Gaseous Elemental Mercury Emissions from Informal E-Waste Recycling Facilities in Pakistan

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Abstract

Detrimental effects of mercury (Hg) on ecosystems and human health have been well-documented. Whereas emissions of gaseous elemental mercury (GEM) from e-waste recycling have been reported in developed countries, much less is known about the situation in the Global South. Using a total of 132 passive air samplers, seasonally resolved concentrations of GEM in air were measured continuously at 32 informal e-waste recycling facilities and background location in Pakistan for a period of one year between September 2020 and December 2021. Annual average GEM concentrations at the studied locations ranged from 1.8 to 92 ng m^{-3}. Among the studied cities, higher concentrations were measured in Karachi (mean ± s.d: 17 ± 22 , range: 4.2 - 92 ng m^{-3}), Lahore (16 ± 4.2, 8.2 - 22 ng m^{-3}) and Peshawar (15 ± 17, 4.9 - 80 ng m^{-3}), while lower levels were measured in Hyderabad (6.9 ± 6.2, 3.1 - 25 ng m^{-3}), consistent with a higher rate of informal recycling activities in metropolitan areas. Seasonally, higher GEM levels occurred during autumn (15 ± 16: 3.3 – 92 ng m^{-3}) and summer (13 ± 8.7: 1.8 – 80 ng m^{-3}) than in winter (12 ± 8.4: 2.5 – 49 ng m^{-3}) and spring (9.2 ± 7.3: 1.8 - 80 ng m^{-3}), possibly reflecting enhanced volatilization at higher temperatures and/or varying magnitude of recycling operations in different seasons. Policies and strict regulations related to e-waste management should be developed and implemented urgently in the country.

Keywords: GEM; E-waste; Informal recycling; Passive sampling; Spatio-temporal variations

1. Introduction

With technological advancements and replacement of old electrical and electronic equipment (Forti et al., 2020), there is an increasing number of recycling operations designed to collect precious metals and enable secondary use of valuable resources. Globally, there is widespread export of e-wastes from some developed countries to developing countries, where informal and unregulated handling of e-waste can result in high worker exposure to mercury (Hg) and other hazardous substances (Wilson et al., 2018; Gravel et al., 2020).

Although Hg emissions to the environment have been reduced in some parts of the world, i.e., Europe and North America, it has risen in others, including Africa and Asia (Streets et al., 2019). Worldwide anthropogenic emissions of Hg to the atmosphere have been estimated to amount to 2225 tons per year in 2015 (Bagnato et al., 2015) and have raised Hg concentrations manifold above natural levels. Activities leading to these emissions include mining, extraction of precious metals, coal combustion, chemical production, manufacturing of Hg-added products and informal e-waste processing (Pirrone et al., 2010; Pacyna et al., 2010; Moody et al., 2020; Anselm et al., 2021; Amponsah et al., 2022).

In the environment, Hg occurs in its elemental form (Hg^0), but also oxidized in inorganic (Hg^{2+} - mercuric, Hg^+ - mercurous) and organic (methyl/ethyl-Hg) forms. In the atmosphere, Hg exists in three forms, namely, gaseous elemental mercury (GEM, Hg^0), gaseous oxidized mercury (GOM, mostly Hg^{2+} and Hg^+), and particle-bound mercury (PBM). Its chemical form determines the temporal and spatial dispersion and, ultimately, its transfer to other environmental compartments. GEM may have a long residence time in the atmosphere (up to one year) and can be transported over long distances, dispersing to remote ecosystems. In contrast, GOM and PBM have a shorter atmospheric residence time and are readily deposited locally and regionally to terrestrial and aquatic ecosystems (Pirrone et al., 2010; Driscoll et al., 2013).

It is critical to study the atmospheric emission of GEM due to its wide dispersion and persistent nature in the atmosphere. Passive samplers have been shown to provide reliable long-term average GEM concentrations in air (Jeon et al., 2020; Naccarato et al., 2021). Because they are relatively inexpensive, easy to handle, and need no power supply, these samplers may be deployed simultaneously and in large numbers, enabling measurements of GEM air concentrations at high spatial resolution at and near potential sources (McLagan et al., 2018a). This in turn allows for the identification of GEM sources to the atmosphere (Tao et al., 2017; Streets et al., 2017) and the assessment of GEM emissions (McLagan et al., 2019).

Pakistan is a signatory of the Minamata Convention (signed October 10, 2013) but lack of baseline data and regulatory measures has hampered implementation of Hg reduction efforts (Sattar, 2020; Hina et al., 2021). In recent years, a few monitoring studies have assessed some potential sources of Hg in Pakistan, namely a chlor-alkali plant near Lahore (Jamil et al., 2015), dental clinics (Khwaja and Abbasi, 2014), gold mining sites (Khan et al., 2012; Biber et al., 2014) and a ship-breaking yard in Gadani (Baluchistan Coast) (Kakar et al., 2021). However, the literature on atmospheric monitoring and source identification of GEM in Pakistan is still very limited.

Recently, it has been reported that e-waste and its informal recycling could potentially be an important source of GEM emissions to the environment (Nipen et al., 2022; Snow et al., 2021). Because of its rapidly growing economy, population, and consumption, Pakistan suffers from insufficient systems for handling e-waste (Iqbal et al., 2017; Sajid et al., 2019). There are no listed formal recycling sites in Pakistan and the processes are carried out in informal ways without any guidelines and safety measures (Iqbal et al., 2017; Umair et al., 2016). Therefore, this study was conceived to monitor and assess ambient GEM levels and emissions from informal e-waste recycling facilities in Pakistan. This study provides the spatio-temporal determination of GEM

levels across major cities in Pakistan, developing a baseline dataset of GEM emissions from Pakistan and constraining the contribution of informal e-waste recycling facilities to the GEM pollution load in the country.

2. Materials and Methods

2.1. *E-waste Streaming in Pakistan*

Details of the e-waste stream in Pakistan are presented in a flow chart in Figure 1. The two major sources of e-waste in Pakistan include domestic generation and legal/illegal import of electrical and electronics equipment (Iqbal et al., 2015). As documented by 'The Global E-Waste Monitor 2020', Pakistan produced 433 kt of e-waste in 2019 (Forti et al., 2020). In addition to the e-waste generated locally, it has been estimated that Pakistan is currently importing (often illegally) over 50 kt of e-waste from developed countries each year (Khan, 2022), with Karachi's maritime port as the major entry point (Imran et al., 2017).

Following disposal from these sources, e-waste is collected by scrapers and sellers, who may disassemble the waste into parts, which are then sold to extractors and dismantlers. Dismantlers and recyclers recover precious metals by informal methods such as physical dismantling, open burning, acid baths, and the use of blow torches. To the best of our knowledge, there are no formal e-waste recycling facilities in the country; all e-waste is treated informally (Hameed et al., 2020).

2.2. *Sampling Method*

Details of the Passive Air Samplers (PAS) for the monitoring of GEM levels used in the current study have been described previously (McLagan et al., 2016). Briefly, a stainless-steel mesh cylinder is filled with a sorbent (sulfur-impregnated activated carbon with a high capacity to sorb Hg) and placed into a porous polyethylene diffusive barrier (white Radiello®), which controls the rate of uptake by standardizing the distance GEM is diffusing from the atmosphere to the sorbent.

The Radiello® is housed inside a polypropylene jar protecting it from wind and rain during deployments, but also doubling as a transport and storage container. Further details, including the effect of temperature on the sampling rate, and the PAS's precision and accuracy, have been reported previously (McLagan et al., 2016, 2017a, 2018b, Naccarato et al. 2021).

2.3. *Study Area and Sampling Scheme*

Initially, informal e-waste recycling sites were identified based on: (1) field surveys; (2) national entry points; (3) potential for generation, collection, and markets; (4) e-waste repairing, dismantling and refurbishment sites; and (5) as reported in previous studies (Umair et al., 2016; Iqbal et al., 2015, 2017; Imran et al., 2017; Sajid et al., 2019; Shaikh, 2021). Between September 2020 and December 2021, PAS were deployed for four sequential seasonal deployments (Autumn, Winter, Spring & Summer) at a total of 32 informal e-waste recycling facilities in nine major cities of Pakistan including Rawalpindi, Faisalabad, Lahore, Gujranwala, and Multan in Punjab Province, Karachi and Hyderabad from Sindh Province, Quetta from Baluchistan Province, and Peshawar from Khyber Pakhtunkhwa (KPK) Province (Figure 2). One background site was located on a COMSATS university campus in Islamabad. Details, including site name, site code, geographical coordinates and a detailed description are summarized for each sampling location in Table S1. Details on sampling duration and site-specific information are presented in Table S2. The PAS were deployed at different distances from, but within a radius of ~ 200 m of e-waste recycling sites.

Deployment lengths of PAS ranged from 62 to 135 days and averaged 100 days. Due to Covid-19 travelling restrictions, sampling periods occasionally exceeded the target of 3 months. Samplers were deployed on the rooftop of double story buildings (~10m height). PASs were harvested at the sampling locations by closing the polypropylene jar with a lid, sealing them tightly with

polytetrafluoroethylene tape, placing the sealed samplers in Ziplock bags and a closed container for transfer to COMSATS University Islamabad, where they were stored in a clean room until being shipped to Bursa Technical University in Turkey for analysis.

2.4. *GEM Analysis*

The Hg collected on the sorbent was quantified with a Direct Mercury Analyzer (DMA-1; Milestone Srl, Italy), which is based on the principles of thermal decomposition, Hg amalgamation and atomic adsorption detection. The sorbent in the stainless-steel mesh cylinder was emptied in a pre-weighed Eppendorf tube (2 mL) and then the amount of the sorbent was weighed. A subsample of sorbent was transferred into the quartz cell of the DMA-1 instrument to measure Hg. In the analysis, the sample which is weighed into a quartz vial is dried in the sample cell of the instrument and then thermally decomposed in an oxygen-rich stream. Mercury and other combustion products are released from the sample and passed through a catalyst tube, where all interfering substances are eliminated. The gold amalgamator selectively traps the Hg with other combustion products being flushed from the system. The amalgamation furnace is heated to rapidly release the Hg and carry it into multiple measuring cells positioned along the optical path of the spectrophotometer, and quantified by atomic absorption at 253.65 nm.

2.5 *Calculation of Atmospheric GEM Concentrations*

Volumetric GEM concentrations C (ng m^{-3}) were calculated by dividing the blank-corrected mass of sorbed Hg m (ng) by the product of a sampling rate SR (m^3 day^{-1}) and the deployment time t (day):

$$C = m / (SR\ t) \quad \text{(Eq. 1)}$$

158 Although, relative humidity has no effect on the *SR*, it does increase slightly with temperature
159 (0.001 m^3 day^{-1} or 0.7 % increase in *SR* for every 1 K increase), because temperature affects GEM's
160 molecular diffusivity. For wind speeds over 1 m s^{-1}, the *SR* increases by 0.003 m^3/day^{-1} for an
161 increase by 1 m s^{-1} (McLagan et al., 2017b). Wind speed influences the diffusion path length by
162 controlling the thickness of the boundary layer surrounding the Radiello diffusive barrier (Zhang
163 et al., 2013).
164 The generic *SR* of 0.135 m^3 day^{-1} obtained during a global-scale calibration study of the PAS
165 (McLagan et al., 2018) was therefore adjusted for temperature and wind speed (McLagan et al.,
166 2017a; McLagan et al., 2018) using:

167 $$SR_{adj} = SR + (T - 9.89\,°C) \cdot 0.0009\ \frac{m^3}{day \cdot °C} + \left(WS - 3.41\frac{m}{s}\right) \cdot 0.003\frac{s \cdot m^2}{day} \quad \text{(Eq. 2)}$$

168 where *T* and *WS* are the average temperature (°C) and wind speed (m s^{-1}) during the deployment
169 period of each PAS. The meteorological data, reported for each deployment in Table S2, were
170 taken from the Meteorological Department of Pakistan (PMD). The adjusted *SR* ranged from 0.132
171 to 0.179 m^3 day^{-1}.

172 **2.6.** *Statistical Analysis*
173 Statistical analyses were performed using SPSS Statistics version 22. Analysis of variance
174 (ANOVA) was used to examine differences in GEM mean concentrations between sites and
175 between different seasons at one site. Linear regression models were used for correlations between
176 GEM levels, background concentrations and meteorological parameters.

177 **2.7.** *Quality Assurance and Quality Control (QA/QC)*
178 QA/QC measures include auto calibration of DMA-1 instrument, proper clean-up of quartz sample
179 cells, monitoring of blanks (field and laboratory) and sample replicates. The instrument run in

autocalibration mode automatically when Hg level in each sample was ≥100 ng, otherwise user preferred autocalibration was performed after every 10 samples. Field and laboratory blanks were prepared using sulfur-impregnated activated carbon. Laboratory blanks were prepared by weighing ~0.011 g of AC taken out of the bulk AC from the supplier into the quartz sample cells. Field blanks were the PAS that were exposed to air for approx. 1 min. during the deployment of the samplers in the field, then were sealed properly and were brought back to laboratory. Average Hg level (ng Hg g^{-1} AC) in four field (0.72±0.16; 0.62-0.95) and five laboratory blanks (1.01±0.35; 0.58-1.4) were similar and their average (0.88±0.30) is in the lower part of the range of blank levels (0.38±0.08 to 36±17) reported for previous studies with this sampler (Hoang et al., 2023). Blank correction of samples was performed by subtracting the average concentration in blanks (in ng Hg g^{-1} AC) multiplied with the mass of the sorbent in a sampler (in g AC) from the amount measured in the sorbent from that sampler (in ng Hg). Instrument detection limit (IDL) was 50% of the lowest level of calibration curve (0.001 ng). The method detection limit (MDL) (0.041 ng m^{-3}) was derived as three times the standard deviation of the concentration in the blank and converted to ng m^{-3} applying average of the adjusted SR (0.154 $m^3 day^{-1}$) and an average deployment duration of 100 days.

Quartz sample cells were cleaned prior to analysis by soaking in 5 % nitric acid overnight, then rinsed with deionized water and heated to 550 °C for several hours to remove any traces of Hg. Before analysis of each sample, Hg residue on empty quartz cells was measured by placing the cells in the DMA-1 instrument and applying the same method as during sample analysis. The process was repeated 3 times and the cells were cleaned in 5 % nitric acid overnight if average Hg residue level of triplicate analysis was higher than 50% of average level detected in blanks (0.0045

ng). Two aliquots of each sample were analyzed and duplicate analyses showed differences ranging from 3.5 % to 11 %; the average was used to calculate the Hg amount in the total sample. The DMA-1 instrument has a wide dynamic range and is suitable for a variety of different matrices (DMA-1 Manufacturer's Brochure). We tested the accuracy of the results by i) spiking known amounts of Hg on quartz filters and ii) analyzing NIST CRM 1648a urban particulate matter. For the first approach, we spiked 20 ng (n=4) and 50 ng (n=4) Hg (from a solution with a concentration of 10 ng/µL in 5% nitric acid solution) on ceramic fiber disks (which are used to hold the samples in quartz cells of the instrument). Measured Hg amount in the spiked samples were 20.3±0.83 ng (19.5 ng-21.2 ng, RSD% between spiked amount and measured amount ranges between 1.9% and 6.2%) and 50.9±2.25 ng (48.9 ng-53.1 ng, RSD% ranging 1.5% and 6.3%). The average Hg level measured in three subsamples of NIST CRM 1648a was 1.33±0.078 (1.27 mg/kg-1.42 mg/kg), with RSD% from the certified value of 1.323±0.064 mg kg^{-1} ranging between 1.2%-7.4%. Overall, these RSD% values were judged to be in an acceptable range.

3. Results and Discussion

3.1. *Average GEM Concentrations at Background and E-Waste Sites*

GEM concentrations levels measured during four deployments periods at all studied sites are given in Table S3. At present, due to the lack of national environmental monitoring, there are no data available to assess typical background GEM concentrations in Pakistan. In this study, concentration measured at the background location in Rawalpindi (3.1± 0.81 ng m^{-3}) and ranged seasonally from 1.9 to 3.8 ng m^{-3}. This is about double and triple the global background concentration in the Northern (1.5 - 1.7 ng m^{-3}) and Southern (1.1- 1.3 ng m^{-3}) hemisphere, respectively (Venter et al., 2015; Sprovieri et al., 2016). For example, GEM levels of 1.5 ng m^{-3} have been measured at a remote mountain peak station in Kodaikanal, India (Karthik *et al.,* 2017;

225 Karuppasamy et al., 2020). The concentration level in Rawalpindi ((3.1± 0.81 ng m^{-3}; Table 1) is
226 similar to what has been reported for background sites in Southern China (2.8 ng m^{-3}) (Fu et al.,
227 2010).
228 There could be various reasons for such elevated background concentrations of GEM in Pakistan,
229 including coal production (Ali et al., 2017), coal combustion (Joy and Qureshi, 2023), chlor-alkali
230 plants (Jamil et al., 2015), and gold mining activities (Riaz et al., 2018). Other possible sources
231 could be traffic-related emissions in urban areas (Yue et al., 2021; Cabassi et al., 2022) and
232 atmospheric transport from other regions particularly from nearby countries with high mercury
233 emissions such as India (Lin et al., 2019).
234 The GEM concentration averaged over all 32 e-waste recycling sites and seasons was 12 ng m^{-3}
235 and thus four times higher than the levels recorded at the background site. It is also approximately
236 double what has been reported for coastal/urban air in Chennai, India (4.7 ng m^{-3}) (Karuppasamy
237 et al., 2020), urban air in Mexico City (Morton-Bermea et al., 2021; Schiavo et al., 2022), and the
238 Pearl River Delta in China (Chen et al., 2013) (Table 1). The levels are also comparable or slightly
239 higher than what has been measured in Guiyang, China (Feng et al., 2003). In general, GEM
240 contamination in those settings has been attributed to high industrialization, with coal fired power
241 and cement production plants as key contributors to Hg emission. These greatly elevated GEM
242 levels observed in Pakistan urban areas suggest that e-waste recycling is potentially a major
243 contributor to Hg emissions, even if other possible sources exist. They are supporting earlier
244 studies from Pakistan that identified informal e-waste recycling facilities as a possible source of
245 GEM emission (Iqbal et al., 2015, 2017; Umair et al., 2016; Imran et al., 2017; Sajid et al., 2019).

246 **3.2.** *Spatial Trends*

An inter-city comparison of individual recycling sites is presented in Table S3 and in Figure 3. While serious contamination with GEM is apparent at all sampling sites, the annual average at those sites ranged from 4.4 to 56 ng m^{-3}, i.e., from slightly above background to well over one order of magnitude above background. This variation likely reflects the range of sources and source strengths in different city environments, as well as the location of the sampling sites relative to the GEM generating activity (upwind vs. downwind, distance) (Ding et al., 2007). In particular, the concentration levels are related to the nature and quantity of the recycling processes responsible for emission. Higher concentrations (in units of ng m^{-3}) were measured in samples from Karachi (mean ± s.d, 17 ± 22, range 4.2–92), Lahore (16 ± 4.2, 8.2–22) and Peshawar (15 ± 17, 4.9–80), consistent with the high rate of informal recycling in these metropolitan areas.

With ~17 million residents, Karachi is the city with the highest population in Pakistan, the country's major business center, and South Asia's largest and busiest seaport, annually receiving 85 kt of imported e-waste (89 % of the nation's total) (Imran et al., 2017). In particular, Shershah (J14) is the hub of the recycling industry in Karachi (Hasan, 2002) along with Gulshan-e-Hadeed (J17) (Rafeeq, 2020). High GEM concentrations of 56 ng m^{-3} (39–92 ng m^{-3}) were measured at site J14, considerably higher than at the other sampling sites in Karachi, (4.2–9.1 ng m^{-3}). Similarly, higher levels at Lahore and Peshawar are consistent with extensive e-waste recycling activities and the illegal import of e-waste items along with other secondhand equipment from across the Afghan border (Miankhel et al., 2016; Imran et al., 2017).

Among the studied cities, GEM concentrations in Gujranwala, Rawalpindi, Faisalabad, Multan, Quetta and Hyderabad were lower, but still well above background. This may be due to less and smaller-scale recycling activities in those cities when compared to Karachi, Lahore and Peshawar (Imran et al., 2017), consistent with what has been reported by Shaikh et al. (2020) and Hameed

et al. (2020). According to Ilyas et al. (2017), small and medium-sized steel and metal extraction industries operate in Gujranwala, suggesting that industrial emissions from chrome-plating facilities, metal smelters and informal melting of e-waste for the extraction of precious metals (Faiz et al., 2015) could be active source of emissions in this city.

3.3. *Seasonal Trends*

Pakistan has four well defined seasons, a warm and rainy summer (June to August), a dry autumn (September to November), and a cold and dry winter (December to February) and spring (March to May). Seasonal variations in the GEM levels are presented in Figure 4. Higher concentrations (in units of ng m^{-3}) occurred during autumn (mean 15: range 3.3–92) followed by summer (13: 1.8–80), winter (12: 2.5–50), and spring (9.2: 3.1–39), possibly reflecting the variable magnitude of recycling operations in different seasons. The small variations between seasons suggest minor impact of meteorological conditions (Wan et al., 2009). Generally, higher concentrations were measured during months with lower temperatures and lower ones in summer and post monsoon months. An increase in coal and biomass combustion or reduced vertical atmospheric mixing might contribute to the elevated atmospheric mercury levels during the cold seasons. Also, higher rates of photochemical oxidation may reduce GEM levels in summer. This trend has been observed in several studies, including Schleicher et al. (2015), Kumari and Kulshrestha (2018), Yi et al. (2020), and Yeh et al. (2021).

3.4 *Comparison with Measurements at Other E-Waste Recycling Sites*

Table 1 compares the annual average GEM concentration measured here with those that have previously been reported for various e-waste recycling sites. The annual average GEM level (12 ng m^{-3}) at e-waste recycling sites in this study is lower than what has been reported in Norway (Snow et al., 2021) and Taizhou, China (Tang et al., 2015) although the highest GEM levels we

recorded (at site J1 in Peshawar and site J14 in Karachi) are comparable to the average levels recorded in these studies. A possible reason for higher levels could be placement of samplers closer to the e-waste handling activities (5-20 m) compared to what was feasible in our study (~200 m). Snow et al. (2021) reported GEM concentrations around an e-waste recycling facility in Norway of 2.8 to 3.8 ng m^{-3}, with much higher levels inside the enclosed facility (30 to >1000 ng m^{-3}). Snow et al. (2021) suggested that very high concentrations occurred close to certain types of Hg-containing e-waste materials such as broken compact fluorescent lamp (CFL) bulbs and fluorescent tubes. Nipen et al. (2022) reported GEM concentrations (5.3 ng m^{-3}) close to at an e-waste recycling site in Dar-e-Salaam, Tanzania, that are similar to the levels in this study (e.g., Hyderabad: 6.2 ng m^{-3}).

3.4 *Implications for Human Inhalation Exposure to GEM*

According to the U.S. Agency for Toxic Substances and Disease Registry (USATSDR, 2015), the Minimal Risk Level (MRL) for chronic inhalation exposure to GEM on a daily basis is 200 ng m^{-3}. The United States Environmental Protection Agency (USEPA) provides a reference value for GEM inhalation of 300 ng m^{-3} (Palma et al., 1999), whereas the World Health Organization (WHO, 2000) and the International Programme on Chemical Safety (Fisher & WHO, 2003) suggest values ranging from 100 to 200 ng m^{-3}. Japan's Ministry of the Environment suggested occupational GEM inhalation limits of 40 ng m^{-3} (MOE, 2003). The values measured here are consistently lower than these thresholds, except for a handful of seasonal deployments at J14 and J1 that exceeded the very strict Japanese limit. We nevertheless believe that our measurements raise serious concerns related to human inhalation exposure to GEM.

First, because our measurements represent average concentrations over periods of approx. three months, we can expect that much higher concentrations prevailed during shorter time periods.

More importantly, we believe that the GEM levels in inhaled air could be manifold higher than the reported average annual levels (12 ng m^{-3}) because our sampling sites were generally ~200 m removed from the actual recycling operations. In particular, we should expect very steep spatial concentration gradients between the site of the actual recycling operations and the sampling sites. For example, Snow et al. (2021) observed significantly higher levels of GEM inside a Norwegian e-waste recycling facility, with concentrations ranging from 31 to 1140 ng m^{-3}, compared to levels of 2.8 to 3.8 ng m^{-3} at a distance of only 100 m from the facility. Similarly, Monaci et al. (2022) reported long term average GEM concentrations near processing facilities of an abandoned Hg-mine that ranged over more than two orders of magnitude (from 17 to 4,200 ng m^{-3}) over a distance of a few 100 metres. Moreover, in Snow et al. (2021)'s study in the Norwegian e-waste recycling facility and two artisanal gold mining communities in Ghana, which involved both stationary sampling locations and personal wearable samplers, the latter had consistently higher GEM concentrations, often by as much as an order of magnitude.

In summary, by measuring annual average concentrations as high as 56 ng m^{-3} at a fair distance from actual recycling activities, we project personal inhalation exposures that will regularly and consistently exceed MRLs for chronic inhalation exposure to GEM. Such exposures may occur not only for those directly involved in the recycling activities, but also to community members in the vicinity to these activities, including children.

4. *Conclusions*

A total of thirty-two informal e-waste recycling facilities across nine major cities of Pakistan were selected for the monitoring of GEM concentrations using a passive air sampling technique. The results indicated that uncontrolled and large-scale informal e-waste recycling practices were associated with high GEM concentrations in metropolitan areas. These levels exceeded those

found at a national background site and are well above hemispheric background levels, suggesting that emissions from e-waste handling and recycling operations were contributing to these elevated levels. The levels remained high throughout the year, indicating that the emission sources were stable. Practices such as dismantling, open burning, acid bath, and refurbishment of e-waste without safety measures were commonly observed at the studied sites. Furthermore, these facilities lacked proper ventilation and were often located in urbanized areas, posing health hazards related to inhalation exposure not only for the those involved in the e-waste recycling operations but also to the larger community. It is important for Pakistan, as a signatory of the Mina-Mata Convention, to develop an e-waste management plan that will help reduce emissions and exposure levels. Additional investigations are needed to better identify and characterize the e-waste handling activities that result in the release of GEM, to determine the spatial concentration variability within the informal e-waste processing sites, to quantify the human GEM inhalation exposure and to investigate the potential health effect it may cause.

Supplementary Information. Supplementary information (SI) contains details on descritption of sampling sites (Table S1), sampling periods, meteorological conditions and site sampling rates (Table S2) and basic descriptive statistics of spatio-temporal concentrations level (Table S3)

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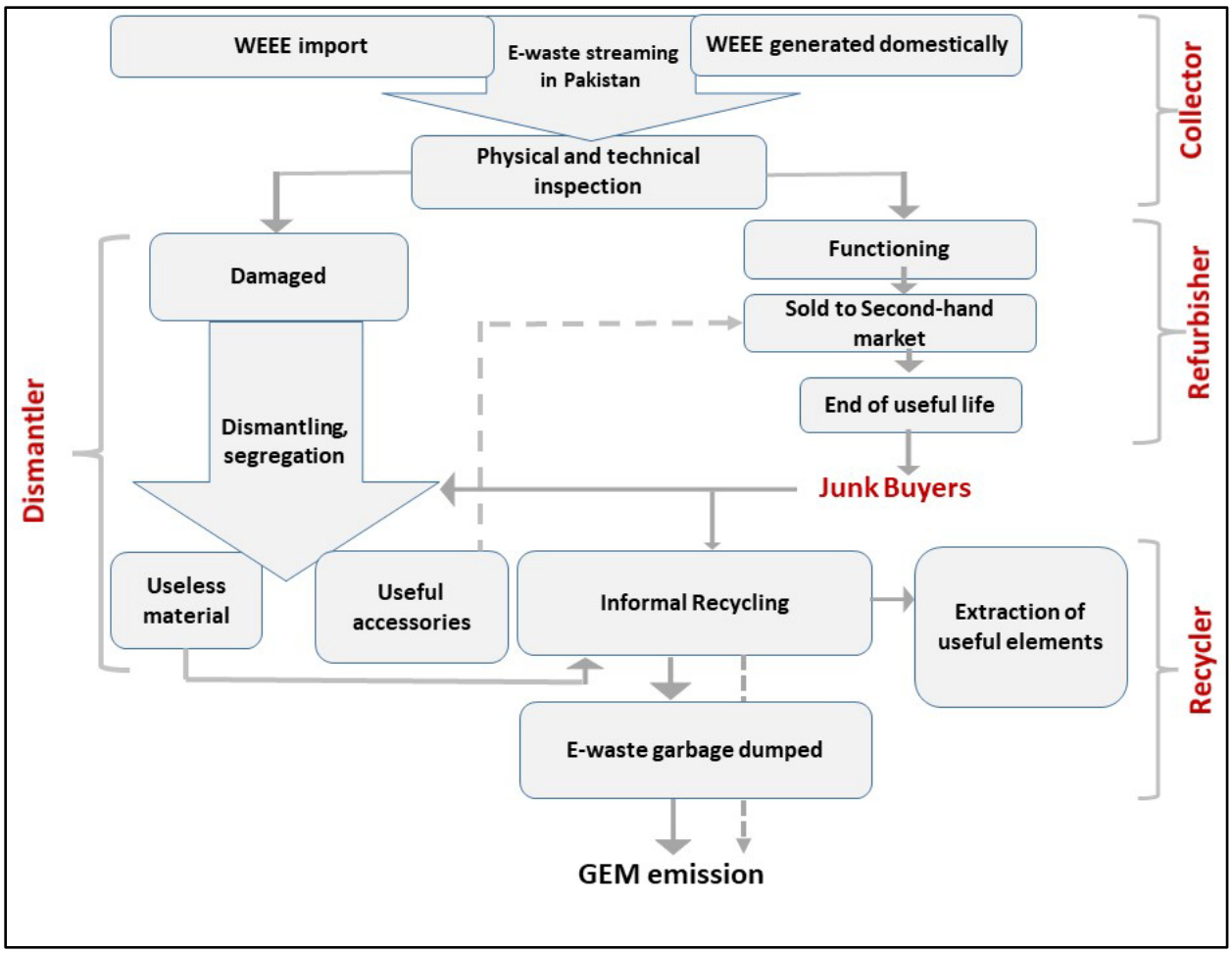

Figure 1: Flowchart showing the e-waste streaming in Pakistan

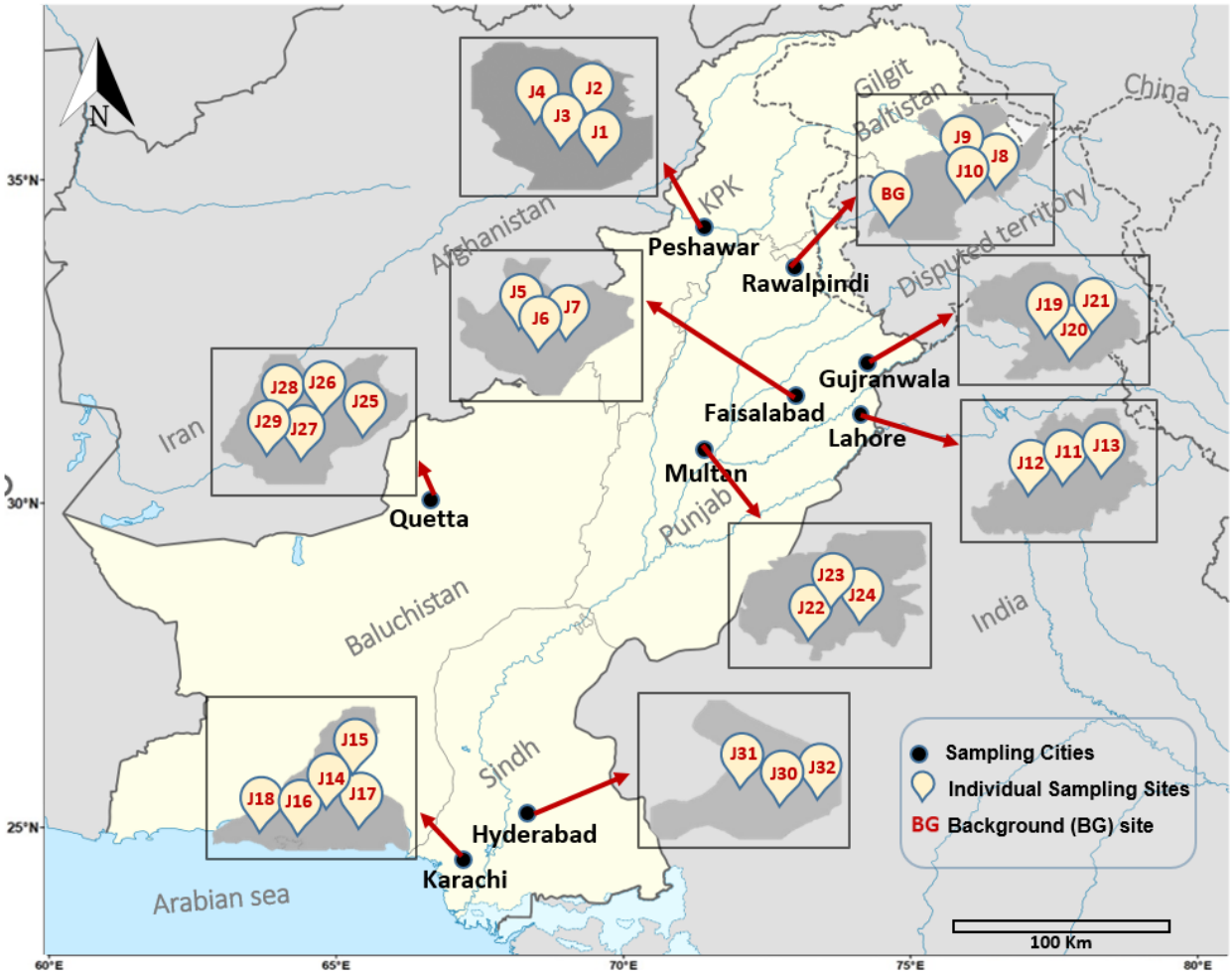

Figure 2: Individual e-waste recycling sites across major cities in Pakistan

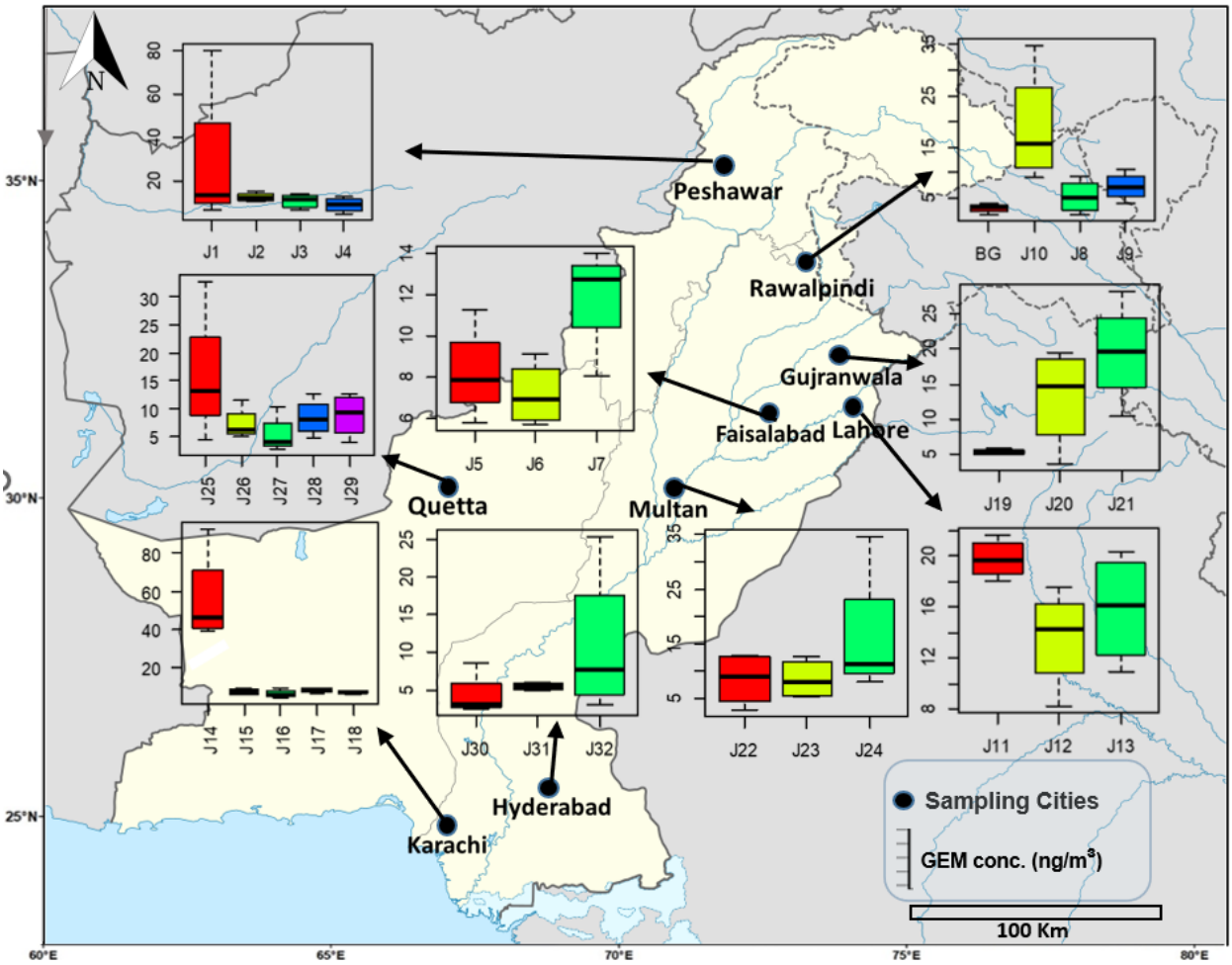

Figure 3: Yearly mean concentrations of GEM (ng m^{-3}) at individual e-waste recycling sites in Pakistan

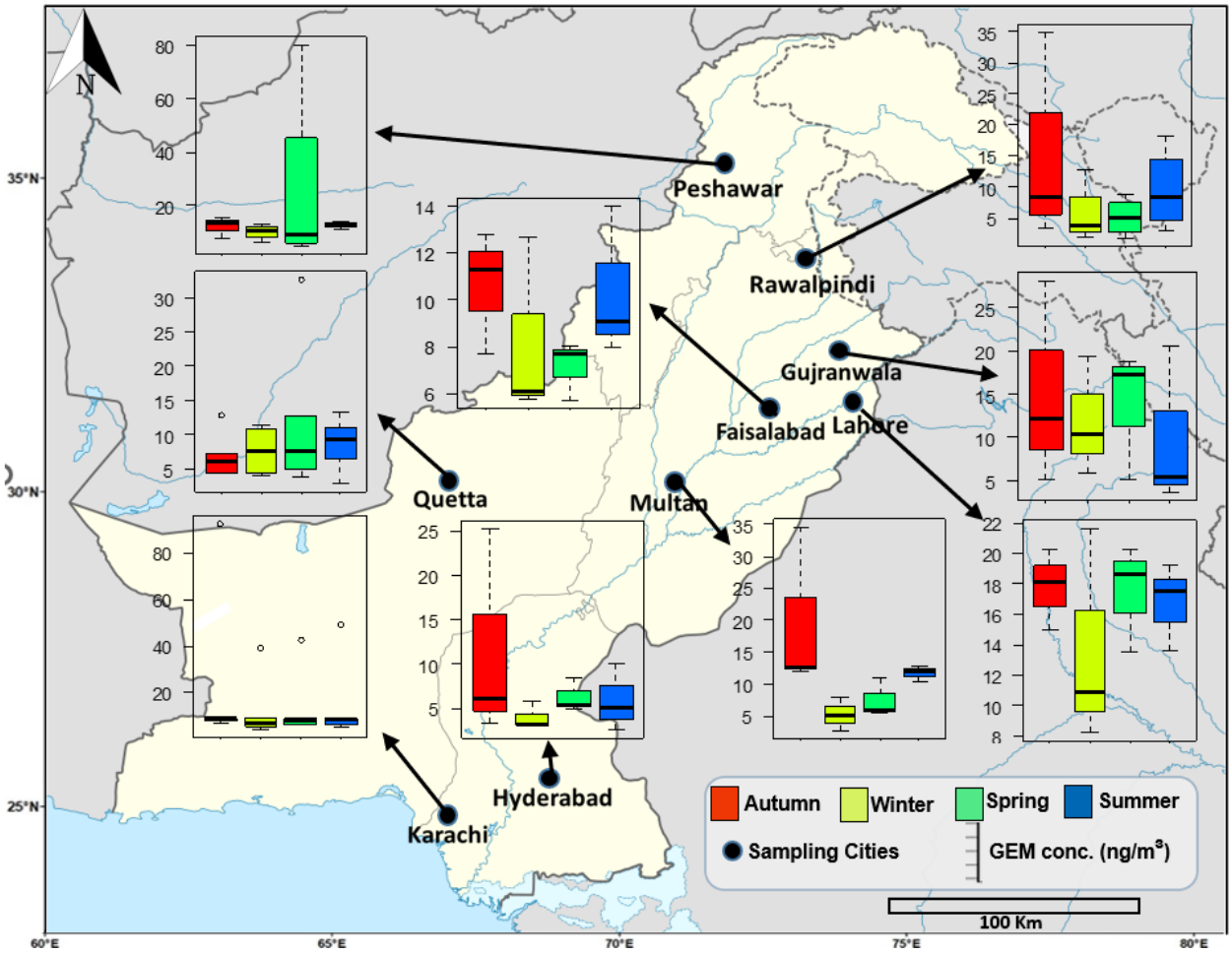

Fig 4: Seasonal GEM concentrations (ng m^{-3}) at informal e-waste recycling sites

1 **Table 1: Comparison of average GEM concentrations (ng m^{-3}) with previous studies across the worldwide**

Location	Country	Sample site	Monitoring period	GEM (Mean ± SD) Range values	Reference
Regional Background Sites					
Rawalpindi	Pakistan	Background	Oct-2020 to Sept-2021	2.99±0.71 (1.94-3.84)	This study
Summit of Mt. Leigong	South China	Ambient Air	May-2008 to May 2009	2.80±1.51 1.88 to 3.59	Fu *et al.,* (2010)
Popocatépetl	Mexico	Volcanic/rural	Mar-2019	1.72 ± 0.83 0.51-5.5	Schiavo *et al.,* (2020)
Kodaikanal*	India	Rural	Nov-2012 to Sept-2013	1.53±0.25 0.83 to 3.25	Karthik *et al.,* (2017)
		Ambient air	Jan-2015 to Dec-2016	1.53 1.38-159	Karuppasamy *et al.,* (2020)
Southern Hemisphere		Background	2016	1.1-1.3 (range)	Sprovieri *et al.,* (2016)
Northern hemisphere		Background	2015	1.5-1.7 (range)	Venter *et al.,* (2015)
E-waste Recycling Sites					
Peshawar, Faisalabad, Lahore, Rawalpindi, Karachi, Multan, Gujranwala, Quetta & Hyderabad	Pakistan	informal facilities	Oct-2020 to Sept-2021	11.88±2.12 (1.78-92.07)	This study
Dar-es-Salaam city	Tanzania	informal sites	Feb-2019 to April 2019	2.13±1.57 0.79-5.34	Nipen *et al.,* 2022
E-waste recycling facility	Norway	formal recycling facility	Dec- 2018	5 Geomean 0.9 – 1140	Snow *et al.,* 2021
Taizhou*	China	informal recycling sites	2015	30.7 ± 9.9 16.7 to 43.4	Tang *et al.,* (2015)
Urban and Impacted Sites					
Mexico City	Mexico	Urban	Mar-2021 to Apr-2021	5.60 ± 2.33 0.20-30.23	Schiavo *et al.*, (2022)
Hefei	China	Urban	Mar-2016 to May-2016	2.53 ± 1.28 0.32-15.10	Yue *et al.,* (2021)
Mexico City	Mexico	Urban	May-2019,	3.80 ± 1.34 0.50-11.90	Morton-Bermea *et al.,* (2021)

			May-2020	5.04 ± 2.95 0.50-32.60	
Chennai*	India	Ambient air	Jan-2015 to Dec-2016	4.68 3.62-5.40	Karuppasamy *et al.,* 2020
Zhongshan*	China	Urban, fluorescent lamp manufacturing facilities	Jul-2019 to Aug-2019	2.4 ± 3.5 0.37–49	Luo *et al.,* (2021)
Abbadia San Salvatore	Italy	abandoned Hg-mining area	Jun-2016 to Jul-2016	1030±1420 17 to 4200	Monaci *et al*., (2022)
Mexico's Pacific coast, Presidente Plutarco Elías Calles (CETEPEC)	Mexico	Coal-fired power plant	2013	2.8 0.3-14	Garcia *et al.,* (2017)
Pearl River Delta	China	Mt. Dinghu Guangzhou	Nov-2010 to Oct-2011	5.07 ± 2.89 1.87 to 29.9 4.60 ± 1.36 2.7 to 11	Chen *et al.,* (2013)
Guiyang*	China	Coal Fired Power Plant and Guizhou Cement Production Plant	Apr-2000 to Nov-2001	7.39 1.7 to 147	Feng *et al.,* (2003)

* **Total gaseous mercury (TGM)**