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EVALUATION OF DUST EMISSION RATE FROM LANDFILL MINING ACTIVITIES

Mohammed Zari [1,2,*], Richard Smith [1,3] and Rebecca Ferrari [1]

[1] *University of Nottingham, Faculty of Engineering, Chemical and Environmental Engineering Department, Coates Building, University Park, Nottingham NG7 2RD, United Kingdom*
[2] *King Abdulaziz University, Faculty of Environmental Sciences, Department of Environment, 21589 Jeddah, Saudi Arabia*
[3] *Industrial Chemicals Ltd, Titan Works, Hogg Lane, Grays, Essex RM17 5DU, United Kingdom*

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ABSTRACT

Mining operations are one of the most significant sources of particulate emissions in the atmosphere. Landfill mining (LFM) process activities, including excavation, screening, shredding, and equipment handling, have the potential to emit particulate matter into the environment as short-term episodic emissions during operational periods. Previous investigations show that LFM activities can potentially cause human health and environmental impacts through exposure of these emissions. This paper evaluates the dust emission rate of such activities to understand factors responsible for higher emissions rate and determine where any pressure points exist in order to mitigate risk. Nine empirical formulas were adopted from surface mining activities, including point, line, and area sources of activity. Parameters identified in the equations were adjusted to LFM application conditions. From emission results, it is observed that point source activities were the major sources of emission. The study area was divided into multiple phases and one phase cumulative for the maximum/average/minimum point sources emissions over the lifetime of the landfill mining operation calculated in this study are approximately 5.04 tonnes (t) / 3.23 (t) / 1.61 (t), respectively. However, the one phase cumulative for the maximum/average/minimum line sources emissions over the lifetime of the landfill mining operation are approximately 100.8 (kg/m) / 40.32 (kg/m) / 20.16 (kg/m), respectively. Mitigation measures to control high emission rate of LFM related activity, such as utilising tankers or bowsers to spray water around the LFM area, to control airborne emissions, should be considered. The results of this research are expected to inform air dispersion modelling for environmental impact assessment studies of air pollution.

1. INTRODUCTION

Particulate matter (PM) is one of the principal pollutants resulting from surface mining operations (here also termed 'dust') (Luo et al., 2021; W. Wang et al., 2022). Landfill mining (LFM) practices also potentially contribute to the production of dust by a series of on-site mechanical processes into the environment (Zari et al., 2022). Dust released during these activities is directly introduced to the ambient atmosphere (Patra et al., 2016), which includes potentially toxic materials in a diverse size range (Entwistle et al., 2019; Guo et al., 2021). These materials can be transported into the surroundings and affect local communities if not controlled properly (Pandey et al., 2014).

Atmospheric particles (dust), with a sizes range of <60 μm can play an important role in the transfer of pollutants into the environment. (Csavina et al., 2012; Omrani et al., 2017). Due to the potential distance, speed, and aerial extent with which contaminants can be transported in the environment, atmospheric transport is likely to be the most significant transport pathway relating to potential impact on the environment and human health (Csavina et al., 2012; Elmes & Gasparon, 2017; Kim et al., 2015; Liu et al., 2023). Furthermore, in the very short-term (hours to days), airborne particles have the greatest ability to transport contaminants within the environment (Csavina et al., 2012; Manisalidis et al., 2020).

Health studies show a strong association between airborne PM and adverse effects such as obstructed airways, cancer, decreased lung function and capacity, pneumoconiosis, and increased cardiovascular disease. (Cristaldi et al., 2022; W. Wang et al., 2022). Smaller particles, especially those <10 μm, are more likely to be inhaled through the respiratory system (Gao et al., 2017). Particles with diameters ranging from 30 to 10 μm remain suspended in the air for

* Corresponding author:
Mohammed Zari
email: mohammed.zari@nottingham.ac.uk

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a short time before getting trapped in the nostrils or mouth and being ingested. (Luo et al., 2021; Patra et al., 2016).

Previous findings by Dino et al. (2016) and Zari et al. (2022) that assessed the suitability of historic landfill sites for LFM, showed that some potentially toxic elements within landfilled waste materials have a potential risk to the human health and the environment. In addition, Adenuga et al. (2022), Fang et al. (2022) and Houessionon et al. (2021) demonstrated that potential risk exposure above an acceptable range exist from recycling activities worldwide. Air dispersion modelling enables the Environmental Impact Assessment of air pollution from such activities to be considered. In order to assess the air quality impact, equations for calculating predicted emission rates (also called emission inventory), are necessary in the absence of actual monitoring data. Only equations for calculating emission rates associated with LFM activities are considered in this study. Generally, there are four methods of particulates emission estimation: mass balance calculations; engineering calculations; sampling or direct measurements; and application of emission factors. For this study, an engineering calculations approach was adopted as the most applicable.

For each source of activity in the mining sector, an equation for estimating emissions has been established (Neshuku, 2012). The United States Environmental Protection Agency (USEPA) and Central Mining Research Institute in India has implemented extensive work to develop emission estimating formulas for these activities (Chakraborty et al., 2002; Chaulya et al., 2003; Chaulya et al., 2019; Lal & Tripathy, 2012; Neshuku, 2012; Richardson et al., 2019). A number of field observations of dust formation from all individual activities was validated against these equations (Lal & Tripathy, 2012). The formulas used in this study were derived from surface mining activities by Chaulya (2006) since the focus of those studies was based on surface mining, which allowed for a more precise prediction of dust production (Triantafyllou et al., 2021). Chaulya (2006) has worked extensively on the formulation and validation of a series of Gaussian dispersion equations for determining the emission rate from various surface mining operations, taking into account ground level, vertical, and horizontal dispersion factors. A field investigation of three surface mines and an average of three data from each monitoring station serve as the foundation for each activity's calculation. It was discovered that the validation study's accuracy ranged from 92% to 97% of the actual field measurement data (Lilic et al., 2018; Patra et al., 2016). These equations have taken into account the key determining factors for each specific activity (Chaulya, 2006).

The purpose of this study is to evaluate the dust that can become airborne during LFM activities. This evaluation was achieved by adjusting parameters identified in the equations to LFM applications and to represent local weather conditions. Nine empirical formulas were adopted to account for nine different LFM processes: cover removal loading, recovered waste loading, cover removal unloading, recovered waste unloading, haul road, transport road, exposed cover removal waste, exposed pit surface and screening plant.

2. MATERIALS AND METHODS

2.1 Site selection and description

Landfilling was mainly implemented by local authorities and private sector companies in the UK prior to the introduction of regulatory standards and modern engineering, implementation of the Landfill Directive, and modern regulation via the environmental permitting regulations. The landfill site selected for this study was based on the identification of a site that is representative of the many hundreds of landfill sites in the UK of this type which contain historic deposits >30 years old, primarily being filled during the 1970s, 1980s, and early 1990s. The waste material used in this research originates from a closed landfill located in Norfolk, eastern England. Norfolk County Council utilised the site as a Local Authority landfill, filling in a mineral working that extracted sand and gravel. The site is about 10 m deep, has a surface area of 3.12 hectares (about 7.71 acres), and was operational from 1978 until 1986. The landfill had no liner or leachate collection system (non-engineered). In 1998, an engineered geo-synthetic clay liner cap was installed to allow for intermittent landfill gas extraction whilst also reducing rainwater infiltration and leachate generation. Municipal solid waste (MSW) with commercial and industrial waste were the dominant wastes accepted in the site. The site was operated and regulated on a co-disposal basis in the early days of waste regulation under the Control of Pollution Act (1974). The landfill is composed of a complex mixture of organic and inorganic waste.

2.2 Excavation and sample processing

Using a rotary drilling rig at a depth of 7-8 m, approximately 40 kg of MSW was collected from four individual wells, numbered 1901, 1904, 1906, and 1907 (10 kg each) (Figure 1 and 2). Although ASTM D5231-92 (2016) specifies a minimum quantity of 91-136 kg for compositional analysis of unprocessed MSW, the collected volume of wastes was limited due to sample storage constraints. To prevent any loss, the samples were placed in thick- labelled plastic bags that were tightly sealed. After being delivered to the laboratory in a sturdy plastic box, the samples were kept in a refrigerator at 4°C until preparation for moisture content measurement. Coning and quartering were used to produce representative samples. Manual sorting was used to remove bulky non-biodegradable components such as metal, plastic, textile, and paper from the obtained MSW samples in order to homogenize the samples for analytical purposes. Coning and quartering yielded approximately 20 kg of representative samples (5 kg per well). These 5 kg samples from each well were further subsampled to roughly 1 kg each using a riffle sample splitter. Following the moisture content test, the samples were oven-dried at 105°C for 3 days to enable the mechanical sieving analysis to divide the samples into selected size fractions.

2.3 Characterization of landfill mining material

2.3.1 Moisture Content (MC)

For the determination of MC, 60 representative subsamples (15 from each well) of 50 g were dried in a drying oven at 105°C for 24 hours, given that the samples were

FIGURE 1: Aerial photograph of the study area marked with a red boundary and sample locations illustrated in purple squares with corresponding well numbers.

well stabilised in terms of organic content and not expected to interfere with the moisture content results. This information is presented in a previous investigation of the same landfill site (Zari et al., 2022).

2.3.2 Particle size distribution (PSD)

For the PSD test, approximately 800 g of representative samples from each well were used. For each well, mechanical sieving was employed for 15 minutes to divide the materials into size fractions. Each sieve's weight and % were recorded.

2.4 Emission Estimation

In this study, a range of each formula component was provided for modelling emission rate variability in order to estimate maximum and minimum emission values. Table 1 (please see the Supplementary material) shows the mathematical expressions used for calculating emission rates for different LFM activities. Equation parameters were modified to account for local meteorological conditions and LFM applications.

The MC and silt content of the recovered waste examined in this study were used to calculate the emission rate. To account for both the best- and worst-case scenarios, wind speed was calculated for a range of 0 to 30 m/s. The required equation components for heavy equipment dimensions and average vehicle speed range were determined using literature from LFM case studies (Jain et al., 2013; Parrodi et al., 2019) and landfill operating reports. Loader sizes range from 1-2.5 m^3 was used for loading and unloading activities (e.g., Caterpillar 320, Hitachi 250LC, and Liebherr 934). The range for unloading activities is also 1-2.5 m^3 if waste is directly unloaded to the screening plant or 22-35 t if waste is delivered to the screening plant using an unloader truck in wide areas. Because direct waste delivery to the screening plant was the method considered in this study, and the capacity of the unloader in the equation was given per t, a range of 0.5-1.5 tonnes was used in the emission rate calculation for unloading activities, which can roughly fill the bucket size of 1-2.5 m^3 used. This is consistent with most published LFM investigations and projects that have employed the direct loading of waste materials into screening plants (Hogland et al., 2014; Jain et al., 2013; LLC, 2009; Lopez et al., 2019; Lucas et al., 2019; Parrodi et al., 2020).

Area units from the current landfill were utilized in the equations whose source type is area (1 phase = 0.0044 km^2). The estimate of the emission rate used average vehicle speeds of 2.6 m/s for haul roads and 11 m/s for transport roads. The silt content of haul and transport road dust range were estimated a little higher than the silt content of waste materials (15-25,10-20%), respectively. The screening plant operational area was estimated as 0.0009 km^2.

FIGURE 2: Photographs of the recovered wastes from drilling (a) well 1901 (b) well 1904 (c) well 1906 (d) well 1907.

For loading and unloading activities, the drop height values employed in the equations ranged from 1-3 m. The frequency of loading values in the equations were approximated using the study's calculated volume of landfill waste.

2.5 LFM Assumed Process Plan

The study area was divided into five phases based on its unit area and the landfill is assumed to be mined on a phased basis as shown in Figure 3. The LFM process plan was carried out on the basis of one phase out of five, which accounts for approximately 0.48 hectares. Area and line source associated activity geometry were approximated using X and Y coordinates over a horizontal convex polygon with four vertices and a straight line joining two vertices, respectively.

The total assumed waste volume range of one phase was obtained by multiplying the length*width*depth of the investigated landfill site by 6 and 7 m, respectively. In addition, an equation was established (1) to be used for the determination of time taken for a LFM project to be com-

FIGURE 3: Assumed plan of LFM phases.

pleted to enable the estimation of emissions cumulative over the lifetime of a LFM operation.

$$T = \frac{\text{Waste volume } (m^3)}{\text{Loader dimension } (m^3) * \text{Total loading frequencies (no./hour)} * \text{Working hours (no.hour/day)}} \quad (1)$$

where T is the timescale of a LFM project. The loading frequencies refer to the total number of repetitions per day in a LFM project. Based on the empirical formulas used in the emission rate calculation and the waste volume range that was estimated, a LFM assumed processes plan flowsheet was developed to visualize the processes and specify volumetric throughput ranges for each unit operation. The road line source is not considered in the LFM assumed processes plan.

3. RESULTS AND DISCUSSION

3.1 Moisture Content

The mean MC results across the 4 wells revealed consistent values. The highest level of MC was relatively represented by well number 1901, reaching 25.9% (Figure 4). Due to the visual appearance during sampling, a higher MC of well no. 1901 was expected.

The average moisture content of all wells in this study was 23.5%, which is consistent with that of Jani et al. (2016) at Högbytorp landfill under comparable conditions. The findings were also comparable to another study by Frank et al. (2017) across seven UK landfill sites and that of Scott et al. (2019) for similar conditions. The landfill site was non-engineered for 13 years after its closure. An engineered Geosynthetic Clay Liner cap was installed in 1998, hence a greater level of infiltration and possibility of contaminant flushing is expected within the landfill site.

3.2 Particle size distribution

One of the key considerations in defining the physical characteristics of landfill waste is the PSD test. (Jani et al., 2016). Analysing this parameter provides a measure of the amount of fine fractions that could be dispersed as dust during LFM activities in conjunction with the MC parameter. The specific waste surface area has an influence on particle size (Dino et al., 2018). The finer a particle is, the higher its specific surface area.

According to earlier research, fine fractions of waste (soil-like material) account for a significant proportion of the overall volume of MSW disposed of in landfills (60%-70%), which is attributable to the application of daily soil cover, construction and demolition waste, and the humification of organic materials (Parrodi et al., 2018; Somani et al., 2018). In addition, there is a noticeable steady decline of waste fractions over time, according to a study by Francois

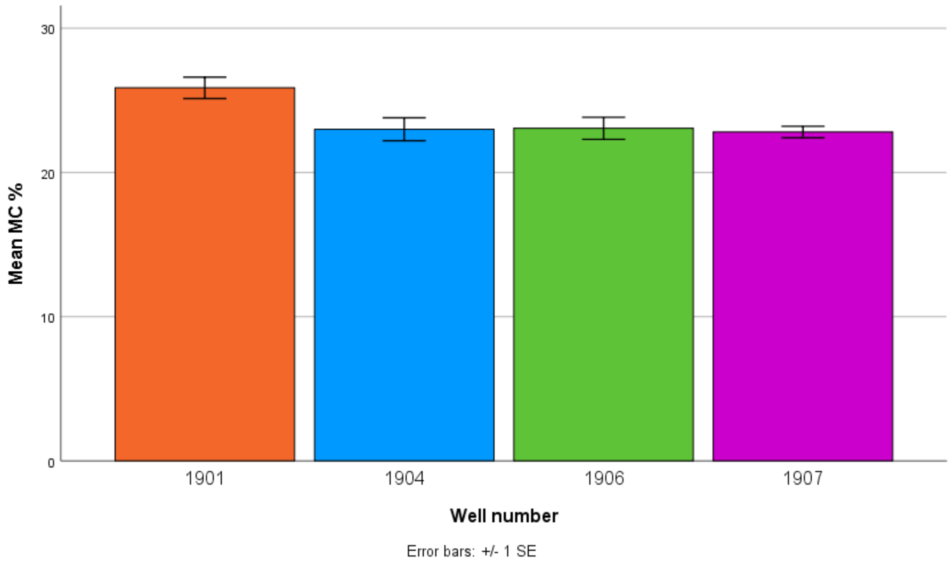

FIGURE 4: Mean moisture content of landfilled waste recovered from the four wells.

et al. (2006) that examined landfilled MSW of various ages (3, 8, 20, and 30 years old) at four different sites (Wang et al., 2021). A study by Singh & Chandel (2022) showed a similar trend of decreasing particle size. As a result, considering the age of the landfill being studied, a significant amount of fine fractions was expected.

Figure 5 shows the results of dry sieving waste fractions of the four wells to the following particle sizes: (>26.5, > 22.4, >19, >16, >12.5, >9.5, >6.7, >4.75, >3.35, >2.36, >1.7, >1.18, >0.85, >0.6, >0.425, >0.3, >0.212, >0.15, >0.106, >0.075, >0.053, >0.038, and ≤0.038 mm). In all wells, PSD revealed similar behaviour. Almost 70% of the total excavated waste fractions from the four pits passed through a sieve with a diameter of 4.75 mm, while 56% of the particles ≤2.3 mm in size. These fractions predominantly consisted of soil-like materials that had soil-like consistency. On average, waste fractions of 0.106 mm account for 11% of the sieve passings within all wells. Within the four wells, well no. 1901 had the greatest proportion of fines (2.5% of the analysed waste fractions < 0.075 mm and ≥ 0.053 mm). Photographs of different size fractions from all wells are shown in Figure 6 (a-d).

Visual observation revealed that soil fraction was notably prevalent within the waste particle size fraction <4 mm. During sample processing and screening, a diverse combination of waste components was detected, as was expected by the heterogeneity and degradation processes associated with landfilled waste. PSD results were comparable to

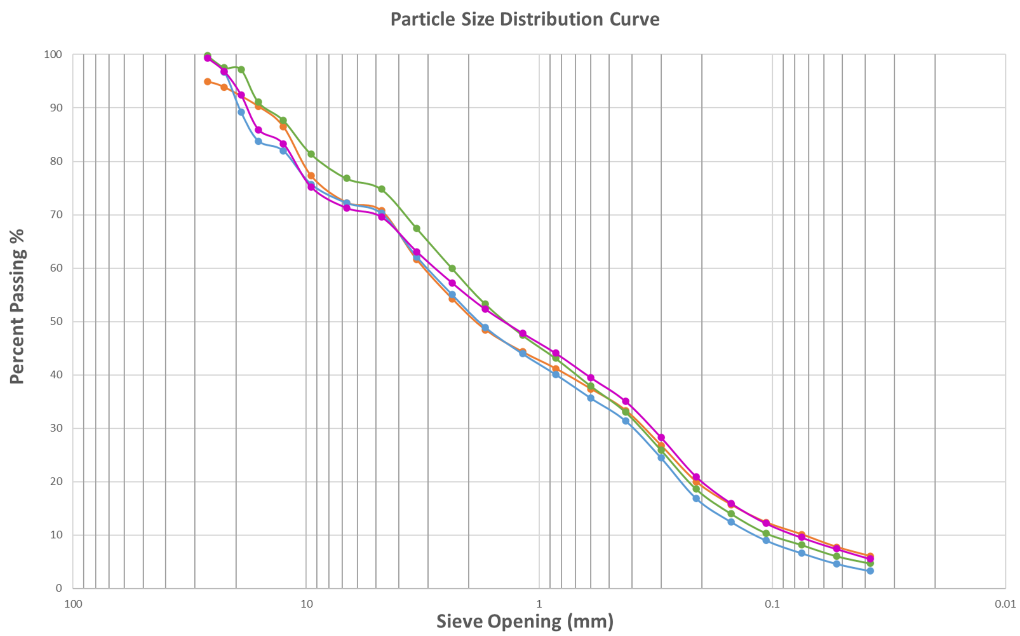

FIGURE 5: Particle size distribution cumulative passing curve for different sieve sizes of all wells.

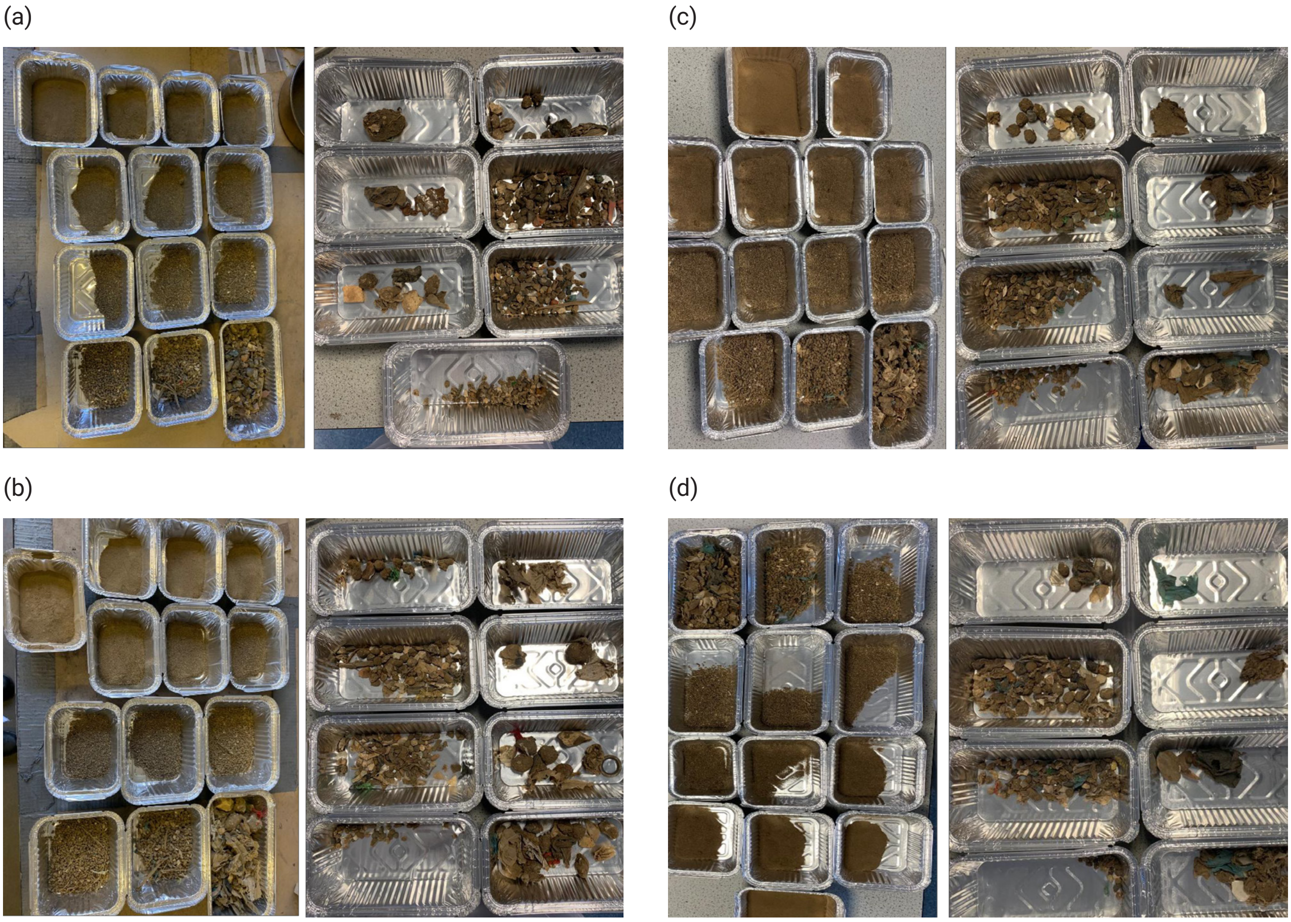

FIGURE 6: Size fractions of wells (a) 1901, (b) 1904, (c) 1906 and (d) 1907.

those of Mönkäre et al. (2016) with 70% of the MSW waste fraction (24-40 years old) having a particle size <5.6 mm. They agreed also with (Masi et al., 2014), with around 65% of the waste fine fractions (30-60 years old) <4 mm. Conversely, (Prechthai et al., 2008), reported that 70% of the MSW fractions of (3-5 years old) were >50 mm, while only 18% were <25 mm. The discrepancy in results among different studies is primarily due to the age of landfilled MSW.

3.3 Emission Estimation

For estimating the suspended emission rate for each LFM activity, nine empirical formulas from surface mining activities were adopted. These activities are: cover removal loading, recovered waste loading, cover removal unloading, recovered waste unloading, haul road, transport road, exposed cover removal waste, exposed pit surface, and screening plant. Table 2 (please see the Supplementary material) displays the calculated range of emission rates, along with descriptive statistics.

The MC and silt content of the recovered waste examined in this study were utilized in the emission rate calculation. MC values varied from 22.8% to 25.8% across all sampling locations. Because fractions between 0.002 and 0.06 mm are defined as silt content (Abril et al., 2016; Neshuku, 2012; Pimolthai & Wagner, 2014) and airborne particular matter (referred to as TSP) (Patra et al., 2016; Ronowijoyo et al., 2020; Sahu et al., 2018), the passing waste size fractions of 0.053 mm sieve obtained from the entire recovered MSW in this study were treated as the silt content range (4.6%-7.7%) in the emission rate calculation. Additionally, it is thought that particles >50 µm cannot penetrate the respiratory tract (Araújo et al., 2014), this range being considered a pollutant criteria in air quality standards (Huertas et al., 2014).

Wind speed, moisture content, and silt content are some of the formula components that have the greatest influence on emission rate (Chakraborty et al., 2002; Chaulya, 2006; Kim et al., 2020; Richardson et al., 2019). This is also valid in this study for all LFM activity sources with the exclusion of wind speed and the inclusion of the drop height parameter as a main driver, especially in the recovered waste loading and unloading activities' emissions.

Point source activities account for the majority of the emission sources in this study, as shown in Figure 7. This finding is consistent with other studies (Chakraborty et al., 2002; Chaulya, 2006). The order of the factors influencing dust emission rate of point source activities was determined as the significant emissions. For cover removal loading, the order of the factors influencing dust emission rate was as follows: moisture content > drop height > silt content > size loader > wind speed > loading frequency.

For recovered waste loading, the order of the factors influencing dust emission rate was as follows: drop height > silt content > size loader > moisture content > wind speed > loading frequency. For cover removal unloading, the order of the factors influencing dust emission rate was as follows: silt content > unloading frequency > drop height > moisture content > capacity of unloader > wind speed. For recovered waste unloading, the order of the factors influencing dust emission rate was as follows: drop height > silt content > moisture content > unloading frequency > capacity of unloader > wind speed.

Overall, the results of the calculated emission rates revealed a higher emission rate with an increase in wind speed, silt content, drop height, size loader/unloader, loading/unloading frequency, and a decrease in moisture content. This is consistent with comparisons of emission rates in previous research (Kim et al., 2020; Richardson et al., 2019; Wang et al., 2022). The rate of increment or decrement of emission rate of all factors is different for each LFM activity source as influencing factor rates differ between LFM activities. They all followed the same trend of increasing or decreasing with emission rate. The overall minimum and maximum emission rate of loading activity sources varied between 0.08 to 1.3 g/s, while unloading activity sources emitted 0.76 to 1.18 g/s, respectively.

Figure 7 demonstrates that the dust emission rate of cover removal loading is significantly higher than recovered waste loading, whilst the inverse is true for the unloading process. This can be explained by the recovered waste unloading being assumed to be unloaded on a screening plant in the calculation of emission rate, while the cover removal unloading was assumed to be unloaded on the ground as a stockpile using lower drop height range as opposed to the recovered waste unloading. The interpretation of the significantly higher emission rate of the cover removal loading over the recovered waste loading might be due to the field measured values of the validation studies used in developing these equations, assuming that the overburden materials contain a higher quantity of fine fractions, thereby producing a higher emission rate, which is comparable with the case of the landfill cover (geosynthetics clay liner) that contained fine materials. The cover removal loading activity source had the highest calculated emission rate of 1.11 g/s. Moisture content was the most influential driver of high and low emission rate for this source. This emphasizes the significance of low moisture content enabling dust formation during LFM activities. Consequently, against high emission rate related activity, dust mitigation or suppression methods such as applying tankers or bowsers to spray water around the mining area should be considered. In order to reduce dust emissions in the LFM area further, a lower drop height and smaller size loader should be considered.

The overall maximum/average/minimum point sources emission rate is 9/ 5.76/ 2.88 kg/hour, respectively. However, line sources maximum/average/minimum emission rates were only responsible for producing 0.18/ 0.072/ 0.036 kg/hour/m, respectively. The total emissions rate of area sources demonstrated a minimal rate of emissions.

Previous studies by Holnicki & Nahorski (2015), Joseph et al. (2018), Ni et al. (2018) and Srivastava et al. (2021), report that atmospheric particle dispersion models are affected by the emission inventory input data. The formulas used in this study were originally developed based on winter season emission inventory to predict the worst-case scenario (maximum concentration) of total suspended particular matter around mining activities in a different region with a larger leasehold mining area (Chaulya, 2006; Chaulya et al., 2019). Consequently, these factors may result in values that are overestimated (Huertas et al., 2012). Additionally, the formulas used for emissions estimation may carry large uncertainties because of variations in site operations, nature of mining, mitigation measures used, and climatic and geological conditions (Chakraborty et al.,

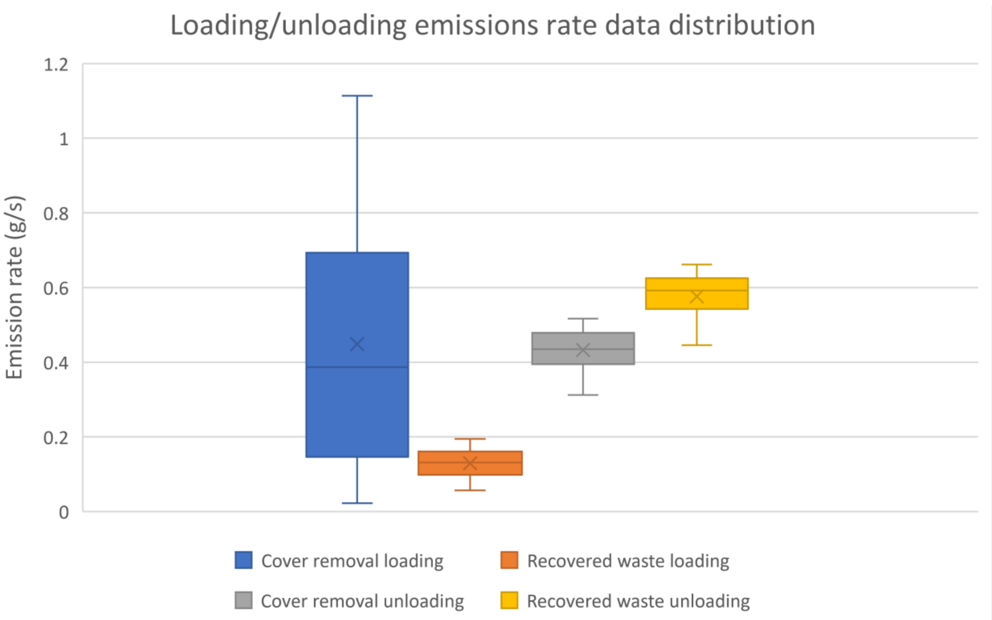

FIGURE 7: Box and whisker plot showing point sources activity emission rates.

2002; Holnicki & Nahorski, 2015; Richardson et al., 2019). This was evidenced in the use of U.S. emission estimation methods to other regions (Triantafyllou et al., 2021). Therefore, to minimise over- or under-estimation of emissions, experimentally based derivation of emission factors pertinent to the region and activities under study is required for a reliable calculation of fugitive dust emissions (Chaulya et al., 2022; Richardson et al., 2019). This can be clearly seen in the screening plant activity results of this study, which is very low, necessitating further research.

3.4 LFM Assumed Process Plan

The cover removal loading is the first unit operation in LFM activities (López et al., 2018; Parrodi et al., 2019; Parrodi et al., 2020). During this process, overlying soils (capping) are removed to expose the waste (Scott et al., 2019). Air pollution is a major concern in surface mining during the removal and transportation of overburden materials (Patra et al., 2016). Similarly, the final cover (capping) of a landfill site might include materials that aid in the removal of heavy metals, thereby increasing the risk of dust exposure and air pollution during LFM projects. Depending on the air velocity, particulate diameter, and drop height, these particles become airborne (Patra et al., 2016). Subsequent mining operation practices have also been identified as substantial causes of airborne particle pollution (Chaulya, 2006; Ghose & Majee, 2000; Onder & Yigit, 2009). The total assumed waste volume for 1 phase out of 5 is approximated as 22,400 m^3 (assuming landfill depth is 6 m) to 26,880 m^3 (assuming landfill depth is 7 m). The LFM project is estimated in this study to take about three months to complete each phase, considering one day of regular working hours (8 hours) and excluding weekends. For the purpose of visualizing the LFM processes and defining volumetric throughput, number of loading frequencies, and temporal ranges for each unit operation, a LFM flowsheet processes plan was produced based on a typical working day of 8 hours. The flowsheet diagram of LFM processes plan is presented in Figure 8 with respective emission rates.

The total number of loading frequencies ranged from 20 to 24 per hour in the flowsheet, based on the estimated volume range of landfilled waste under the study using a size loader of 2 m^3 operating for 8 hours per day. As illustrated in the flowsheet diagram (Figure 8), these numbers were distributed to cover and waste removal loadings and unloadings and adapted to the nature of LFM activities. Because a landfill cover (capping) is typically 1-2 m thick (Hölzle, 2017), the majority of loading numbers were in favour of waste recovery. The diagram provided a timescale for each individual activity dependent on the number of loading frequencies.

According to research by Lopez et al. (2019) and Parrodi et al. (2019), only recovered waste was supposed to be introduced to the screening plant operation that divides the fractions into various sizes. Therefore, it is the scenario considered in this study and thereby the volumetric throughput was in the range of 36-42 m^3/hour. The screening plant heavy machinery intended for use in the proposed plan has a capacity of 49.7 m^3/hour (TR510). Stockpiled exposed cover removal of waste volumetric throughput ranges were assumed to include the reclaimed soil from the screening plant activity, which is estimated to be 100 m^3/day out of the 288-336 range m^3/day. The volumetric throughput range estimates for exposed pit surface were calculated using both cover and waste removal volume per day. Screened waste is assumed to be loaded on to an articulated dump truck for off-site transportation (Parrodi et al., 2019). The screened waste volumetric throughput ranges meant to be transferred off-site were assumed to be 23.5-29.5 m^3/hour since the reclaimed soil from the screening plant was estimated to be 12.5 m^3/hour. Reclaimed soil is assumed to remain on site until completion of a LFM project. Based on articulated dump truck capacity of 13.7 m^3 (Caterpillar 725), the haul road number of loading frequencies per hour was estimated.

The one phase cumulative for the maximum/average/minimum point sources emissions over the lifetime of the LFM operation are approximately 5.04 tonnes (t) / 3.23 (t) / 1.61 (t), respectively. However, the one phase cumulative for the maximum/average/minimum line sources emissions over the lifetime of the LFM operation are approximately 100.8 (kg/m) / 40.32 (kg/m) / 20.16 (kg/m), respectively. These cumulative emissions are based on the assumed waste volume range of 22,400 m^3 to 26,880 m^3.

4. CONCLUSIONS

Predicting particulate matter pollution prior to the initiation of any landfill mining project is now possible with the adoption of surface mining activities using equations to calculate the suspended emission rate for each individual LFM related-activity. An equation was established to be used for the determination of time taken for a LFM project to be completed to enable the estimation of cumulative emissions during the lifetime of a LFM operation. Cumulative maximum, average and minimum emission rates of different sources over the lifetime of the landfill mining operation based on one phase were determined. Results show that the estimated emission rate is higher with an increase in wind speed, silt content, drop height, size loader/unloader, loading/unloading frequency, and a decrease in moisture content for all LFM activities. It was also observed from the emission results that point source activities were the major sources of estimated emissions in this study. The use of fog cannons or water spraying, as well as the method of road sprinkling should be considered to reduce PM emissions. The screening plant activity should be explored in more detail since the results showed a relatively low emission rate.

The one phase out of five cumulative for the maximum, average and minimum point sources emissions over the lifetime of the landfill mining operation calculated in this study are approximately 5.04 t, 3.23 t and 1.61 t, respectively. However, the one phase cumulative for the maximum/average/minimum line sources emissions over the lifetime of the landfill mining operation are approximately 100.8 kg/m, 40.32 kg/m and 20.16 kg/m, respectively.

The formulas used in this study were not validated against LFM activities. Therefore, experimentally based derivation of emission factors reported here should be validated against actual LFM site activities for use in air

FIGURE 8: Flowsheet illustrating LFM processes and defining volumetric throughput, number of loading frequencies and timescales ranges for each unit operation with their respective emission rate.

quality decision-making frameworks that are informed by emission inventories. Equations should be validated using monitoring stations for each LFM activity separately to minimise overlapping of emission rates for the different activities. Validation studies should be conducted within a small scale to limit dust emissions into the atmosphere. They also should be performed without using mitigation measures between the source of emission and monitoring stations, as well as avoiding obstructive objects where monitoring equipment is placed around a LFM area in order to obtain an optimum level of accuracy in comparison with formulas used in this research methodology.

Due to the increase in development activities and projected climate change impacts (rainfall intensity, warmer temperatures), transport of atmospheric particulates is potentially becoming more significant.

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