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Potential Risks of Open Waste Burning at the Household Level: A Case Study of Semarang, Indonesia

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ABSTRACT

Open waste burning (OWB) is a common disposal practice in several countries. However, this activity can serve as a source of particulate matter and black carbon, which possess a greater greenhouse effect potential than CO_2. Moreover, particulates can produce chronic health impacts on residents around the burning areas. Therefore, the present study aimed to examine the environmental effects and health risks associated with the open burning of household waste in Semarang, Indonesia. Four steps were followed to answer the research questions: (1) data collection through a random questionnaire survey, transect walk, and field survey; (2) estimation of environmental risk using the IPCC calculation method; (3) multiplication of emission factors to determine black carbon emissions; and (4) estimation of health risks based on chemical speciation bound to particulate matter. Open burning remained the second most common waste disposal practice even after the implementation of waste collection services by the government. Specifically, approximately 240.28 tons of waste is not collected by the environmental agency service every day, and 88.6% of the uncollected waste in the city is openly burned. Plastic burning contributed to the highest emission share among waste components, and annual total emissions due to OWB were estimated at approximately 53,809.66 tons. Although the carcinogenic risk was low, non-cancer disease risk exceeded the standard. Therefore, direct exposure of residents to OWB may pose significant health risks. The present work fills the scientific and knowledge gaps in the OWB studies.

Keywords: Open waste burning, Chemical speciation, Environmental risk, Health risk

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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),

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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). In particular, a number of 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 been 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 the cancer posed by dioxin is 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 the 9% of mortalities from $PM_{2.5}$ emissions is 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).

To this end, the present study is a continuation of previous work by Ramadan *et al.* (2022b). Specifically, the existing disposal practices were identified through a randomized survey among the citizens of Semarang. In addition, the transect walk method was used to detect the potential waste burning incidents, and environmental risk was determined by multiplying the total waste burning in Semarang by the potential BC and GHG emissions 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 was focusing to the FA/BA residues and not discussing the exhaust gas because the information was already presented in the previous work. Our findings can fill the gaps in high-level data inventories of OWB and support appropriate policy and decision making aimed at reducing emissions from the waste sector.

2 METHODS

The study involved four sub-activities: an online questionnaire survey, a transect walk survey, environmental risk assessment, and health risk estimation. Detailed information on each sub-activity is provided in the following sub-sections.

2.1 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 Eq. (2) were used.

$$\sum_{a-d} FM_{OB} = \frac{FM_{OB_{a-d}}}{r_{a-d}} \tag{1}$$

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

In Eq. (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.

2.2 Transect Walk Survey

The transect walk survey followed the method described by Ramadan *et al.* (2022b). The surveys were conducted at the same location and following the same methods during the dry season from May to July 2021 in semi-lockdown due to the COVID-19 pandemic. The surveys were conducted in 16 of the 144 sub-districts of Semarang. Sub-districts with similar geography, demographics, and waste collection services were selected based on clustering results. In each sub-district, a transect line of approximately 10 km was set in a loop or straight line. Each cluster comprised four sub-districts representing urban, inner peri-urban, outer peri-urban, and rural areas. Before the field survey, the surveyor was trained to ensure the same perception of waste pile measurement along the transect line. The surveyor documented the waste pile; flattened it; and then recorded the coordinates, dimensions, and burning conditions. Unburnt waste piles along each transect line were characterized and brought to the laboratory for burning tests. Detailed information on the burning test can be seen in Ramadan *et al.* (2022b). The amount of waste burned at the city scale was extrapolated by multiplying the average waste pile density by the total sub-district area in each cluster. This coarse estimation was used to calculate the environmental risk of OWB.

2.3 Environmental Risk Assessment

After estimating the total waste burned, the environmental risk caused by CO_2, CH_4, and N_2O emissions was calculated using the equation derived from 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). Eqs. (3–6) were used to estimate CO_2, CH_4, N_2O, and BC emissions from open burning incidents in Semarang.

$$\mathrm{CO_2\ Emissions} = \sum_j \left(M_{wbj} \times dm_j \times CF_j \times FCF_j \times CE_j \right) \times \frac{44}{12} \tag{3}$$

$$\mathrm{CH_4\ Emissions} = \sum_j \left(M_{wbj} \times CH_4\, EF_j \right) \times CoF \tag{4}$$

$$\mathrm{N_2O\ Emissions} = \sum_j \left(M_{wbj} \times \mathrm{N_2O}\, EF_j \right) \times CoF \tag{5}$$

$$BC\ \mathrm{Emissions} = \sum_j \left(M_{wbj} \times BC\ EF_j \right) \times CoF \tag{6}$$

$$\mathrm{Total\ GWP} = \sum_j Em \times GWP_k \tag{7}$$

M_{wb} represents the wet weight of waste burned in the city (t 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 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.*, 2019). 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. (7).

2.4 Health Risk Estimation

The 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

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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.

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 Eqs. (8–10), presented by Keshavarzi *et al.* (2015), Liang *et al.* (2019), and Khan *et al.* (2020).

$$D_{ing} = \frac{c \times IR_{ing} \times FE \times ED \times CoF}{BW \times AT} \tag{8}$$

$$D_{derm} = \frac{c \times SA \times AF \times ABS \times FE \times ED \times CoF}{BW \times AT} \tag{9}$$

$$D_{inh} = \frac{c \times IR_{inh} \times FE \times ED}{PEF \times BW \times AT} \tag{10}$$

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 Eqs. (11–14) (Liang *et al.*, 2019), and chronic risk exposure caused by heavy metals from each pathway was determined using Eq. (15). The hazard index (HI) was determined to estimate the overall chronic risk (Eq. (16)). 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 kg^{-1} d^{-1} of CSF_{ing}. The human threshold set by the U.S. EPA (2001) for CR is $> 10^{-6}$. The higher the CR value, the greater the carcinogenic risk to humans (Khan *et al.*, 2020).

$$CR_{ing} = D_{ing} \times CSF_{ing} \tag{11}$$

$$CR_{derm} = D_{derm} \times \frac{CSF_{derm}}{GIABS} \tag{12}$$

$$CR_{inh} = D_{ing} \times CSF_{inh} \tag{13}$$

$$CR_{total} = CR_{ing} + CR_{derm} + CR_{inh} \tag{14}$$

$$HQ_i = \frac{D_i}{RfD_i} \tag{15}$$

Table 1. Reference data for exposure factors.

Exposure variable	Child	Adult	Unit	Reference
Ingestion rate (IR_{ing})	200	100	mg d^{-1}	U.S. EPA (2011)
Inhalation rate (IR_{inh})	7.6	20	m^3 d^{-1}	U.S. EPA (2011)
Frequency of Exposure (FE)	180	180	d y^{-1}	Ferreira-Baptista and De Miguel (2005)
Exposure Duration (ED)	6	30	y	U.S. EPA (2011)
Average body weight (BW)	16.2	61.8	kg	U.S. 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}	U.S. EPA (2011)
Skin adherence factor (AF)	0.7	0.07	Mg cm^{-2} d^{-1}	U.S. EPA (2011)
Skin absorption fraction (ABS)	0.001	0.1	unitless	U.S. EPA (2011); Man *et al.* (2010)
Particle emission factor (PEF)	1.36×10^9	1.36×10^9	m^3 kg^{-1}	U.S. EPA (2011)
Gastrointestinal absorption factor (GIABS)	1	1	unitless	U.S. EPA (2011)
Ingestion cancer slope factor (CSF_{ing})	7.3 for hydrocarbon 0.0085 for Pb		mg $kg^{-1}d^{-1}$	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}	Khan *et al.* (2020)
Skin cancer slope factor (CSF_{der})	25 for hydrocarbon		mg kg^{-1} d^{-1}	Knafla *et al.* (2006)

$$HI = \sum_{i=1}^{n} HQ_i \tag{16}$$

CSF represents the ingestion (CSF_{ing}), dermal (CSF_{derm}), and inhalation (CSF_{inh}) cancer slope factors (mg kg^{-1} d^{-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 1.

3 RESULTS

3.1 OWB Practices in Semarang

Indonesia has two classifications of cities, namely 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. This may reduce the possibility of improper waste disposal practices. However, from our randomized questionnaire survey, OWB was found to be the second most common waste disposal practice. As shown in Fig. 1(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 Fig. 1(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

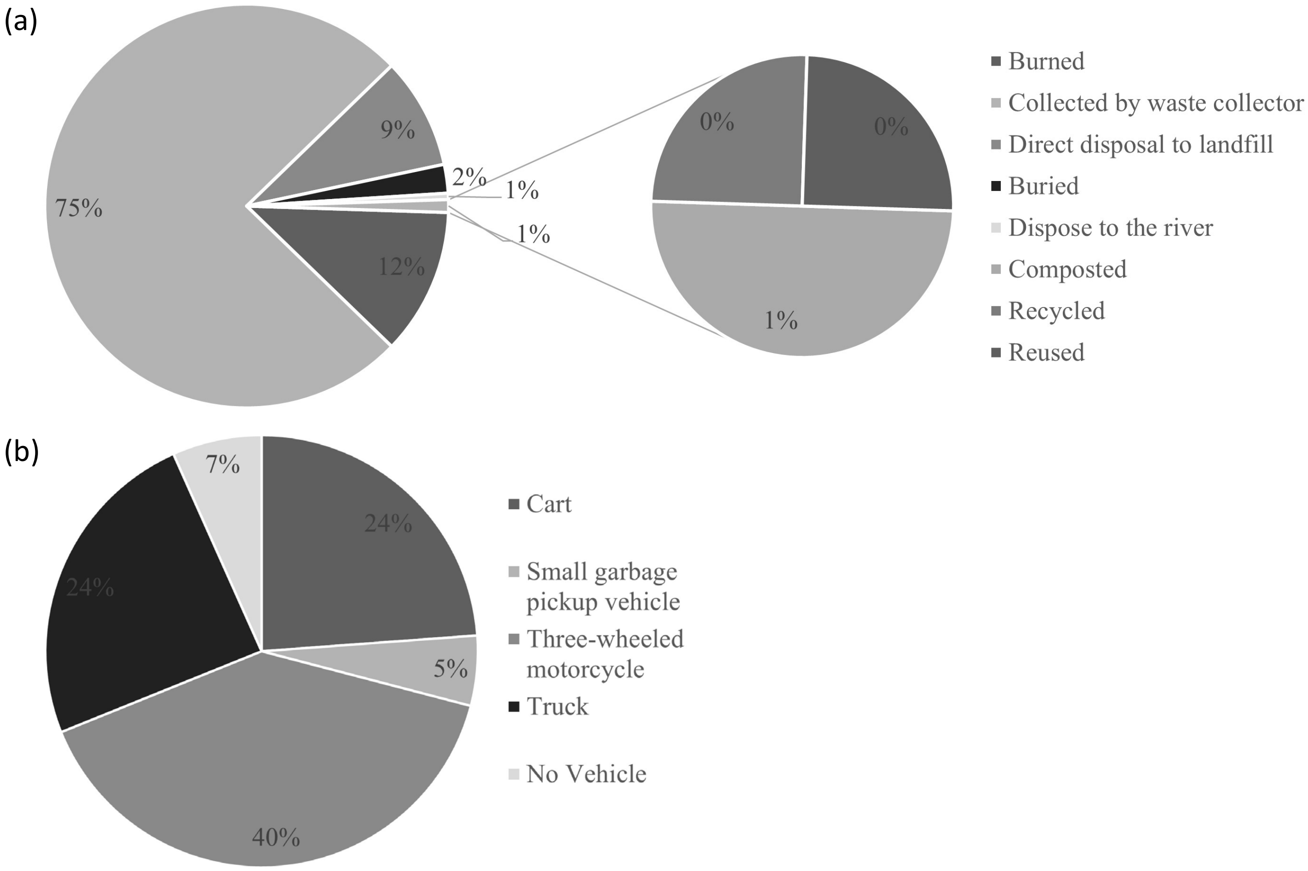


Fig. 1. (a) Common waste disposal practices and (b) door-to-door waste collection service vehicles in Semarang.

rural and outer peri-urban areas are most likely to burn their waste rather than bringing it to the nearest WCS.

Fig. 2 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.

3.2 Number of OWB Incidents

Burned waste in Semarang exhibited specific characteristics. As such, organic matter and wood or branches dominated the waste composition of burned waste in Semarang (62.7%). As shown in Fig. 3, plastic waste (15.7%) is burned along with organic waste, which may contribute to higher BC emissions. Inert materials, such as glass and metals, were detected in outer peri-urban area, which accounting for 22.8% 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 as much as 82.4%. 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 which is 16.3% and 25.0%. Then, the greatest contributor to plastic waste burning was the inner peri-urban area. Interestingly, outer peri-urban area

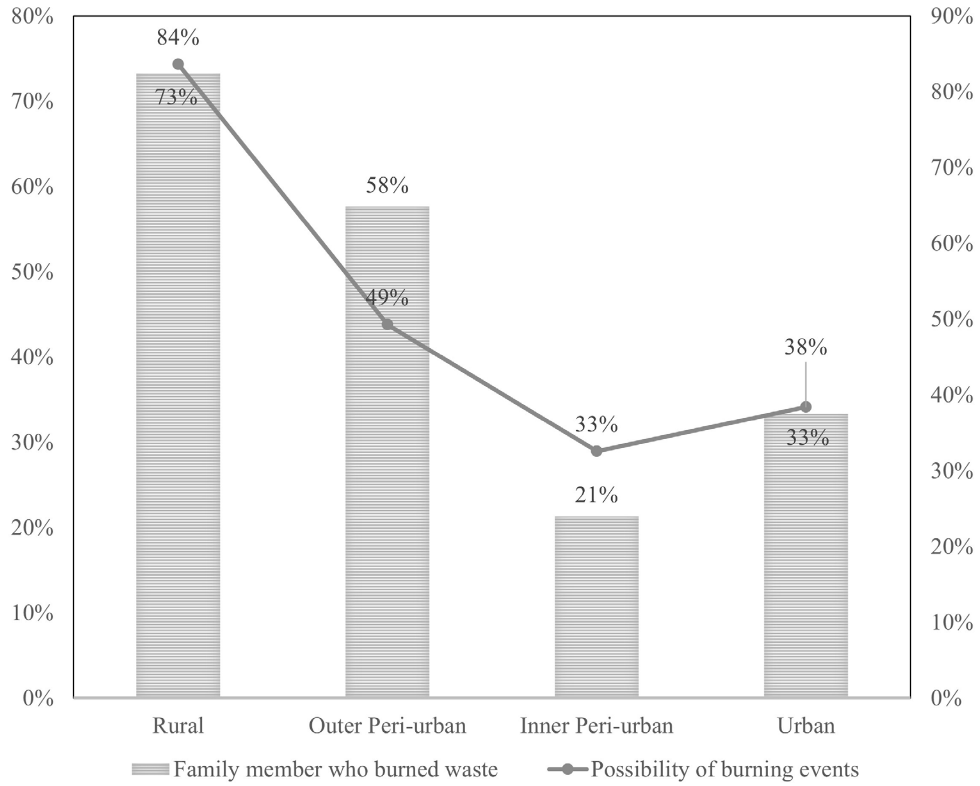
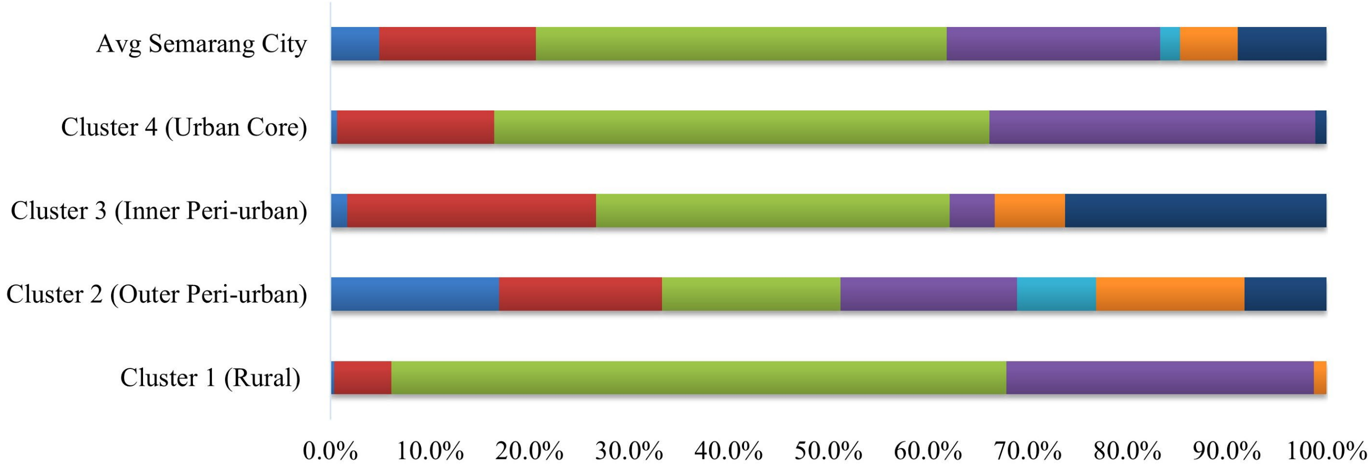


Fig. 2. Fraction of family members who burned waste and the possibility of burning events in each cluster.



	Cluster 1 (Rural)	Cluster 2 (Outer Peri-urban)	Cluster 3 (Inner Peri-urban)	Cluster 4 (Urban Core)	Avg Semarang City
Paper	0.4%	17.0%	1.7%	0.7%	4.9%
Plastic	5.8%	16.3%	25.0%	15.8%	15.7%
Organic	61.8%	17.9%	35.5%	49.7%	41.2%
LWTR	30.9%	17.7%	4.5%	32.7%	21.5%
Glass	0.0%	7.9%	0.0%	0.0%	2.0%
Metals	1.3%	14.9%	7.1%	0.0%	5.8%
Miscellaneous	0.0%	8.2%	26.3%	1.1%	8.9%

Fig. 3. Composition of burned waste.

Table 2. Average pile density, number of incidents, and waste burned per capita of open waste burning based on transect survey.

Cluster name	Transect walk survey results					Scaling-up	
	Total area (km^2)	Total piles	Average piles density (ton km^{-2})	Total burning Incidents	Waste burned per capita (kg waste day^{-1})	Coarse estimation of OWB in the city (ton day^{-1})	(%)
Cluster 1 (Rural)	57.44	58	1.17 ± 0.96	48	1.10 ± 0.92	59.38	27.89
Cluster 2 (Outer Peri-urban)	152.87	35	0.75 ± 0.51	33	0.31 ± 0.26	100.91	47.39
Cluster 3 (Inner Peri-Urban)	79.18	33	0.59 ± 0.66	27	0.09 ± 0.13	41.39	19.44
Cluster 4 (Urban Core)	60.49	28	0.210 ± 0.134	28	0.012 ± 0.008	11.26	5.29
Total	349.97	154		136		212.94	100

showed a greater diversity of waste being burned since almost all the waste composition except glass and miscellaneous exceed 10% of total composition. The average pile density of waste found in Semarang was 232.62 ± 26.98 kg m^{-3}. 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%).

As shown in Table 2, waste burning per capita was the highest in rural areas, followed by the outer peri-urban, inner peri-urban, and urban core areas. Since the scaling-up method is based on total areas in the cluster, outer peri-urban areas were the highest potential contributor to OWB. Overall, rural and outer peri-urban areas are the potential hotspots of OWB. Even urban core areas were not free of burning incidents, indicating that burning practices are preferred for waste management even in urban cores regardless of the availability of a waste collection system. However, as in the case of urban cores, in the other places, the frequency of non-burning incidents was lower than that of burning incidents.

3.3 Environmental Impact of OWB

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). 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.

3.4 Health Risks of OWB

As shown in Table 4, 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,

Table 3. 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

Table 4. Hydrocarbons and metals detected in FA and BA.

Hydrocarbon compounds	Abbreviation (Rings)	Concentration (mg kg^{-1})		Metal elements	Concentration (mg kg^{-1})	
		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			

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 4, 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.

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 5. 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 5. 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}

Table 6. 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
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

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 6 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).

4 DISCUSSION

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.*, 2018). 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.*, 2018), India (Nagpure *et al.*, 2015), 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).

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). Ramadan *et al.* (2022b) conducted a transect survey in the rainy season and noted that 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. 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). 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.*, 2018). 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 (See Table 7). 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.

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).

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.

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

Table 7. 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 = 11.26 Inner peri-urban = 41.39 Outer peri-urban = 100.91 Rural = 59.38 Overall = 212.94	9.7	Urban = 0.012 Inner peri-urban = 0.090 Outer peri-urban = 0.306 Rural = 1.098 Overall = 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)
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)

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, 2013). 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 consideration when evaluating the cause behind OWB practices in the city level.

5 CONCLUSIONS

To the best of our knowledge, the present study is the first to comprehensively reveal the extent of waste burning in Semarang, the associated amount 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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ADDITIONAL INFORMATION AND DECLARATIONS

Credit Author Statements

Bimastyaji Surya Ramadan: Conceptualization, methodology, writing-original draft, writing – review and editing; Raden Tina Rosmalina: Methodology, writing-review, conceptualization; Munawir: Visualization, investigation; Syafrudin: Funding acquisition, writing – review and editing, resources, Hafizhul Khair: Software, methodology, formal analysis; Indriyani Rachman: Conceptualization, validation, writing-original draft; and Toru Matsumoto: Supervision, formal analysis, validation.

Supplementary Material

Supplementary material for this article can be found in the online version at https://doi.org/10.4209/aaqr.220412

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