. Total GWP of OWB was calculated using Eq. (16).

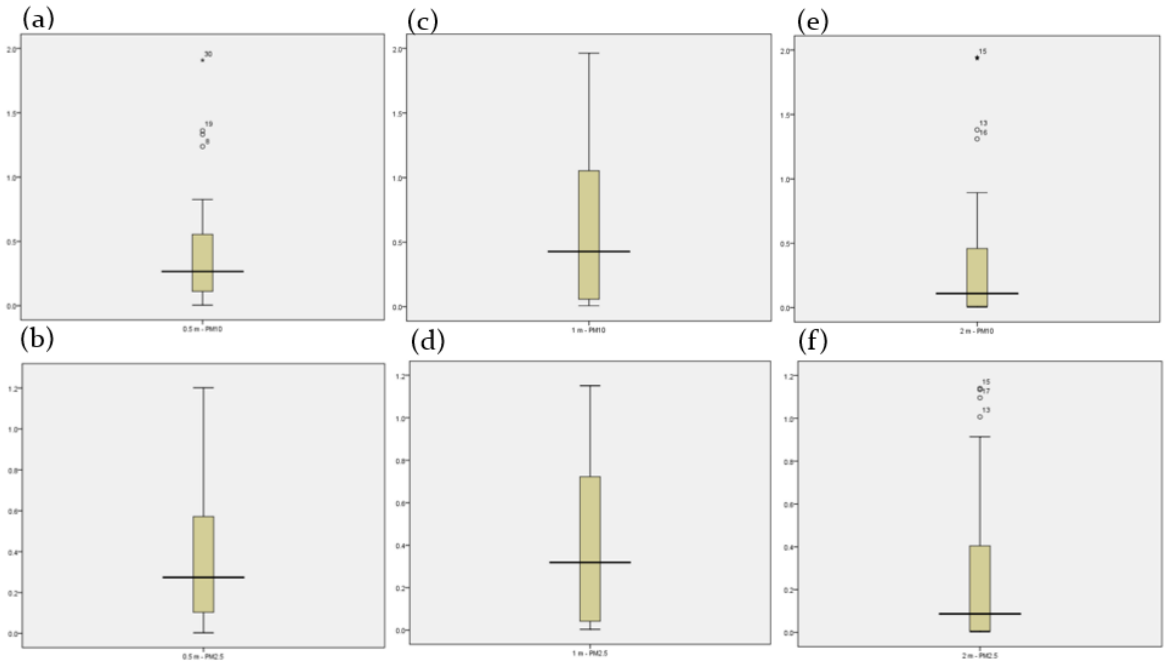


Figure 3.3 Boxplot of PM concentration at the distance of **(a)** 0.5 m – PM_{10}; **(b)** 0.5 m – $PM_{2.5}$; **(c)** 1 m – PM_{10}, **(d)** 1 m – $PM_{2.5}$; **(e)** 1 m – PM_{10}; and **(f)** 1 m – $PM_{2.5}$

3.3. Results and Discussion

3.3.1. Potential PM Exposure during Burning Activity

It is common for people in rural and peripheral areas of cities to burn their waste due to absence of waste collection services (Ramadan et al., 2022b). This can release significant amounts of PM, including BC, metals, and hydrocarbons into the air. The concentrations of PM_{10} and $PM_{2.5}$ have been reported to vary during the burning process, with the highest levels often occurring during the initial stages (Reyna-Bensusan et al., 2018). In this study, the concentrations of PM_{10} and $PM_{2.5}$ were 0.419 – 0.607 mg/m^3 and 0.289 – 0.399 mg/m^3, respectively. As shown in Figure 3.2, both PM_{10} and $PM_{2.5}$ concentration fluctuated during burning activities. The highest PM_{10} and $PM_{2.5}$ concentrations were approximately 1.964 and

1.201 mg/m^3, respectively. The average value was lower than the emissions emitted by wood burning and in semi-gasifiers used for heating purposes during the winter season (Snider et al., 2018). The average PM_{10} concentration was also lower than the level reported by Khan et al. (2020) in the industrial and residential sites of Islamabad, Pakistan (1.38–1.62 mg/m^3). The traffic load in these areas was predicted to contribute to the higher PM concentration. However, the composition of burned waste and the condition of the waste pile may affect PM emissions. Different waste compositions emit different levels of organic and elemental carbon. This study also supported Vreeland et al. (2016), who found that a distance of both 0.5 m and 2 m around the burn piles will pose the same level of exposure risk of contaminants and aerosols in OWB practice.

Table 3.2 shows that the distance from the smoke plume does not appear to significantly affect the concentration of fine particulate ($PM_{2.5}$ and PM_{10}) during open burning. This suggests that people close to the smoke plume are at high exposure risk to harmful particles regardless of their distance from the burn site (Vreeland et al., 2016). In the case of landfill fires, the presence of scavengers near the smoke plume can also increase their vulnerability to health risks through inhalation of large amounts of smoke and PM (Dada, 2021). Household or backyard burning is a common disposal practice in many rural areas that can pose a risk to nearby people. If the burning piles are located close to homes, the residents may be vulnerable to the harmful effects of the smoke and PM (Peter and Nagendra, 2021). It is essential for people to be aware of the potential health risks associated with open burning and to consider alternative methods of waste disposal. It is also important to note that different waste types can produce different emissions when burned. Conducting metal speciation analysis can help to understand these differences and to formulate the development of effective strategies for reducing the negative impacts of open burning.

Table 3.2 Statistical test for PM concentrations at different exposure distances (mg/m^3)

Results	**PM_{10}**			**$PM_{2.5}$**		
	0.5 m	**1 m**	**2 m**	**0.5 m**	**1 m**	**2 m**
Average	0.432 ± 0.084	0.607 ± 0.107	0.419 ± 0.112	0.372 ± 0.063	0.399 ± 0.067	0.289 ± 0.069
Min	0.004	0.008	0.005	0.003	0.004	0.000
Max	1.908	1.964	1.945	1.201	1.151	1.140
Median	0.266	0.427	0.110	0.274	0.319	0.870
Chi-square	4.476			4.089		
Asymp. Sig.	0.107			0.129		
Interpretation	No significant difference			No significant difference		

3.3.2. *Health Risks of OWB*

As shown in Table 3.3, nine hydrocarbon compounds and eight metal elements were detected in FA and BA collected from OWB. Among individual PAHs, some compounds, such as Nap, Bip, Ant, Flua, Nnt, and Tp, were detected in both FA and BA samples. Meanwhile, Ace and Fle were detected in FA alone, whereas Pyr was detected in BA alone. Among PAHs bound to particulates, the highest concentrator (FA and BA) was Tp with the average concentration of 0.896 mg kg^{-1} and 0.403 mg kg^{-1} for the FA and BA, respectively. The order of concentration from the highest to lowest in FA after Tp was Pyr (0.236 mg kg^{-1}) > Flua (0.180 mg kg^{-1}) > Ant (0.139 mg kg^{-1}) > Nnt (0.067 mg kg^{-1}) > Nap (0.052 mg kg^{-1}) > Bip (0.046 mg kg^{-1}). While for BA, the order after Tp was Nnt (0.316 mg kg^{-1}) > Bip (0.232 mg kg^{-1}) > Ant (0.241 mg kg^{-1}) > Fle (0.187 mg kg^{-1}) > Flua (0.130 mg kg^{-1}) > Ace (0.103 mg kg^{-1}) > Nap (0.067 mg kg^{-1}). Heavy metal concentrations in FA and BA were comparable. Specifically, concentrations of Zn of FA and Mn of BA were 2,072.35 and 1,699.26 mg kg^{-1}, which were significantly higher than those of the other metals. Since the present study is the first to evaluate the open burning of municipal waste, specifically in Indonesia, no historical or background concentrations are available for comparison. In the present study, as it can be seen in Table 3.3, the order of metal concentrations from the highest to lowest was Zn (2,072.35 mg kg^{-1}) > Mn (1,383.40 mg kg^{-1}) > Cu (124.81 mg kg^{-1}) > Cr (87.63 mg kg^{-1}) > Pb (43.58 mg kg^{-1}) > As (17.25 mg kg^{-1}) > Cd (16.18 mg kg^{-1}) > Ni (14.19 mg kg^{-1}) in FA and Mn (1,699.26 mg kg^{-1}) > Zn (975.31 mg kg^{-1}) > Cu (124.81 mg kg^{-1}) > Cr (41.44 mg kg^{-1}) > Pb (39.53 mg kg^{-1}) > As (22.10 mg kg^{-1}) > Cd (5.96 mg kg^{-1}) > Ni (5.32 mg kg^{-1}) in BA. Of the nine metal elements selected, only Hg was not detected during measurement.

Table 3.3 Hydrocarbons and metals detected in FA and BA.

Hydrocarbon compounds	**Abbreviation**	**Concentration (mg kg^{-1})**		**Metal elements**	**Concentration (mg kg^{-1})**	
	(Rings)	**Fly ash**	**Bottom ash**		**Fly ash**	**Bottom ash**
Naphthalene	Nap (2)	0.0521 ± 0.000684	0.0674 ± 0.0027	As	17.25 ± 2.95	22.10 ± 10.80
Biphenylene	Bip (2)	0.0461 ± 0.008149	0.2317 ± 0.1338	Cd	16.18 ± 1.87	5.96 ± 1.77
Acenaphthene	Ace (3)	nd	0.1032 ± 0.0767	Cr	87.63 ± 7.47	41.44 ± 2.97
Fluorene	Fle (5)	nd	0.1872 ± 0.1099	Cu	124.81 ± 5.36	138.58 ± 5.26
Anthracene	Ant (3)	0.1398 ± 0.0375	0.2408 ± 0.0724	Mn	1,383.40 ± 44.53	1,699.26 ± 45.98
Fluoranthene	Flua (4)	0.1803 ± 0.0144	0.1031 ± 0.0191	Ni	14.19 ± 5.87	5.32 ± 6.71
Pyrene	Pyr (4)	0.2356 ± 0.0492	nd	Pb	43.58 ± 38.09	39.53 ± 21.98
Naphthacene	Nnt (4)	0.06726 ± 0.03264	0.3159 ± 0.0661	Zn	2,072.35 ± 68.52	975.31 ± 29.38
Triphenylene	Tp (4)	0.8955 ± 0.3264	0.4028 ± 0.2382			

Furthermore, CR was measured to evaluate the potential carcinogenic effects of exposure to environmental pollutants. Three potential exposure pathways exist: ingestion, dermal contact, and inhalation. Details of calculation for each compound and element are provided in supplementary material, and the carcinogenic risk of exposure is presented in Table 3.4. Children are at a higher risk of exposure to metals and PAH-bound particulates, which produce adverse effects. Ingestion is the greatest risk pathway for both PAHs and metals emitted from OWB activities, followed by dermal contact and inhalation. The total carcinogenic risk from inhalation is identical for children and adults, although adults are at a greater risk of dermal contact. The maximum observed CR was approximately 4.77×10^{-6}, which is still within the tolerance threshold for humans.

Table 3.4 Cancer risk from exposure to OWB among local children and adults.

Exposure/Pathway		**Pollutant**		
		PAHs	**Metals**	**Total carcinogenic risk**
CR_{ing}	Child	2.98×10^{-7}	4.30×10^{-6}	4.60×10^{-6}
	Adult	3.90×10^{-8}	5.64×10^{-7}	6.03×10^{-7}
CR_{derm}	Child	9.99×10^{-9}	–	9.99×10^{-9}
	Adult	5.33×10^{-8}	–	5.33×10^{-8}
CR_{inh}	Child	4.38×10^{-12}	1.62×10^{-7}	1.62×10^{-7}
	Adult	4.38×10^{-12}	1.62×10^{-7}	1.62×10^{-7}
CR_{total}	Child	3.08×10^{-7}	4.46×10^{-6}	4.77×10^{-6}
	Adult	9.23×10^{-8}	7.26×10^{-7}	8.18×10^{-7}

Although the CR value was within the tolerance threshold for humans, HI indicated a greater potential for chronic health problems due to open burning activities. According to Keshavarzi et al. (2015), an HI of > 1 implies adverse health effects due to burning activities. Accordingly, FA may produce adverse health effects on children and adults. An aggregate HI was found in Table 3.5 to be more than 1 for FA in children (1.05) and adults (1.26) which indicates the possibility of non-carcinogenic risks in the burning activities. Therefore, adults may experience more significant health effects than children due to FA. Dermal contact was the most significant pathway of adverse health effects with the maximum value of HQ is 1.15, followed by ingestion (0.83), and inhalation (0.00081). Specifically, the HQ through inhalation is the lowest than ingestion and dermal contact. The HQ for child through ingestion, both in BA (0.75) and FA (0.85) were found to be higher than adult (0.11 and 0.10 for FA and BA). The different result found in dermal contact where the higher HQ value found in adult (1.15 and 0.61 for FA and BA).

Table 3.5 Chronic risk caused by exposure to metal-bound particulate among local children and adults.

Metal elements	**Hazard Index (HI)**			
	Child - Fly Ash	**Adult - Fly Ash**	**Child - Bottom Ash**	**Adult - Bottom Ash**
As	0.3514	0.0527	0.4502	0.0676
Cd	0.1953	0.5283	0.0719	0.1946
Cr	0.2655	0.4888	0.1256	0.2312
Cu	0.0196	0.0058	0.0218	0.0064

Metal elements	Hazard Index (HI)			
	Child - Fly Ash	Adult - Fly Ash	Child - Bottom Ash	Adult - Bottom Ash
Mn	0.0838	0.1337	0.1029	0.1643
Ni	0.0045	0.0014	0.0017	0.0005
Pb	0.0807	0.0361	0.0732	0.0328
Zn	0.0441	0.0165	0.0208	0.0078
Total	1.0449	1.2635	0.8680	0.7051

3.3.3. Environmental Impact of OWB

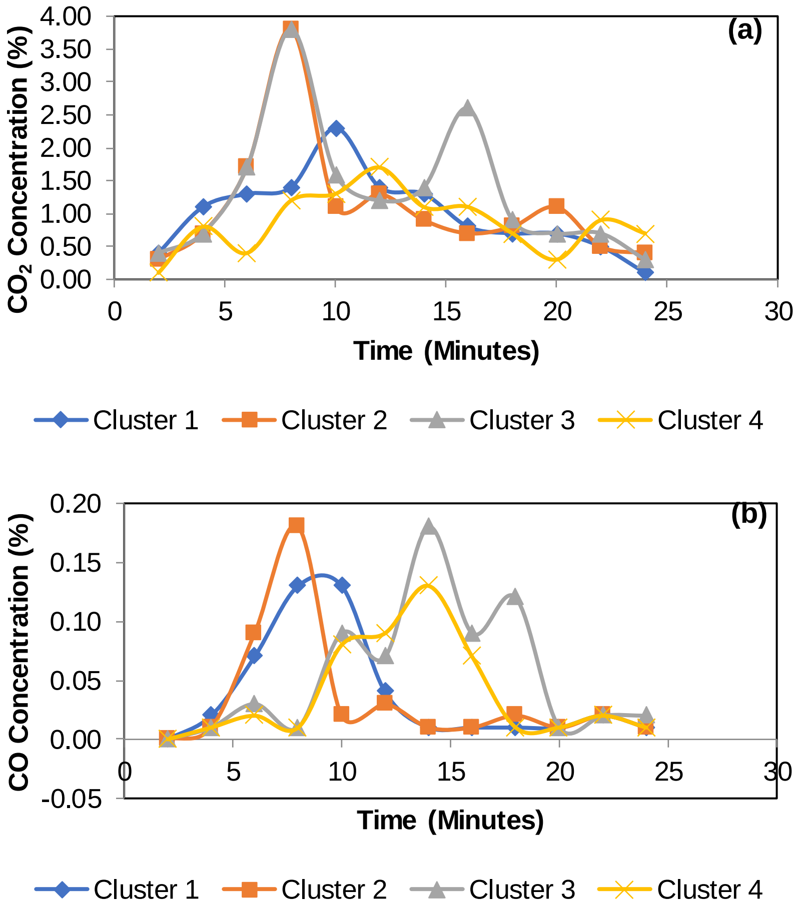
The emission of uncontrolled waste burning varies significantly according to the composition of the waste (Park et al., 2013). As shown in Table 3.6, different waste compositions produce a variety of emissions. For instance, when the concentration of plastic waste was high, the average concentration of CO was relatively higher than that in other burning incidents. In addition, a higher paper/cardboard composition in burning incidents results in higher NOx. During the 24 min of open waste burning, a significant amount of CO and CO_2 is produced at the beginning of the burning activity. The CO and CO_2 emissions reach their peak after 8 min of burning, and NO and hydrocarbons increase after 10 min of uncontrolled burning. Therefore, the burning efficiency was found to differ among the four samples.

Table 3.6 Concentration of CO_2, CO, HC, and NO_x emission during uncontrolled burning

Cluster	Waste burned composition ratio*	Parameter (g/kg)											
		CO			CO_2			HC			NOx		
		Min	Ave	Max	Min	Ave	Max	Min	Ave	Max	Min	Ave	Max
Cluster 1 (Rural)	85 : 4 : 10 : 1	0.10	0.38	1.30	1.00	10.00	23.00	0.09	0.14	0.22	52.00	96.33	147.00
Cluster 2 (Outer periurban)	79 : 8 : 3 : 10	0.10	0.34	1.80	4.00	11.08	38.00	0.08	0.11	0.21	45.10	76.23	123.40
Cluster 3 (Inner periurban)	51 : 42 : 2 : 5	0.10	0.54	1.80	3.00	13.33	38.00	0.09	0.12	0.21	69.80	84.88	116.70
Cluster 4 (Urban Core)	80 : 16 : 2 : 2	0.10	0.38	1.30	3.00	8.58	17.00	0.09	0.14	0.22	67.50	81.13	118.10

*backyard waste : plastic waste : paper : other waste

The highest burning efficiency (91.81%) was found in Cluster 1, where the highest backyard waste was found; this was followed by cluster 4, which had a lower proportion of non-combustible waste. Clusters 2 and 3 were found to have the lowest burning efficiency, with only 57–59% of waste being burned owing to the presence of many incombustible wastes in the waste composition. The concentration of all pollutants decreased significantly when the fuel was exhausted. Accordingly, the findings of this test indicate that pollutants are emitted significantly during the burning of waste, ultimately harming the environment (see Figure 3.4). It was also estimated that 0.48 g/kg or 28.37 ton/year of TPM is emitted from waste burning in Semarang City. This TPM concentration is three-fold lower than the previous research [18]. Therefore, the results obtained may be higher depending on the characteristics of the waste burned and the burning conditions (Akagi et al., 2011). The emissions from the burned waste in Semarang City were lower than the global estimation. For instance, open burning emitted 2.470 Gg/year of CO or 30-fold lower emission than that estimated in Ibadan City, Nigeria (Okedere et al., 2019). In addition, another researcher estimated that the PM2.5 emission in Semarang City was 1.5-fold higher than that in the Delhi municipalities (Guttikunda and Calori, 2013). Therefore, it is estimated that the emissions from Semarang City are higher than those reported in other studies that used the same transect walk methods (Das et al., 2018). These differences among studies may be due to the dynamic situation of each city.



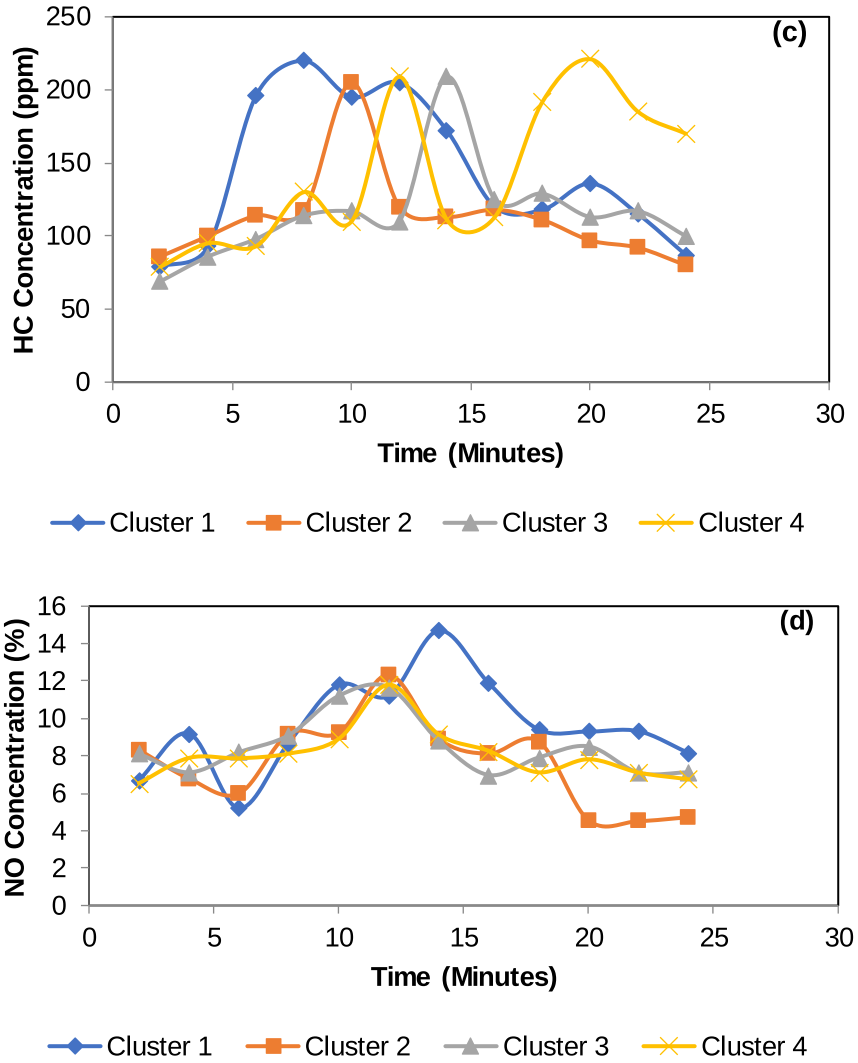


Figure 3.4 Concentration of (a) CO_2, (b) CO, (c) HC, and (d) NO during 24 minutes of burning under different waste composition per cluster sample

Coarse estimation results demonstrated that CO_2 (25,260.32 $t.y^{-1}$) is the largest pollutant emitted from OWB, followed by BC (365.30 $t.y^{-1}$), CH_4 (310.89 $t.y^{-1}$), and N_2O (18.65 $t.y^{-1}$). Because the GWP of BC over a 100-year horizon is higher than that of other pollutants, OWB practices emit a higher CO_2 equivalency than methane (Table 3.7). Specifically, at least 53,809.66 tons of $CO_{2\text{-eq}}$ are emitted annually in Semarang from OWB practices. Based on data from 2018, Syafrudin et al. (2021) estimated that the overall emissions from the waste sector were approximately 1,650 kt; however, the authors ignored the potential of waste burning events and attributed the highest emissions to uncontrolled landfills. Moreover, previous studies used different approaches (tiers 1 and 2) to create data inventories.

Table 3.7 Environmental impact of OWB in Semarang.

Parameters	Values (t y^{-1})	$CO_{2\text{-eq}}$ emissions (GWP 100-year, t y^{-1})
CO_2	25,260.32	25,260.32
CH_4	310.89	10,570.38
N_2O	18.65	5,558.77
BC	365.30	12,420.19
Total		53,809.66

3.3.4. Discussion

Based on the chemical speciation of particulate emissions, adults and children are at a potential chronic risk due to open burning incidents. Some metals can enter the body via dermal contact, ingestion, and inhalation pathways. Even though internationally accepted precautionary criteria have been set against metal- and hydrocarbon-bound particulates, residents may still experience pulmonary and respiratory illnesses in the case of lack of interventions against OWB. Since the value of HIs were all higher than the permissible limit, the more contact with both PAHs and trace elements can cause several disorders (Keshavarzi et al., 2015). In addition, people who are directly exposed to open burning may experience certain health problems, such as abdominal pain, headache, hypertension, glioma, and mental effects because of metals-bound particulate (Khan et al., 2020). However, those symptoms can be derived from other causes which need further in-depth study.

Some gaseous pollutants and PM are emitted during burning. This issue is well known, because open burning also emits BC, which shows a higher GWP than methane and carbon dioxide. However, as BC is not included in calculations according to the IPCC methodological, its emissions are often underestimated and beyond prediction (Reyna-Bensusan et al., 2019). BC has been categorized as an SLCP, different from other long-lived GHGs (Bond et al., 2011). Reyna-Bensusan et al., (2018) estimated annual BC emissions of approximately 24,840 tons over a 20-year horizon in Huejutla, Mexico, which is higher than that estimated in Semarang. In the present study, BC emissions from OWB contributed to over 5% of the relative total emissions in the city. Open burning can act as a source of many local respiratory illnesses and problems through inhalation of the generated smoke. However, this activity is underestimated because of the lack of data (Reyna-Bensusan et al., 2018).

Regarding problems and solutions, some lessons learned from previous studies may help decision makers reduce the environmental and health effects of OWB. First, a decentralized waste management system may be an appropriate short-term solution for an isolated and unserved waste collection system. As reported by Chaudhary et al., (2022), improved burning devices can reduce the emissions and health effects of waste burning, including landfill fires. Further, community-based solid waste management, as a decentralized system, can be used to reduce OWB activities (Budihardjo et al., 2022). Second, promoting circular economic opportunities among local leaders, such as upcycling and selling of valuable waste, can improve the economic benefits to citizens even in rural areas (Mihai et al., 2021). Third, increasing environmental knowledge through specific planned activities may encourage people to better manage their waste and stop burning waste. Inadequate waste management systems, which are supported by the lack of environmental consciousness, may increase the possibility of exposure to PAH- and metal-bound particulates emitted from open burning activities. Finally, a consolidated approach from waste management stakeholders is required to obtain an appropriate solution to reduce burning incidents (Permadi and Kim Oanh, 2013a). Since the present study used some emission factors derived from literature, future studies should analyze precise emission factors for OWB to obtain a higher-tier inventory of health hazards and emissions. Moreover, different demographic characteristics should also be considered when evaluating the cause behind OWB practices at the city level.

3.4. Summary

In this study, the PM concentration at a distance range of 0.5 – 2 m between the receptor and the smoke plume is similar or not statistically different. Therefore, people around pile burning in this range will experience the same health impact. To the best of our knowledge, the present chapter is the first to comprehensively reveal the associated number of emissions and the potential health risks of OWB incidents. Rural and outer peri-urban areas are the highest contributors to OWB and should be noted as focus areas for reducing the climate impacts of OWB. Furthermore, BC emissions from open burning significantly contribute to GWP. Therefore, preventing OWB may contribute to the achievement of SDGs. From our findings, OWB is associated with a small CR, particularly due to emitted particulate matter. However, exposure to OWB may be associated with a high risk of certain chronic diseases. Thus, preventive measures are warranted against OWB at the household level.

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Chapter 4

Optimizing Waste Collection Points by using Spatial Analysis to Reduce OWB in Semarang City

4.1. Introduction

Waste management in developing countries has been a topic of concern for decades, as many of these nations struggle with inadequate infrastructure, limited resources, and weak governance (Ramadan et al., 2022b). Several studies have focused on the low waste collection rates in these countries and their associated environmental and health hazards (Nagpure, 2019; Ramadan et al., 2022a). Additionally, waste collection rates are often low in developing countries because of inadequate funding, weak regulatory frameworks, and limited public awareness of the importance of proper waste disposal (Marshall and Farahbakhsh, 2013). Waste management systems in developing countries are often informal and rely on scavenging and recycling in the informal sector. Investment in formal waste management systems, including waste collection and transportation, is essential for mitigating the negative impacts of waste on the environment and public health (Fatimah et al., 2020). Therefore, many developing countries have lower waste collection rates due to inadequate funding, weak regulatory frameworks, and limited public awareness of the importance of proper waste disposal (Kumari, 2019).

Several studies have examined the potential consequences on communities that are far from waste collection sites and their impacts on the environment and public health. In India, many communities conduct illegal dumping because they are far from waste collection sites (Nagpure, 2019). This study also highlights that burning, burying, and disposal of waste directly into the environment can lead to air and water pollution, which can have adverse effects on human health and the environment. Similar to Indonesia, communities far from waste collection sites are more likely to burn their waste, resulting in the emission of harmful pollutants (Ramadan et al., 2023). In Nigeria, inadequate waste management infrastructure in rural communities' results in the dumping of waste into open spaces, which can attract rodents and insects that transmit diseases to humans. The study also noted that burying waste can lead to groundwater contamination, which can affect the quality of drinking water (Ndukwe et al., 2019). These studies indicate that communities that are far from the waste collection site/point will potentially burn, bury, and dispose of their waste directly into the environment (Das et al., 2018). Improving waste management infrastructure and services in cities is critical to prevent improper disposal of waste and reduce the associated negative impacts.

As improving waste management infrastructure is necessary, there is a need for more research on appropriate methods for conducting optimization analysis. While some studies have used simple proximity-based methods, such as the average nearest neighbor (ANN) approach, others have used more complex optimization techniques, such as integer programming and network flow models (Miftahadi et al., 2022; Stopka et al., 2019). To continue exploring the strengths and weaknesses of these different methods and identify the most appropriate method for different contexts is essential. The spatial multi-criteria decision analysis (SMCDA) approach is a powerful tool for identifying priority areas for waste management. However, many studies have focused on landfill sites or central municipal solid waste (MSW) plants (Hazarika and Saikia, 2020; Kamdar, 2019; Lim and Afifah Basri, 2022; Mussa and Suryabhagavan, 2021). In this case, SMCDA can define the unserved area, provide a better visualization of waste management facility distribution, and analyze the priority area of its development (Yalcinkaya and Uzer, 2022). Location-allocation analysis can also be an option for bin allocation in the cities. Therefore, only a few studies using this tool to provide an appropriate placement of public facility such as waste collection site (WCS) and recycling facilities (Morsink-Georgali et al., 2021; Rathore et al., 2020).

Several researchers have focused on WCS optimization by conducting different methodologies. Boskovic and Jovicic (2015) and Rathore et al. (2020) designed a fast methodology using a location-allocation analysis to calculate the number of optimal waste bins and collection points. Danbuzu et al. (2014) employed spatial distribution analysis using the nearest neighbor to understand the difference between the distribution of illegal and formal collection points. Another distribution analysis was conducted by Afzal et al. (2021), who used kernel density (KD) to determine the risk of open dumping sites in Karachi, Pakistan. The SMCDA for mapping out the potential location of WCS can be found in the study by Amri et al. (2021), who used several criteria to add more WCS in a sub-district. Yalcinkaya and Uzer (2022) used a combination of location-allocation analysis, vehicle routing problem (VRP), and analytic hierarchy process (AHP) as MCDA tools for determining the optimum WCS. Therefore, as it is limited to find the literature on the optimization of WCS, a proper analysis on this field is necessary to ensure optimal reallocation of waste collection sites (WCS).

This study used a geographic information system (GIS) to evaluate the suitability of existing waste collection sites (WCSs) using several spatial modellings. The optimization of waste collection sites (WCS) may reduce uncollected waste, thus reducing the open waste burning intensity in Semarang City. This study aimed to (1) validate the existing data of WCS and its capacity from field surveys and government databases, (2) describe the WCS distribution based on spatial patterns, (3) propose a new suitable WCS, especially in the unserved area, (4) optimizing existing WCS location and capacity, (5) analyze potential reduction of unmanaged waste in Semarang City.

4.2. Material and Methods

4.2.1. Data Collection

The WCS database was obtained from the Environmental Agency of Semarang City and contains the following data:

- Type and identity of vehicles that put the waste from each WCS
- Address of the WCS,
- The number of containers in the WCS.

The database does not contain exact coordinate/full address of the WCS or detailed information on the existence of the WCS. Field surveys and short interviews were conducted to recheck and coordinate existing WCS. The percentage of collected waste ($\%W_c$) in each WCS was estimated by dividing the WCS capacity (WCS_c) by the total waste generation (total population per sub-district (P_x) × Semarang City waste generation per capita (W_{pc}), as shown in Eq. (1).

$$\%W_c = \frac{WCS_c}{P_x \; x \; W_{pc}} x100\% \quad (1)$$

Data that were collected by interview and field survey, include:

- Name of WCS
- Geocoordinate (latitude and longitude) using Global Positioning System (GPS)
- Detailed address
- Number of container (C)
- Estimated area of WCS
- Type of vehicle and its capacity (T), where it is assumed that the arm roll truck has a capacity of 6 m3 and dump truck has a capacity of 8 m3
- Source of waste
- The estimated volume capacity of the WCS (WCS_c), calculated using Eq. (2).

$$\mathrm{WCS_c} = \mathrm{C \; x \; T \; x \; t} \quad (2)$$

where t stands for trips (assuming that all vehicles have three trips/day). $\mathrm{WCS_c}$ was validated by comparing the estimated volume of WCS with the total waste transported into the landfill using Eq. (3).

$$\sum WCS_{cw} = \sum WCS_c \; x \; \rho \quad (3)$$

where ρ is the average waste density in the trucks (185 kg/m^3). Geocoordinate data were captured using My GPS Coordinates®, an Android phone application. The surveyor was trained before conducting the field survey to ensure that all WCS datasets were valid. The land use and slope geodatabase for the SMCDA was derived from the National Digital Elevation Model (DEMNAS – https://tanahair.indonesia.go.id/). The data on the total population of each sub-district in Semarang City were generated from the Statistical Agency of Semarang City in 2021. The population density was calculated using Eq. (4). The area of each sub-district (A_x), city roads, land use, disasters, and housing maps were obtained from the municipal Government of Semarang City in 2021.

$$P_D = \frac{P_x}{A_x} \tag{4}$$

4.2.2. Research Methods

The conceptual framework and methods used in this study are illustrated in Figure 4.1. The research was divided into two main procedures: descriptively analyzing the current WCS distribution in Semarang City and spatial multi-criteria decision analysis (SMCDA) to determine the potential location of new WCS that can improve waste collection, thus reducing the generation of uncollected waste in Semarang City. ArcGIS 10.3 software was used for geocoding, analysis of the ANN, KD, buffer, SMCDA, location-allocation, and preparation of all output maps.

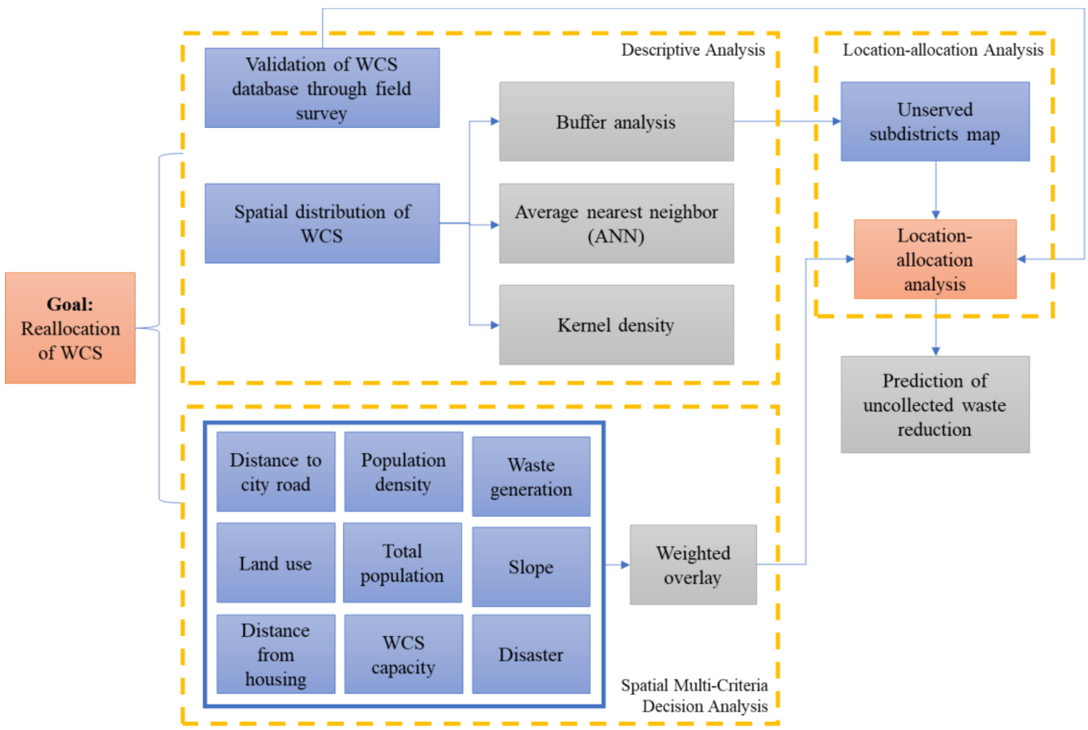

Figure 4.1 Conceptual framework for WCS reallocation

4.2.3. Identifying Spatial Pattern

The WCS point distributions were analyzed using the average nearest neighbor (ANN) and incremental spatial autocorrelation (ISA) as preliminary statistics. These tools are used to observe the distribution of WCS through simple geostatistical modelling (Le et al., 2022). Buffer tools were used to generate buffers with service radii of 500 m to 1,000 m. Determination of these distances was based on the Public Works Ministry of Indonesia Regulation Number 3/PRT/M/2013. The spatial join tool was used to produce sub-district areas without services based on the buffer zone. Subsequently, KD was used to generate a heatmap and the thickest density location of the waste collected by the WCS. The overcapacity of the WCS map was generated by comparing the total WCS capacity (WCS_c) with the prediction of waste generation in a sub-district (W_x). The W_x was calculated using Eq. (5) When the WCS_c is higher than W_x the sub-district or WCS in the sub-district area is assumed to have overcapacity. Therefore, this map will be used as a baseline to reallocate the WCS.

$$W_x = P \ x \ W_{pc} \qquad (5)$$

4.2.4. Spatial Multi-Criteria Decision Analysis

SMCDA is a decision-making method that combines geographic information systems (GIS) with multiple-criteria decision analysis (MCDA) techniques. SMCDA provides decision-makers with a comprehensive framework to evaluate alternative options based on a range of criteria that are spatially referenced (Amri et al., 2021; Yalcinkaya and Uzer, 2022). In SMCDA, spatial data are used to create maps that represent different criteria relevant to the decision-making process (Bosompem et al., 2016). In the context of allocating WCS in Semarang City, criteria such as distance to city roads, land use, disaster, distance from housing, slope, population density, total population, WCS capacity, and waste generation were used. The details of each criterion, its classification, and references as background theories are shown in Table 4.1. Each criterion is assigned a weight that reflects its relative importance in the decision-making process. Once the criteria have been defined and weighted, the SMCDA evaluates the alternatives and assigns a score to each option based on how well it meets the criteria. These scores are then combined to produce a final ranking of the alternatives (Dolui and Sarkar, 2021).

Table 4.1 Criteria for allocating WCS

References	Suitability criteria	Classification	Reason/Hypothesis	Influence in SMCDA (%)
Syafrudin et al. (2023)	Population density	0 – 5,000; 5,001 – 10,000; 10,001 – 15,000; 15,001 – 20,000; > 20,000	The bigger the density, the bigger the anthropogenic activity will be	15
Antczak, (2020)	Total population per sub-districts	0 – 5,000; 5,001 – 10,000; 10,001 – 15,000; 15,001 – 20,000; > 20,000	The bigger the population, the higher possibility of waste generation	5
Amri et al. (2021)	Distance to city road	1 (> 200 m); 2 (150 – 200 m); 3 (100 – 150 m); 4 (50 – 100 m); 5 (0 – 50 m)	WCS should be placed near to the city road so the vehicles can access easily	10
Dolui and Sarkar, (2021)	Distance from housing	1 (> 200 m); 2 (150 – 200 m); 3 (100 – 150 m); 4 (50 – 100 m); 5 (0 – 50 m)	WCS should be near the residential areas to prevent waste burning or other improper disposal activity	10
Agovino and Musella, (2020); Nagpure, (2019)	Total container capacity per sub-districts	1 (72-90 m^3); 2 (54-72 m^3); 3 (36-54 m^3); 4 (18-36 m^3); 5 (<18 m^3)	The bigger value of container capacity per sub-districts, the less the need of container to be added in WCS	5
Cheniti et al. (2021)	Waste generation	0 – 20; 21 – 40; 41 – 60; 61 – 80; 81 – 156 (m^3/day)	The bigger the waste generated by the residents, the bigger need of WCS	15
Bosompem et al. (2016)	Land use	1 (Worship, fisheries, offices, housing, defense and security, mining, forest, river-border, sea-border, and small-medium enterprises industrial area); 2 (industrial area, health facility, and crops area); 3 (sports facility, education facility, cultural heritage area, market facilities); 4 (Forest facility, tourism area); 5 (Plantation area, green and non-green open space, transportation facility)	The value for classification is based on Semarang City Government Regulation	10
Syafrudin et al. (2023)	Slope	0 – 8%; 8 – 15%; 15 – 25%; 25 – 45%; > 45%	The bigger the slope, the less possibility to reallocation the facility	15
Amri et al. (2021); Saputra et al. (2021)	Disaster-prone areas	1 (vulnerable to disaster event) and 5 (no disaster event)	The area which is vulnerable to disaster may not be used as WCS	15

Several factors, including distance to city roads, land use, disaster-prone areas, distance from housing, slope, population density, total population, WCS per capacity, and waste generation, were weighted, as they influence SMCDA. The influence percentage was calculated by expert judgement and a literature survey. This SMCDA method is considered a fast method for determining the potential locations that can be generated through several criteria, as mentioned in Table 4.1.

4.2.5. Location-Allocation Analysis

There would be 2 analyses of location-allocation analysis. The first analysis will focus on finding a suitable location for additional WCS. This additional location is extracted from previous SMCDA results. Each candidate location is converted into point features. There would be a residential map outside the range of existing WCS by intersecting the residential areas and a buffer map of existing WCS. Location–allocation analysis, which is part of the network analysis in ArcGIS, was set up to solve the minimized candidate facilities, while the impedance cutoff was set up to 1,000 meters. This distance was the maximum buffer coverage of each new WCS location. The second analysis is focused on the existing WCS overlayed with the centroid of residential areas. In this analysis, the maximum coverage location problem (MCLP) was determined as the problem that wants to be solved. The impedance cutoff is different from the previous analysis, while the existing WCS was set to be 2,000 meters. Location allocation analysis determined which WCS should be closed to reduce waste collection emissions and idling time for the waste vehicle. All maps are generated for further analysis. Total predicted unmanaged waste based on the optimization of WCS and before optimization was then analyzed to understand the proposed optimization study better.

4.3. Results and Discussion

4.3.1. Overview of Study Area and Field Survey Result

Indonesia is a country with a lower middle-income status in East Asia, according to the World Bank. The country's Gross Domestic Product (GDP) and GDP per capita in 2021 (constant 2015 US$) were USD 1.07 trillion and USD 3,892.5, respectively. Semarang City, the sixth largest city in Indonesia, has a GDP of USD 24,800 million and GDP per capita of USD 3,790. As of 2021, the minimum wage in Semarang City is USD 161.39 per month. The city has a population of 1,595,267, with a population density of 4,552 people per km^2, and approximately 4.14% or 73,600 people are considered poor. Semarang City consists of 16 districts and 177 sub-districts, with 1,499 neighborhood associations (NAs) and 10,423 neighborhood units (NUs). Informal recyclers, including community-driven material recovery facilities (CdMRFs) or locally known waste banks, scavengers, scrap collectors, and traders, are responsible for most inorganic waste recycling activities, and they typically work individually or for small or micro enterprises. CdMRFs usually collect recycled waste from nearby households, offices, or restaurants, which are then resold to scrap collectors or processed by the CdMRF (Budihardjo et al., 2022). According to https://sipsn.menlhk.go.id/, the city generates 5,905.5 m^3/day of waste in 2022 (assumed waste density is equivalent to 200 kg/m^3), which is equivalent to 1,181.1 tons per day or 431,085.2 tons per year, with a waste composition dominated by food waste (61%), followed by plastic (17%), paper and cardboard (10%), LWTR (leather, wood, textile, and rubber; 7.73%), metals (1.22%), and others (2.88%). The largest source of waste came from households, accounting for around 72%, followed by public facilities (9%), markets (8%), and others (11%).

The waste collection system in Semarang City, Indonesia is managed by the municipal government and involves several steps. Households generate most of the waste in Semarang. The municipal government does not provide door-to-door waste-collection services. Each household or neighborhood association (NA) manages their waste by renting a private company to take the waste from each household or independently assigning someone or people in their community to dispose of waste regularly using specific vehicles that visit neighborhoods on scheduled days. Several vehicles, such as three-wheeled motorcycles, carts, pickups, and trucks, are used to remove waste from households (Ramadan et al., 2023). The commercial sector generates a significant amount of waste in cities. The municipal government provides waste collection services to commercial establishments, such as markets, shops, and restaurants, using separate collection trucks (arm-roll and dump trucks). Once the waste is collected from households and commercial establishments, it is transferred to the WCS located throughout the city or directly sent to the Jatibarang landfill, particularly when the location source is near the landfill site (Ramadan et al., 2022b).

The amount of waste sent to landfills, was approximately 1,050.0 ton/day in 2021. Therefore, based on the assumption that the waste density in trucks was 200 kg/m^3, the total amount of waste sent to the landfill, which was calculated from the total WCS capacity becomes 1,192.8 ton/day. The total waste capacity value was then used for spatial analysis. At the WCS, the waste is collected in larger trucks for transport to the final disposal site. The total WCS of Semarang City WCS is 198. The number of WCS reported by the municipal government is 208. Some were missing or not found during the field survey. The maximum area of WCS can be found in the Lingkar Tanjung Mas WCS which has 84 m^2 of area, located in the Panggung Lor sub-district; and the minimum area of WCS can be found in the Mbok Berek WCS which has 6 m^2 of Area, located in the Manyaran sub-district. Similar to the reported data from the SIPSN website, the biggest waste source comes from households, which accounted for 73% of the waste sent to landfills, followed by the market, industry, restaurants, hotels, hospitals, and others. Arm-roll trucks, which have the capacity to pick up approximately 6 m^3 of waste in each trip to landfill, dominate the vehicles in Semarang City. The dump truck (8 m^3 capacity) only accounted for 5% of the total waste pickup vehicles operated in Semarang City (See Figure 2).

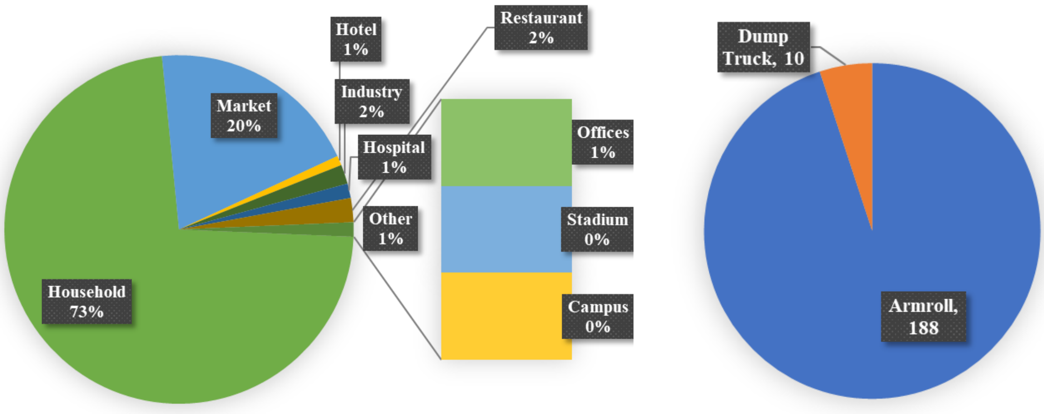

Figure 4.2 Waste sources (left) and types of vehicles (right)

4.3.2. Distribution Analysis of WCS

The distribution of WCS as a formal waste collection point in Semarang City using the nearest neighbor (ANN) method shows that the nearest neighbor ratio is 0.778 with a Z-score of -5.967, which indicates that the WCS is significantly clustered. There is also a report showing that a clustered pattern could be the result of random chance. The results of this study showed a different pattern from the previous study in Nigeria, which concluded that the waste collection points showed a different pattern from the urban population (Danbuzu et al., 2014). In Semarang City, the pattern of WCS seems to be clustered, as was found in the urban population. A KD map is shown in Figure 4.3, showing that the hotspot area or the largest waste production is in the three zones indicated by red circles. Semarang Tengah and Semarang Selatan Districts have the darkest colors, indicating high anthropogenic activity in those districts. This result agrees with the findings of Ramadan et al. (2022b), who also considered the two districts as the urban core. Therefore, these districts have the highest population and housing density compared to other districts, which is relevant to the number of WCS found in the area. The second and third red circles are found in Tembalang District and Banyumanik District, which are the growing settlement areas against the flood-prone areas in the northern part of Semarang City. A previous study confirmed that the settlement growth direction of Semarang City is gradually increasing toward the southern part of Semarang City, such as Banyumanik, Tembalang, and Gunungpati District. Road network development is also growing into a triangular form, which consists of Genuk District, Tugu and Mijen District, and Banyumanik – Tembalang District.

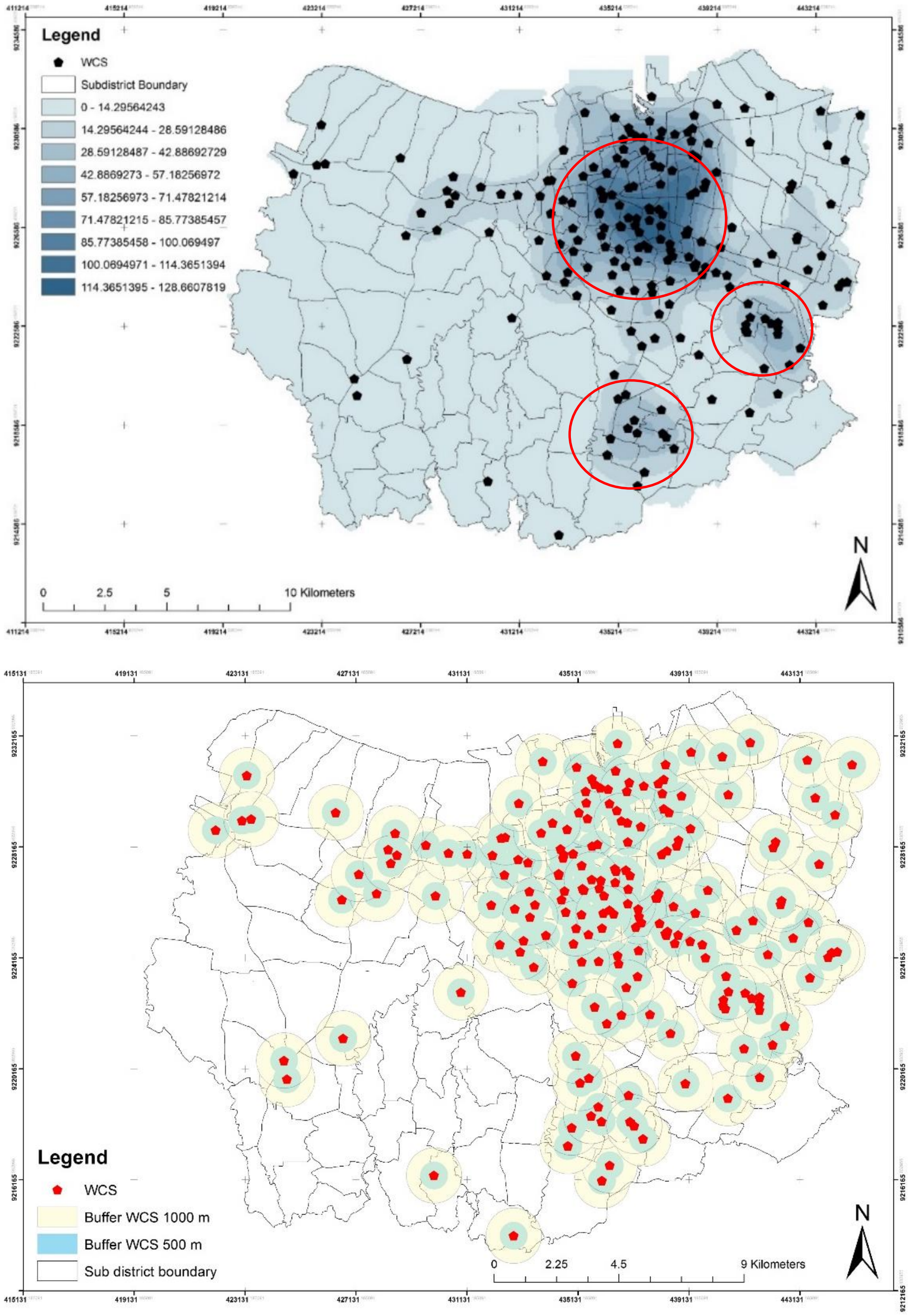

Figure 4.3 Kernel density (KD) (above) and buffer (below) map

WCS was evaluated based on the total capacity to receive waste. Using the Eq. (5), among 198 WCS, 112 WCS are found to have less capacity than the waste generation produced in the sub-districts area. Besides, there are 68 sub-districts out of 177 which has no WCS in their area. An optimization of WCS location is needed in each subdistrict, especially in which is at overcapacity. The WCS capacity data are used as a baseline to determine the potential location of WCS reallocation in the SMCDA.

4.3.3. Spatial-MCDA Results

This study is a significant development in filling a crucial void in identifying the potential location of WCS and improving the cost-effectiveness and efficiency of waste management initiatives. GIS is a fitting tool for site selection studies, as it can handle extensive amounts of spatial information from various sources (Hazarika and Saikia, 2020; Mussa and Suryabhagavan, 2021). Therefore, by combining different suitability criteria maps using weights, a suitability index map was created, which helped identify the optimal locations for the new WCS. The resulting suitability index map for the study area is shown in Figure 4.4(a). Within the total study area, approximately 0.01% (2.48 hectares) met the requirements for high suitability based on satisfying criteria in social, environmental, and geospatial areas. Most of these areas are located in the southwestern part of Semarang. Areas deemed suitable cover 4.35% (1,682.35 ha), moderately suitable areas cover 52.45% (20,260.09 ha), less suitable areas cover 41.82% (16,154.65 hectares), and the remaining 1.37% (531.38 hectares) are not appropriate for reallocation of WCS. The existing WCS is denoted by black points on the map (Figure 4.4(a)).

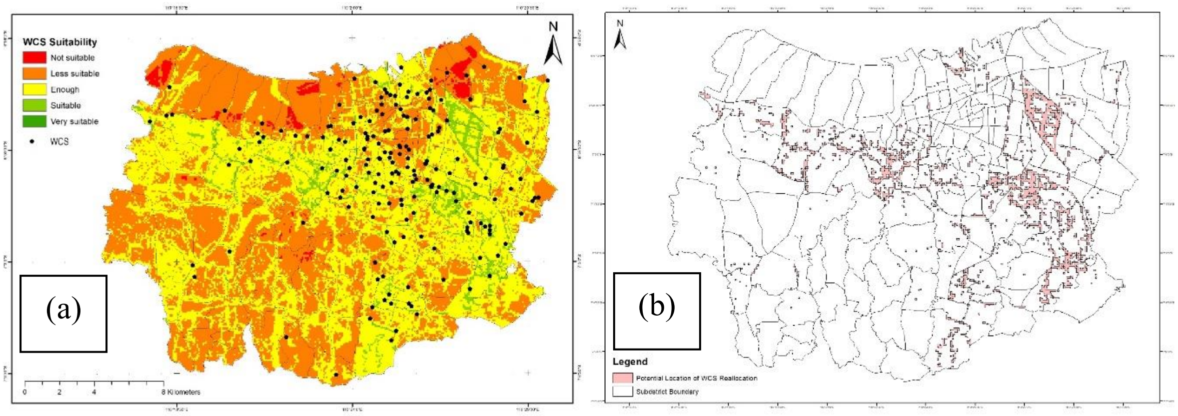

Figure 4.4 (a) Weighted overlay map of optimal WCS reallocation and (b) potential location of new WCS

According to Figure 4.4(b), 744 potential locations are found by assuming the minimum areas for WCS facility is set to 200 meters square. In the current waste collection system, the distribution of WCS is unbalanced. WCSs that require a longer transport distance are not designated when closer options are available for a given waste-generation point. This results in unnecessary WCSs and may cause overflow if there are insufficient waste bins located in the area. An analysis of the existing collection system shows the importance of a systematic approach in determining the number of WCS for MSW, highlighting the potential benefits of this study for the MSW collection system. When identifying suitable locations for waste transfer stations, economic considerations, such as land acquisition costs, development expenses, facility operation expenses, land use, land ownership, and utility availability should be considered (Bosompem et al., 2016). Economic factors are not considered in this study. This approach provides ample space for the future expansion of this study. These potential locations need further investigation, as the municipal government needs to add the WCS to the city.

4.3.4. Location-Allocation Analysis Results

As the final spatial analysis, location-allocation provides a fast methodology for determining the exact location for WCS candidates. As it is shown in Figure 4.5(a), the residential areas which are outside of the existing WCS facilities are prioritized and overlayed with the potential location generated in SMCDA. It is found that 32 new locations are suitable for allocating the WCS in the unserved sub-districts. Besides, according to Figure 4.5(b), from the 198 existing WCS locations, it is found that only 84 locations are needed for maximizing coverage of the service. Based on the summation of location-allocation 1 and 2, there is, at minimum, 116 WCS that should be presented in Semarang City which is much lower than the existing WCS (See Figure 4.5(c)).

It is noted that the new proposed WCS was designed by the demand from the nearest demand point/residential areas in each sub-district. As there would be 81 existing WCS closed, the WCS capacity in the proposed scenario is set to the nearest value of the demand point generated by location-allocation analysis. It is predicted that the uncollected waste from the present scenario is 1,080 m^3/day while the proposed scenario is 462 m^3/day. As it is said by Rathore et al. (2020), the decrease in collection point will reduce the idling time, fuel consumption, emission cause of the collection vehicle, and nuisance cause of waste dumping. Therefore, optimizing collection site can also reduce the emission caused by open waste burning and other health risk caused by inappropriate waste management practices (Ramadan et al., 2023).

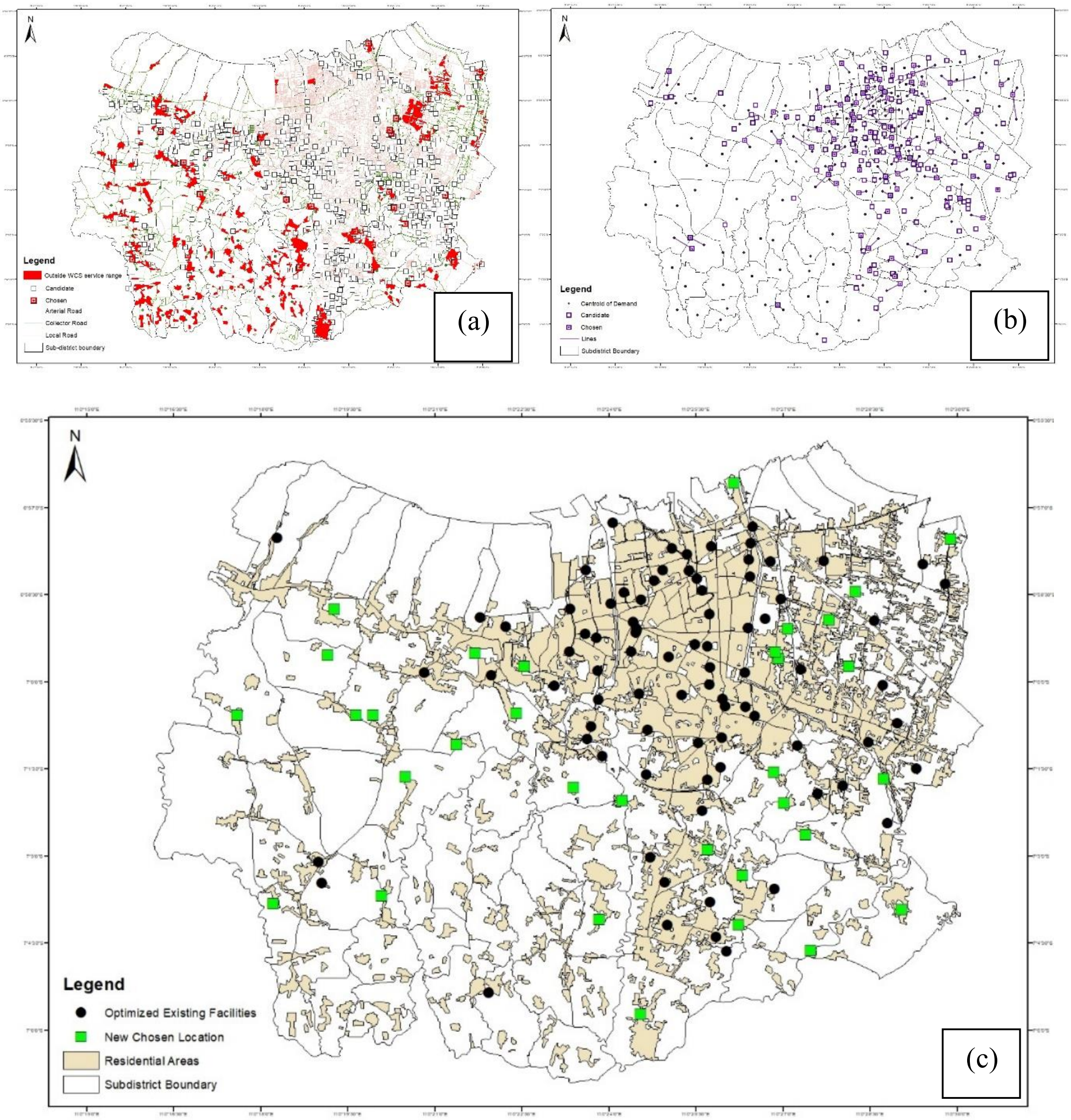

Figure 4.5 Location-allocation result of (a) new WCS location, (b) optimization of existing WCS, and (c) final map result

4.4. Conclusion

The same pattern of ANN and KD was found, where the WCS distribution was clustered in one large cluster area. In the field observations, the clustered area was mainly the urban core. Meanwhile, rural, and peri-urban areas lack WCS, which may lead to improper waste management in these areas. Semarang Tengah and Semarang Selatan Districts have a thicker WCS density compared to other districts. This situation represents the high anthropogenic activity in these two districts. These districts also have the highest population and housing densities compared with other districts. The WCS density in other neighboring districts, such as Semarang Timur, Semarang Barat, Semarang Selatan, and Tembalang, gradually decreases with increasing distance from the urban core. WCS reallocation, as presented in this study, can be used to increase the effectiveness of waste collection activity. Through sequential spatial analysis, there are 116 proposed WCS that can be used to treat 6,019 m^3/day of waste in Semarang City. The optimization of WCS may reduce uncollected waste, thus, reduce open waste burning intensity and other improper waste management practice at the household level in Semarang City.

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Chapter 5
Policy Direction as Countermeasures Reducing OWB in Southeast Asia (SEA) Countries

5.1. Introduction

Waste management in Southeast Asian (SEA) countries is a complex and pressing issue due to the rapid economic and population growth in the region (Ngoc and Schnitzer, 2009). Many SEA countries struggle with inadequate waste management infrastructure and ineffective policies and regulations to address the problem. As a result, open burning, landfill dumping, and illegal dumping of waste are common practices in many SEA countries (Ramadan et al., 2022a). Open burning is setting fire to materials such as agricultural, household/roadside, and forestry waste in an open area rather than properly disposing of them through waste management practices (Ajay et al., 2022).

Open burning can have several negative impacts on the environment and public health (See Figure 5.1). It releases a range of pollutants into the air, including particulate matter, carbon dioxide, and other harmful gases such as carbon monoxide, sulfur dioxide, and nitrogen oxides. These pollutants can have detrimental effects on air quality and contribute to the formation of smog, which can have severe health impacts, particularly for people with respiratory conditions (Chean-Yiing et al., 2022; Ha Chi and Kim Oanh, 2021). In addition to air pollution, open burning can contribute to climate change by releasing short-lived climate pollutants (SLCPs) such as methane and black carbon (BC) (Premakumara et al., 2018). To address this issue, it will be necessary to develop some strategies to reduce the impact of this harmful practice.

Figure 5.1 Backyard waste burning practices in Indones

This chapter aims to analyze some stakeholder initiatives and challenges to stop burning the practice of municipal waste and suggest strategic actions and recommendations for the SEA countries on how to reduce the practice. Therefore, strength-weakness-opportunity-threat (SWOT) and quantitative strategic planning matrix (QSPM) were used to prioritize the strategic actions which have been developed.

5.2. Materials and Methods

5.2.1. Data Collection

To develop the SWOT factors, The Preferred Reporting Items for Systematic reviews and Meta-Analyses (PRISMA) methodologies 2020 were employed by following previous research by Budihardjo et al. (2021) and Page et al. (2021). The step consists of identifying, screening, and inclusion phase of metadata. The first step was retrieving the metadata from the Scopus database. The keyword of "Open Burning" was inputted with several exclusion criteria such as publication age (2022 – 2012), document type (article, conference paper, review), source (journal and proceedings), language (English), and publication stage (final and in-press). From the identification phase, 3,234 documents were found and continued to the next step. In the screening phase, only the documents which were affiliated with the ASEAN countries were included in the analysis. In this part, several documents that did not assess the open or uncontrolled burning in their abstract and title were removed. At the end of this phase, 152 documents were retrieved and analyzed for further treatment. VOSviewer software was used to develop and extract several terms which were connected to the open burning topic.

5.2.2. Qualitative Content Analysis

Qualitative content analysis was used to identify the definition of the SWOT factors from the documents or metadata collected. This analysis consists of summarization and reorganization of the developed SWOT factors (Budihardjo et al., 2021). Therefore, descriptive analyses were also employed to identify stakeholder initiatives to reduce open burning of waste. In this part, there are 5 manuscripts that were in-depth studied which related to 4 initiatives to reduce open burning.

5.2.3. SWOT-QSPM Analysis

From the metadata (152 documents) gathered in the data collection, selected terms from VOSviewer were taken and defined based on qualitative content analysis developed in the previous sub-section. There were 7 and 10 experts were invited to answer the SWOT and QSPM questionnaire, respectively. In the SWOT matrix, rank has values 1 – 4, which is determined by expert judgment. The value 1 stands for weakness, 2 for minor weakness, 3 for strength, and 4 for greater strength. The QSPM methods can objectively measure the appropriate strategies from many strategies. The total attractiveness score (TAS) of strategies (ST1-8) on QSPM methods was determined by rounding up the average value of the questionnaire results. The attractiveness score (AS) is determined to 1-4, which indicates how the strategy will influence or be connected to each factor. Value 1 indicates that the strategy is unattractive, and value 4 is desirable for each factor. The total attractiveness score (TAS) indicates the attractiveness of each factor individually to the strategy. The TAS is then multiplied by the normalized weight generated in the internal/external factors evaluation (IFE/EFE) SWOT matrix. The experts are also invited to prioritize the strategy based on their points of view without considering the SWOT factors. This value is then considered as "individual prioritizing strategy". The value of individual prioritizing strategy and the TAS result were then normalized to get the final score of prioritizations. Other strategies suggested by the experts were also recorded and documented at the end of this chapter.

5.3. Results and Discussion

5.3.1. International and National Initiatives

In this part, the qualitative analysis was used to highlight several initiatives from the previously published literature to reduce the open burning of waste. The first is about the Climate and Clean Air Coalition – Municipal Solid Waste Initiative (CCAC-MSWI) program. The MSW Initiative is a program by CCAC focused on reducing methane and BC emissions in cities. The initiative aims to support cities acting on SLCP reduction and management through partnerships, political will, and technical capacity building. This global initiative is part of Stockholm Convention on Persistent Organic Pollutants (POPs) and the Agenda 2030 of the United Nations (UN) related Sustainable Development Goals (SDGs) under SDGs number 3, 7, 11, and 13 which is initiated to achieve clean air and reduce air pollution from waste sector (Wieser et al., 2021). The other case is the decentralized waste management system which is being endorsed in the Philippines (Premakumara et al., 2018). This decentralized management system was a localized approach for implementing MSWM policy which also may help reduce open burning. The decentralized waste management system can force policymakers to commit to lowering burning activity. The other initiative is community empowerment (Brotosusilo and Naldi, 2021; Budihardjo et al., 2022). In many places in

Indonesia and Thailand, waste management, which empowers local actors to manage their own waste, was introduced and strengthened in the national policy. This initiative helps boost recycling, thus reducing waste-burning practices. The other initiative is the zero-burning policy in Upper Northern Thailand, which focuses on lowering biomass burning event (Yabueng et al., 2020). The government has applied the approach from February to April (dry season) since 2016. As a result, the biomass burning hotspot was reduced in implementing the policy, while this situation could also be done for open burning of waste.

5.3.2. Strategies and Policy Instruments to Reduce OWB in SEA Countries

The factors, challenges, and strategies to reduce the open burning of waste are analyzed using SWOT-QSPM methods. As can be seen in Table 5.1, the SWOT factors and their normalized weight were developed based on the terms generated in the bibliometric analysis. Therefore, the rank was generated from expert judgment. The results showed that strength and opportunity dominate over weakness and threat. It is also found that the overall result of the internal and external factor evaluation (IFE-EFE) scores is 2.131 and 1.436, respectively. According to the strategic position and action evaluation (SPACE) matrix, the type of strategy fits the defensive strategies quadrant meaning the program or project to reduce the open burning of municipal waste is in an industry with low growth and market share (Abbasi et al., 2019). This result implies that the program/project must be proliferating and have a small market share, especially in the waste management system. In this situation, the program/project may face intense competition and cannot grow or increase its profits through aggressive expansion. There is also a possibility that the program/project has already reached a saturation point and is no longer overgrowing. In this quadrant, threats exist, but the external opportunity and project strength can face them, fix unprofitable activities, and reduce costs to increase profitability (Symeonides et al., 2019).

Table 5.1 SWOT factors in the IFE-EFE matrix.

Internal Factors	**Normalized Weight**	**Rank**	**Total**	**External Factors**	**Normalized Weight**	**Rank**	**Total**
<u>Strength</u>				**<u>Opportunity</u>**			
Massive economic growth (S1)	0.021	3	0.063	Potential markets for recycling are increasing (O1)	0.017	2	0.034
Less investment competition of the private sector in the market (S2)	0.082	4	0.328	Job creation potential (O2)	0.055	2	0.11

Internal Factors	Normalized Weight	Rank	Total	External Factors	Normalized Weight	Rank	Total
Cheaper operational cost (S3)	0.036	3	0.108	Financial support from the governments and international funding institutions (e.g., CCAC) (O3)	0.172	1	0.172
Waste can be a renewable source of energy and revenue (S4)	0.156	3	0.468	Regional cooperation and commitment to reducing SLCPs emissions (O4)	0.158	3	0.474
The abundance of human resources (S5)	0.136	3	0.408	Potential alternative energy generation from waste conversion (O5)	0.047	2	0.094
Total Strength			**1.375**	**Total Opportunity**			**0.884**
<u>Weakness</u>				**<u>Threat</u>**			
Lack of capacity of the local authorities and leader (W1)	0.047	1	0.047	Lack of environmental behavior, awareness, attitude, and participation of the residents (T1)	0.194	1	0.194
Insufficient waste management infrastructures and services (W2)	0.234	1	0.234	Environmental management inconsistencies (T2)	0.026	1	0.026
Gaps in regulation, policy, law enforcement, and program or plan (W3)	0.094	2	0.188	Public health and environmental risks, especially for informal actors (T3)	0.154	1	0.154
Limited financial and technical resources (W4)	0.163	1	0.163	Unsystematic coherence and political instability between regional, national, and local authorities (T4)	0.178	1	0.178
Slow response to new initiatives and changes (W5)	0.031	4	0.124				
Total Weakness			**0.850**	**Total Threat**			**0.552**
Total IFE			**2.131**	**Total EFE**			**1.436**

The literature survey shows several possible actions and recommendations for reducing open burning from the waste sector. First, there is a need to provide technical support for data collection and management, coaching and mentoring of available emission calculation tools, knowledge sharing, and baseline understanding (ST1) (Premakumara et al., 2018). Second is strengthening enforcement and issuing or implementing laws, regulations, legislation, and national action plan on the waste management sector (ST2) (Ramadan et al., 2022a). The third is upgrading and expanding existing waste infrastructure, including waste collection, transportation, and disposal facilities (ST3) (Hong Phuong et al., 2022; Ramadan et al., 2022a).

There is also a need to build the capacity of national and local authorities or leaders (ST4) (Hong Phuong et al., 2022; Ramadan et al., 2022b) and provide better incentives and access to the market for waste management initiatives and recycling activities (ST5) (Budihardjo et al., 2022). Besides, stimulating public and private sector participation (PPP) to support waste management infrastructures and waste service provisions (ST6) (Jeske et al., 2021) should be conducted in line to reduce waste burning. The other thing is improving community participation in waste management, monitoring, campaign, and education through awareness-raising activities (ST7) (Hong Phuong et al., 2022; Ramadan et al., 2022b) and promoting recycling activities of informal actors at household levels, including their safety and working conditions (ST8) (Ramadan et al., 2022a, 2022b). The sum of TAS is determined and sequenced according to the strategy's highest to lowest priority, which can be seen in Table 5.2.

Table 5.2 Priority recommendations for reducing open burning from the waste sector based on QSPM methods.

Factors	**Normalized Weight**	**ST1**	**ST2**	**ST3**	**ST4**	**ST5**	**ST6**	**ST7**	**ST8**
S1	0.021	0.021	0.021	0.042	0.042	0.021	0.042	0.042	0.042
S2	0.082	0.082	0.164	0.164	0.164	0.328	0.164	0.164	0.082
S3	0.036	0.072	0.072	0.072	0.036	0.072	0.108	0.072	0.072
S4	0.156	0.312	0.156	0.312	0.312	0.312	0.312	0.312	0.468
S5	0.136	0.272	0.136	0.136	0.272	0.272	0.272	0.408	0.272
W1	0.047	0.141	0.094	0.094	0.188	0.047	0.094	0.047	0.047
W2	0.234	0.468	0.234	0.702	0.468	0.468	0.468	0.468	0.468
W3	0.094	0.188	0.282	0.188	0.188	0.188	0.188	0.188	0.188
W4	0.163	0.163	0.326	0.326	0.326	0.326	0.326	0.326	0.326
W5	0.031	0.062	0.062	0.062	0.062	0.031	0.031	0.062	0.031
O1	0.017	0.017	0.017	0.017	0.034	0.051	0.051	0.051	0.051
O2	0.055	0.11	0.055	0.11	0.11	0.11	0.22	0.165	0.165
O3	0.172	0.344	0.172	0.688	0.516	0.344	0.344	0.344	0.344
O4	0.158	0.632	0.474	0.316	0.316	0.158	0.158	0.316	0.158
O5	0.047	0.094	0.094	0.141	0.094	0.094	0.094	0.094	0.047
T1	0.194	0.194	0.388	0.388	0.388	0.388	0.388	0.776	0.388
T2	0.026	0.078	0.052	0.052	0.078	0.078	0.052	0.052	0.052
T3	0.154	0.154	0.154	0.154	0.154	0.308	0.308	0.308	0.616
T4	0.178	0.356	0.534	0.356	0.712	0.356	0.356	0.356	0.356
Total Strategy Attractiveness Score (TAS)		**3.827**	3.76	3.487	4.32	4.46	3.952	3.976	4.551

According to Table 5.2, the highest score is achieved by ST3, followed by ST7 and ST4. This result means that upgrading and expanding existing waste infrastructure is the most attractive strategy considering all external and internal factors in reducing open burning from the waste sector. Therefore, the experts could prioritize the developed strategy without considering the SWOT factors. Interestingly, the results showed a different pattern where the higher score is given to the ST2, which is strengthening enforcement and issuing some laws or regulations that can support actions in the waste management sector. To achieve a good priority recommendation, the normalized scores of QSPM and the individual prioritizing strategy are summarized, and the highest strategy that can be given is ST4, followed by ST7 and ST2. Detailed information on the priority recommendations can be found in Table 5.3.

Table 5.3 Priority recommendations based on the total of individual prioritizing strategy and QSPM normalized scores.

Priority	Strategy Code	Normalized Score		Total Normalized Score	Description
		Individual Prioritizing Strategy	TAS QSPM		
1	ST4	0.164	0.136	0.300	Building the capacity of national and local authorities or leader
2	ST7	0.131	0.139	0.270	Improving participation of the community in waste management, monitoring, campaign, and education through awareness-raising activities
3	ST2	0.153	0.107	0.259	Strengthening enforcement and issuing or implementing law, regulation, legislation, and national action plan on the waste management sector
4	ST3	0.122	0.132	0.254	Upgrading and expanding existing waste infrastructure, including waste collection, transportation, and disposal facilities
5	ST1	0.136	0.115	0.251	Providing technical support for data collection and management, coaching, and mentoring of available emission calculation tools, knowledge sharing, and baseline understanding
6	ST6	0.111	0.122	0.233	Stimulating public and private sector participation (PPP) to support waste management infrastructures and waste service provisions
7	ST8	0.089	0.128	0.217	Promoting recycling activities of informal actors at household levels, including their safety and working conditions
8	ST5	0.094	0.121	0.215	Providing better incentives and access to the market for waste management initiatives and recycling activities

Based on the expert opinion, other factors influencing the actions to reduce open burning from the waste sector include the need for more consideration for the sustainability of waste management-related projects. This factor is necessary as project sustainability is often ignored and left behind after the projects are done. Therefore, some strategies have also been added for future research implementation. First, the performance of a pricing system for carbon emissions resulting from open burning should be considered as it can be a good indicator of the sustainability of a project. Second, promoting waste separation at source by benchmarking or rewarding system. The formalization of informal recycling may also be considered to reduce the open burning activities in SEA countries. Fourth, all governments at the city level should have a clear-cut responsibility between regulators, operators, and inspectors. Since solid waste management is a public service that the municipal government must provide, straightforward tasks and responsibility for each stakeholder in the government body may give a better understanding of the service itself.

5.4. Summary

The study found that the reduction of open burning intensity in SEA countries needs some defensive strategies that need the strength factors to take advantage of the opportunities. Therefore, several action recommendations are developed and prioritized as follows: (1) building the capacity of national and local authorities or leader; (2) improving participation of the community in waste management, monitoring, campaign, and education through awareness-raising activities; (3) strengthening enforcement and issuing or implementing law, regulation, legislation, and national action plan on the waste management sector; (4) upgrading and expanding existing waste infrastructure, including waste collection, transportation, and disposal facilities; (5) providing technical support for data collection and management, coaching, and mentoring of available emission calculation tools, knowledge sharing, and baseline understanding; (6) stimulating public and private sector participation (PPP) to support waste management infrastructures and waste service provisions; (7) promoting recycling activities of informal actors at household levels, including their safety and working conditions; and (8) providing better incentives and access to the market for waste management initiatives and recycling activities. Several strategies can also be considered to reduce open burning emissions, such as implementing a carbon pricing system for carbon emissions at the municipal level, promoting waste separation at sources, formalizing informal recycling, and more precise job descriptions for each level of stakeholder inside the government body.

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Chapter 6
Conclusions and Further Studies

6.1. Conclusions

There are several important findings that can be generated from the previous chapters.

1. Mismanagement of waste in many developing countries is the cause of open burning practices. Besides, there are many other reasons that make people do these practices. Open burning can emit significant pollutants which negatively impact human health and the environment. Meanwhile, the data on municipal and e-waste burning still needs to be improved in the SEA context. It is estimated that 57% of waste in SEA countries is not collected, which may be subjected to improper waste disposals such as open burning, direct dumping into the environment, or others.
2. In Semarang City, most of residents are found to burn the waste once a week. Open burning is the second disposal practice of municipal waste after waste collection by the officers. Higher burning intensity and activity with a lower plastic and organic portion of burned waste was found in the dry season. Rural areas contribute to the highest per capita waste burning incidents than the urban areas.
3. People around the burning pile have the potential impact on the metals- and hydrocarbon-bound particulate. Based on the health risk assessment, children are the vulnerable group of certain chronic diseases from the OWB practices. Rural and peri-urban areas are the biggest contributor to the emission of OWB. An innovative countermeasure related to waste management shall be considered in these areas.
4. Waste collection site (WCS) in Semarang City is clustered in urban areas. While the rural and peri-urban areas lack WCS, improper waste management happens in this area. Therefore, WCS reallocation could be a step forward towards reduction of uncollected waste in Semarang City. Based on the model calculated in the optimization of WCS, the reallocation of WCS can reduce the number of WCS, thus reduce the emissions and number of wastes burning intensity in the rural and peri-urban areas.
5. Some initiatives have already been implemented to reduce the open waste burning practice, which can be best practices for reducing the waste burning intensity. Reduction of open waste burning in SEA countries is in the defensive strategy quadrant, meaning the project has low growth and market share. Building the capacity of national and local

authorities or leaders is the most attractive and appropriate strategy to reduce open waste burning in SEA countries.

6.2. Limitation of This Study

There are several limitations in the present study. For the temporal study of OWB activities (second chapter), the estimation of waste generation in Semarang City is based on the literature study which has bigger uncertainties. The same limitation of the study happens in the third chapter since the data for estimating the risk and environmental impact of OWB is mostly coming from the literature review. Updating the value of this data would be necessary to reduce uncertainty in the modelling results. In the fourth chapter, there is only one expert involved in the decision making of SMCDA which makes the decision results more subjective. The last chapter focuses on the SWOT QSPM analysis for prioritizing the action of reducing OWB at regional level. In this chapter, a focus group discussion (FGD) is needed before filling out the questionnaire to reduce inconsistencies during the filling process. As we only send an email to the respondents, they might have a different perception on conceiving the questions.

6.3. Future Research Directions

Future OWB studies should consider the following points.

1. Behavior change studies on OWB practice should be conducted by considering several factors. A deeper understanding of the motivation of people to burn their waste could have a significant impact on the ongoing studies.
2. In Indonesia, there is a big difference between city and regency. The bigger proportion of rural cluster area is bigger than the city while the service area is also much bigger. OWB practice shall be higher in the regency area since the less waste collection efficiency is presented. Therefore, it is suggested to study this area, especially how the waste management model should be implemented in the area.
3. Future studies should explore the social and economic factors that could contribute to the reduction of unmanaged waste practices in Semarang City, as well as determine whether the mismanaged waste in this transition area is higher than that in rural and urban areas.
4. The impact of dioxin as part of the plastic burning emission should also be considered in future research. Therefore, as it is written in the limitation of the study, a higher tier of data is needed to reduce the uncertainty of the assessment. Besides, actual exposure

level and duration to the OWB piles should be determined and considered in the future study.

5. Future research should also consider the optimization of waste collection route by using the proposed collection points. Therefore, to increase the possibility of emission reduction in the waste sector, there should be a consideration regarding the optimization of recycling point in Semarang City.
6. Focus group discussion (FGD) for policy makers should be conducted to determine the possible actions for each country to reduce the OWB practices.

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