Informal E -Waste Recycling in Nine Cities of Pakistan Reveals Significant Impacts on Local Air and Soil Quality and Associated Health Risks

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Abstract

The global increase in electronic waste (e-waste) has led to a rise in informal recycling, emitting hazardous heavy metals (HMs) that threaten human health and ecosystems. This study presents the first comprehensive assessment of HM levels in dry deposition and soils at proximity of forty (40) informal e-waste recycling sites across Pakistan, between September 2020 to December 2021. Findings reveal that Zn (1410), Pb (410) and Mn (231) exhibited the higher mean deposition fluxes (μg/m^2.day), derived from air samples, particularly in Karachi. Similarly, soils showed higher mean concentrations (μg/g dw) of Mn (477), Cu (514) and Pb (172) in Faisalabad, Lahore, and Karachi, respectively. HMs concentrations were found higher in winter or autumn and lower in summer. In addition, HM levels were significantly (p=0.05) higher at recycling sites compared to background sites year-round, highlighting the e-waste recycling operations as the major source of their emissions. The I_{geo} index indicated moderate to extremely contaminated levels of Cu, Pb, Cd, and Ni in Karachi, Lahore and Gujranwala. Ingestion was found as a leading human exposure route, followed by dermal and inhalation exposure, with Pb posing the greatest health risk. The Cumulative Incremental Lifetime Cancer Risk (ILCR) model suggested moderate to low cancer risks for workers. Strategic interventions recommend mitigating health and environmental risks, prioritizing human health and ecosystem integrity in Pakistan's e-waste management.

Keywords: E-waste; Informal recycling; HMs; Particulate matter; Soils; Health risks

1. Introduction

Due to rapid technological advancements electronic and electrical equipment (EEE) production and use has experienced exponential growth in the last two decades which in turn has led to a rise in electronic and electrical waste (e-waste) (Kumar et al., 2017; Murthy et al., 2019). E-waste denotes to the end-of the lifespan of electrical and electronic products such as, televisions mobile phones, computers with plastic as major constituent and contain heavy metals (HMs) like Manganese (Mn), Cadmium (Cd), Copper (Cu), Lead (Pb), Chromium (Cr), Zinc (Zn), Nickel (Ni) as well as other toxic compounds e.g., polychlorinated biphenyls (PCBs) and polybrominated diphenyl ethers (PBDEs) (Brindhadevi et al., 2023; Qin et al., 2022).

As per recent data (2019) the global production of electronic waste reached 53.6 million metric tons (mt), equating to 7.3 kilograms per capita annually, up from 5.8 kilograms per capita in 2014 (Houessionon et al., 2021; Forti et al., 2020). E-waste has emerged as a significant waste stream worldwide, with its generation rate increasing by 1 to 5% annually (Kumar and Fulekar, 2019). Typically, developed countries either dispose of their e-waste in landfills or export it to developing nations under the title of second-hand item or recycling (Dutta et al., 2022; Arya et al., 2021). E-waste recycling facilities have been reported in many countries of the world, like China, India, Pakistan, the Philippines, Turkey, and Vietnam (Hashmi et al., 2022). It is estimated that developing nations receive approximately 80% of the world's e-waste (Sthiannopkao & Wong, 2013), and up to 2% of the Asian population relies on e-waste recycling for their livelihood (Imran et al., 2017). In developing countries, cheap labor force, weak legislation, and a substantial market for recovered materials make them a target and vulnerable destination for the informal recycling and dumping of e-waste (Singh et al., 2018). Informal recycling practices include acid baths for

71 the recovery of gold and other valuable metals, stripping/shredding and open-air burning, grilling,
72 chipping, and melting plastics, disposing of unusable material in open fields and water bodies.
73 Through these activities, several hazardous elements such as Pb, Cd, Ni, Cu, and other metals can
74 be emitted into the surroundings. Thus, exposure of these toxic elements to humans and their
75 effects are of great concern (Song and Li, 2014). For example, exposure to Pb can affect the liver,
76 kidney, and nervous system and impair cognitive development (Obeng-Gyasi, 2018; Bellinger,
77 2011). Cr can cause respiratory irritation, kidney, and liver damage, weakened immune systems,
78 and cancer of the nose, sinus, or lung (Tchounwou et al., 2012). Ni may contribute to dermatitis
79 and bronchial asthma (Kuntawee et al., 2020).
80 Pakistan has been documented as one of the major e-waste importers; yet, there is no quantification
81 of the e-waste inventory flows (Iqbal et al., 2017). Pakistan also generates a considerable amount
82 of e-waste with nearly 433kt of e-waste was produced in the year 2019 as compared to 301kt in
83 2016, showing a 43.8% rise and attaining it the 20^{th} biggest e-waste producer (Forti et al., 2020).
84 In Pakistan, e-waste primarily originates from two main sources: legal/illegal import of electrical
85 and electronic equipment and domestic generation (Iqbal et al., 2015). Once discarded, sellers and
86 scrappers collect and disassemble the waste into various parts. Dismantlers and extractors make
87 their way by treating such waste using different informal methods. Informal techniques such as
88 open burning, acid baths, physical dismantling and treatment using blow torches are employed by
89 these dismantlers and recyclers to recover valuable metals. Notably, Pakistan lacks formal e-waste
90 recycling facilities and entire e-waste is processed through informal channels (Hameed et al.,
91 2020). Driven by profits, this extensive informal and illegal recycling of e-waste operates at the
92 stake of human and environmental health.

Previously, several studies highlighted the close correlation between human exposure and e-waste recycling (Arya et al., 2021; Mowla et al., 2021; Perkins et al., 2014; Pradhan and Kumar, 2014; Shakil et al., 2023). Numerous studies have reported HMs in various environmental matrices in Pakistan (Saleem et al., 2018; Khan et al., 2015; Mahmood and Malik, 2014), however only one study reported soil contamination from e-waste dumping and recycling sites in Lahore (Shakil et al., 2023). As per the reported literature, this is the first comprehensive research in Pakistan that provides the temporal and/or seasonal trends of HMs in ambient air and soils, and their exposure risks covering an extensive sampling campaign across Pakistan.

2. Materials and methods

2.1. *Study Area Description and Sampling Scheme*

Informal e-waste recycling sites were determined through the following methods: (1) conducting field surveys to locate sites involved in e-waste repair, dismantling, and refurbishment; (2) identifying national entry points; and (3) referencing previous studies (Shaikh, 2021; Sajid et al., 2019; Imran et al., 2017; Iqbal et al., 2017, 2015; Umair et al., 2016). A total of 40 e-waste recycling facilities through nine major urbanized cities in Pakistan were selected for sampling. These cities include Rawalpindi, Faisalabad, Lahore, Gujranwala, and Multan (Punjab); Karachi and Hyderabad (Sindh); Quetta in Baluchistan; and Peshawar in Khyber Pakhtunkhwa (KPK) province. Additionally, the COMSATS University Campus in Islamabad was selected as the background site (refer to Figure 1). Further details, such as site names, codes, geographical coordinates, and descriptions for each sampling location, are summarized in Table S1.

Overall, 160 Passive air deposition samplers (PASs-DD) and 160 soil samples were collected for assessing the atmospheric load and soil residues of heavy metals in four seasons (autumn, winter, spring, and summer) between September 2020 and December 2021. Details about the deployment

of PASs at each sampling location are given in Table S2, while meteorological conditions are presented in Table S3. To collect ambient particulate deposition in air, PASs-DD were deployed at varied distances from e-waste recycling sites but all within a radius of ~ 200 m. Deployment time of PASs-DD varied between 62 to 135 days with an average of 100-days. Samplers were deployed on the rooftops of buildings at ~10 m height, to minimize the influence of re-suspended soil dust which can occur near ground level.

2.2. *Sampling*

2.2.1. *Soil samples collection*

A hand-held corer was used to collect soil samples between 0-10 cm depth. At least 10 cores were collected randomly at individual sampling locations and a composite sample was obtained after mixing all-together. Soil samples collected were sieved through a 2 mm mesh metal sieve. Moisture and organic content in soil samples were determined according to ASTM D-2974-87 (ASTM, 2000) method and soil pH measurements were made according to EPA method 9045-D (US EPA, 2004). The detail of physicochemical analysis is presented in supporting Information (Table S4).

2.2.2. *Atmospheric Particle-bound Heavy Metals*

Sampling of air deposition, which for heavy metals is mainly associated with particulate matter, was performed by deploying passive dry deposition (PASs-DD) collectors using a polyurethane foam disk (PUF-disk). PASs-DD have successfully been used to determine the flux of polycyclic aromatic hydrocarbons (PAHs) (Eng et al., 2014) and HMs (Gaga et al., 2019) in urban environments and collect both particle-phase dry deposition and gas-phase. The information about design and sampling rate of -PAS can be found elsewhere (Gaga et al., 2019; Eng et al., 2014). To the best of our knowledge, this is the first research to employ PASs-DD to map the atmospheric

fluxes of HMs in and around Pakistani e-waste recycling facilities. PUF disks were precleaned prior to deployment in accordance with the standard clean-up procedure applied for persistent organic pollutants (POPs) analysis (Birgul and Karakus, 2024). After harvesting at the sampling locations, PUF disks were tightly closed in aluminum bags, then placed within zip-lock bags and a locked container for transport to COMSATS University Islamabad. At the university, they were stored in a dry, clean room until they could be shipped to Bursa Technical University in Turkey for sample preparation. Prepared samples were analyzed for HMs in Dokuz Eylul University.

2.2.3 *Sample Preparation*

Weight of each PUF disk was recorded before and after deployment in the field, hence approx. dust amount collected was determined (0.04 g-4.39 g). Subsamples taken out of each PUF disks that were deployed for approx. 3 months at the sampling sites were used to analyze heavy metals. A stainless-steel corer was used to cut 1 cm diameter cores from 5 randomly selected points on each disk to obtain sub-samples. The PUF disk was weighed again after 5 cores ($\sum$0.11 g) were taken out to determine the weight of the cores as well as weight of dust collected in these 5 cores (0.02 g-0.22 g). Wet digestion method was applied to prepare the samples for instrumental analysis. The subsample consisting of 5 cores were placed in a 40 mL glass vial, 2 mL of H_2SO_4 and 6 mL of nitric acid HNO_3 and were added into the vial. Vials containing PUF disk subsamples and acid mixture were placed on an aluminum heating block and digestion was carried out by heating samples at temperatures ranging from 160 °C to 180 °C until the formed brownish fume disappeared and the solution became clear. After the digestion was completed, the samples were kept in the fume hood until they are cooled to room temperature and diluted to 50 mL using ultrapure water. To remove any impurities an aliquot of 15 mL of the digested sample was filtered through a 0.45 μm pore size Teflon syringe filter. Filtered samples were placed in 15 mL volume

plastic falcon tubes, the caps of the tubes closed tightly and refrigerated at -18 °C until instrumental analysis.

2.2.4. *Instrumental Analysis*

Analysis of heavy metals was conducted using an Inductively Coupled Plasma-Mass Spectrometer (ICP-MS) (Agilent 7700x). The operational parameters of the device were as follows: RF power set to 1550 W; carrier gas flow rate at 0.90 L/min; plasma gas flow rate at 15 L/min; plasma sampling depth of 8 mm; nebulizer flow rate set at 1.01 L/min; extractor lens potential at -160 V; conical spray chamber temperature maintained at 2°C; nebulizer pump operated at 0.10 cycles/second; nebulizer type used was micro-mist; and the ion lenses model employed was x-lens. Readings were taken as three replicates and average value of three readings were used as the concentration value of a sample.

2.3. *Quality Assurance / Quality Control (QA / QC)*

Accurate quality assurance and quality control practices were pursued from sample collection through to analysis. All laboratory equipment was either high quality polypropylene or Teflon. All chemicals were analytical grade. Field blanks were brought to the laboratory in closed boxes/bags after being exposed to ambient air for 1-2 minutes. A mixture of acids which were used for sample digestion was used in preparation of laboratory blank samples. A total of 12 blank samples for PUFs and a total of 14 blank samples for soils were prepared and they were handled in the equal manner as the samples. Reproducibility of the obtained results were checked by analyzing CRM 540. Relative standard deviation of results between certified value and analysis ranged between 1.38% (Mn) and 14.4% (Pb) with an average RSD of 6.31±4.41%. Further details on RSD values detected for target contaminants are given in Supporting Information Table S5. For digestion, method spike samples (n=10) were prepared by adding known quantity of target elements (100

ppb each) into acid mixture and process was carried out similar manner as done for samples. The average recovery ratio was 96.4±4.89% ranging between 89.7% (Co) and 103% (As and Pb). The Instrument Detection Limit (IDL) was based on half the concentration value of the lowest calibration level that the instrument was able to determine in the set of calibration solutions. Samples were not blank corrected. Therefore, method detection limit (MDL) was calculated based on average blank concentration + 3 x Standard deviation (SD) of the concentration detected in the blank samples. IDL and MDL values of each targeted element are given in Supporting Information Table S6.

2.4. *Calculation of Particulate Phase Pollutant Fluxes*

Fluxes were presented as mass per unit area per unit time (1) (i.e., μg/m^2.day) in Eq. 1 as follows:

$$Deposition\ Flux\ (F) = \frac{m}{t} \times A \quad (1)$$

Where m is the quantity of HMs determined in the PAS-DD sample (μg), t is the total deployment duration of the sampler in the field (days), A is the surface area where deposition of particles occurred (0.00785 m^2, as edges and bottom of the DD-PAS sampler sampling medium holder unit is relatively closed, therefore it was assumed that deposition occurred mainly on to the top side of PUF disk).

2.5. *Geo-accumulation Index (*I_{geo})

The contamination level in soils from this study were determined by calculating the values of the geo-accumulation index (I_{geo}) (also known as Muller index) (Han et al., 2018; Muller, 1981). This indicator is used to calculate the scale of contamination by assessing the relation between calculated concentration level and background level of the contaminant (Muller, 1969). Eq. (2) is used to calculate I_{geo}:

$$I_{geo} = \mathrm{Log2} \times \frac{C}{1.5BG} \quad (2)$$

Here C represents the heavy metal concentrations in the soil samples analyzed while BG is the geo-chemical background concentration of the element in the earth's crust and 1.5 is background matrix correction factor as consequence of lithogenic effects. This coefficient depicts any anthropogenic effect in the computation as well as the influence of geological and depositional features.

2.6. *Enrichment Factor (EF)*

Assessing the contamination level of heavy metals from human activities involves comparing the concentrations of heavy metals in soil and particulate samples with those of reference elements found in the Earth's crust. For this purpose, enrichment factor (EF) is calculated as shown in the Equation (3) below (Al-Khashman, 2013; Abdulaziz et al., 2022; Tepe et al., 2022) for soil and particulate matter, separately.

$$EF = \frac{(\frac{C_i}{C_{ref}})_{sample}}{(\frac{C_i}{C_{ref}})_{earth\ crust}} \quad (3)$$

Where Ci is the concentrations of target heavy metals (µg/g) in particulate matter or soil and Earth crust. C_{ref} represents reference elements concentration in particulate matter/soil and Earth crust. As Mn is one of the reference elements used in previous studies (Rahman et al., 2021; Pasha et al., 2015; Zajusk-Zubek et al., 2015; Fabretti et al., 2009; Sakata and Asakura, 2011; Abdulaziz et al., 2022), it is used as reference element in the current study. Tayler (1964) provides Heavy metals concentrations in Earth's crust. An EF level of ≤10 indicates the cause of the metal from the natural source of Earth's crust while EF value of >10 suggests anthropogenically enriched (Duan et al., 2021; Kodat et al., 2023)

2.7. *Human health risk assessment*

Exposure of HMs to the human body can occur through ingestion via mouth, inhalation via mouth and nose, and dermal exposures via skin when in proximity of informal e-waste recycling. The present study calculates the non-carcinogenic health and lifetime cancer risk based on inhalation, ingestion, and dermal exposure routes of HMs. In addition, overall data used for the calculation of average daily intake is given in Table S7.

2.7.1 *Estimation of Daily Intake Through Contaminated Soil*

The potential risk from heavy metals in contaminated soils collected from informal e-waste recycling sites are calculated based on recommendations proposed by United States Environmental Protection Agency (USEPA 1989; 1997; 2000 and 2001). The average daily intake (ADI) (mg/kg-day) of each heavy metal through soil ingestion, inhalation and dermal contact pathways was calculated using the following equations (4-5) (Ajani et al., 2022):

$$ADI_{Soil-Ing} = \frac{C_{soil}\ x\ IngR_{soil}\ x\ EF\ x\ ED}{BW\ x\ AT\ x\ 10^{6}} \quad (4)$$

$$ADI_{Soil-Inh} = \frac{C_{soil}\ x\ InhR\ x\ EF\ x\ ED}{BW\ x\ AT\ x\ PEF} \quad (5)$$

$$ADI_{Soil-Dermal} = \frac{C_{soil}\ x\ SA\ x\ FE\ x\ AF\ x\ ABS\ x\ EF\ x\ ED}{BW\ x\ AT\ x\ 10^{6}} \quad (6)$$

Where $ADI_{Soil\text{-}Ing}$, $ADI_{Soil\text{-}Inh}$ and $ADI_{Soil\text{-}Dermal}$ are the average daily intake doses through soil ingestion, inhalation, and dermal absorption, respectively (mg/kg/day) and C_{soil} is the concentration of heavy metal in soil.

2.7.2 *Estimation of daily Intake through air particulate matter*

Human exposure is also measured in terms of average daily intake (ADI) via ingestion (Eq. 7) and dermal absorption (Eq. 8) (mg/kg/day) of air particulate matter and exposure concentration via inhalation (EC) of air particulate matter (Eq.9) (Abdulaziz et al., 2022).

$$ADI_{PM-Ing} = \frac{C_{PM} \; x \; IngR_{PM} \; x \; EF \; x \; ED}{BW \; x \; AT \; x \; 10^6} \quad (7)$$

$$EC_{PM-Inh} = \frac{C_{PM} \; x \; ET \; x \; EF \; x \; ED}{AT_n} \quad (8)$$

$$ADI_{PM-Dermal} = \frac{C_{PM} \; x \; SA \; x \; AF \; x \; ABF \; x \; EF \; x \; ED}{BW \; x \; AT \; x \; 10^6} \quad (9)$$

Where $ADI_{PM\text{-}Ing}$, and $ADI_{PM\text{-}Dermal}$ are the average daily intake doses through particulate matter ingestion and dermal absorption, respectively (mg/kg/day) and $EC_{PM\text{-}Inh}$ is the exposure concentration via inhalation (μg/m^3) of air particulate matter. C_{PM} is the concentration of heavy metal (mg/kg for $ADI_{PM\text{-}Ing}$, and $ADI_{PM\text{-}Dermal}$; mass per unit volume (μg/m^3) for $EC_{PM\text{-}Inh}$).

2.7.3 *Non-Carcinogenic Health Risk*

Non-carcinogenic adverse health effects assessment is carried out for both carcinogenic and non-carcinogenic heavy metals. The IARC has classified As, Cr, Cd and Ni as Group 1 carcinogens, whereas Pb and Co were classified as Group 2A carcinogens (IARC, 2024). USEPA (1989; 1997; 2000 and 2001) proposed that target hazard quotients (HQ) and hazard index (HI) characterize the potential health risk. HQ is a ratio of determined average daily intake (ADI, (mg/kg/day)) to reference dose (RfD, (mg/kg/day)) of an individual element. HQ values ≤1 indicate no significant or acceptable risk, while HQ values >1 indicate the potential for adverse health effects (USEPA, 2001). For a given heavy metal, HQ values for exposure through soil ingestion, inhalation, and dermal contact in addition to particulate matter ingestion and dermal contact is calculated using the equations given below (Ajani et al., 2022; Abdulaziz et al., 2022).

$$HQ_{Soil-Ing} = \frac{ADI_{Soil-Ing}}{RFD} \quad (10)$$

$$HQ_{Soil-Inh} = \frac{ADI_{Soil-Inh}}{RFD} \quad (11)$$

$$HQ_{Soil-Dermal} = \frac{ADI_{Soil-Dermal}}{RFD} \quad (12)$$

$$HQ_{PM-Ing} = \frac{ADI_{PM-Ing}}{RFD} \quad (13)$$

$$HQ_{PM-Dermal} = \frac{ADI_{PM-Dermal}}{RFD} \quad (14)$$

Whereas RfD is the reference dose (mg/kg/day) (Table S8).

HQ value for exposure through particulate matter inhalation can be calculated based on the equation given below (Abdulaziz et al., 2022)

$$HQ_{PM-Inh} = \frac{ADI_{PM-Inh}}{RfCx1000} \quad (15)$$

Whereas RfC is the reference concentration of the heavy metal (mg/m^3) (Table S8)

Health risks associated with exposure to multiple metals is estimated by using Hazard index (HI) (the summation of hazard quotients (HQ_k) of individual metal "k") which can be calculated using the following equation (USEPA 2001; Khan et al., 2020)

$$HI = \sum HQ_k \quad (16)$$

For non-carcinogenic or carcinogenic heavy metals, a value of HI > 1 represents that there is a chance of occurrence of non-carcinogenic effects, while the exposed individual is unlikely to experience obvious adverse health effects when HI < 1.

2.7.4 *Lifetime Cancer Risk*

The probability of developing cancer because of human exposure to carcinogenic heavy metals (As, Cr, Cd and Ni as Group 1 carcinogens and Pb and Co as Group 2A carcinogens (IARC, 2024)). Cancer risk over the lifetime (ILCR) can be estimated using equations below for ingestion, inhalation, and dermal contact respectively.

$$ILCR_{\sum ing} = (ADI_{Soil-Ing} + ADI_{PM-Ing})\ x\ SF \quad (17)$$

$$ILCR_{\sum inh} = (ADI_{Soil-Inh}\ x\ SF) + (EC_{PM-Inh}\ x\ IUR) \quad (18)$$

$$ILCR_{\sum dermal} = (ADI_{soil-dermal} + ADI_{PM-dermal})x\ SF \quad (19)$$

Where $ILCR_{\Sigma ing}$, $ILCR_{\Sigma inh}$, $ILCR_{\Sigma dermal}$ represents incremental lifetime cancer risks via soil+particulate matter ingestion, inhalation, and dermal contact, respectively. Values of Slope factor (SF, mg/kg.day) and inhalation unit risk (IUR, μg/m^3) for carcinogenic metals are given in Table S8. Classification of ILCR is as follows: ILCR≤ 1x10^{-6} (very low); 10^{-6}≤ILCR≤10^{-4} (low); 10^{-4}≤ILCR≤10^{-3} (moderate); 10^{-3}≤ILCR<10^{-1} (high) and ILCR≥ 10^{-1} (very high) (Zhang et al., 2021). The Cumulative ILCR for a given carcinogenic metal can be calculated as the sum of ILCR values occurred due to ingestion, inhalation, and dermal contact (Sun et al., 2021) and this value should be maintained below 10^{-4} (Chalvatzaki et al., 2019).

$$Cumulative\ ILCR = ILCR_{\Sigma ing} + ILCR_{\Sigma inh} + ILCR_{\Sigma dermal} \quad (20)$$

2.8. *Statistical analysis*

IBM SPSS Statistics software (version 20.0) was used for statistical analysis. One-way repeated measures multivariate analysis of variance (RMANOVA) was applied to determine significance of differences in HMs concentration over four seasons in each city. Arc-GIS software (version 10.2.2) was used for site identifications at all sampling sites across nine cities.

3. Result and Discussion

3.1. *Concentrations levels of Heavy metals at e-waste sites*

Mean level of HMs in air and soil as well as background site in four seasons are detailed in Table 1. At background site in Islamabad, the average deposition flux of HMs in air during four deployment seasons was noted as 41.8 $\pm$ 33.8 μg/m^2.day. Whereas the average deposition flux was found to be 161 $\pm$ 111 μg/m^2.day ranging from 56.0 μg/m^2.day (Rawalpindi) to 331 μg/m^2.day (Karachi). Substantially, higher deposition flux at study sites (> 3 times the background site, Table 1) presumably due to the presence of active sources of HMs at studied sites. Deposition flux of HMs in air was found in following sequence Zn > Mn > Pb > Cu > Ni > Cr > Cd. Since, the present study is first of its kind from Pakistan in which passive samplers were deployed to study HMs and report concentration in the units of flux (μg/m^2.day), therefore the comparison of deposition flux of HMs in air was not viable with regional or global studies using different methodologies. The compositional trend of HMs in present study were accorded well with those of other e-waste recycling sites in previous studies with higher concentrations of Pb, Zn and Mn than Cr, Ni, Cu and Cd (Table S9). The elevated levels of HMs at study sites compared to the background site indicate the contribution of emissions from e-waste recycling sites to the local atmosphere.

For soil samples, the mean concentrations from recycling facilities were much greater (~1.5 to 13 times) than the background concentration suggesting the influence of extensive e-waste recycling operations (Table 1). The mean concentrations of HMs in soils were found in the following sequence Mn>Zn>Cu>Pb>Cr>Ni>Cd which is almost identical to those reported for previous studies at e-waste recycling, dumping and/or dismantling sites (Table S10). This shows that e-waste dismantling and recycling activities substantially contribute towards the contamination of soil. The mean concentration and their ranges for Mn, Cr and Ni were comparable to those noted

in Nigeria (Isimekhai et al., 2017), Ghana (Teye et al., 2023), and China (Han et al., 2019), whereas for Cu, Zn, Cd and Pb, the concentrations were much lower than reported by those studies (Table S10). Soil pollution is often assessed either by comparing total metal concentrations with standard guideline values or by classifying using pollution indices (Wu et al., 2018). In this study, it is apparent that the mean concentrations of Zn and Cd in soils from e-waste facilities exceeded the safe regulatory limits of WHO i.e., 50 µg/g (Osobamiro et al., 2019) and 0.003 µg/g (Ahmad et al., 2021), respectively. Out of nine selected cities, mean concentrations of Ni, Cu and Pb in four major industrial cities *i.e.,* Karachi, Lahore, Faisalabad and Gujranwala were higher than WHO standard limits of 35 µg/g (El-Naggar et al., 2021), 100 µg/g and 50 µg/g (Teye and Tetteh, 2023), respectively. The soil concentration of Cr in all cities was found under the standard limit of WHO *i.e.,* 100 µg/g (Teye and Tetteh, 2023) except for Gujranwala city where the concentration (112 µg/g) just exceeded the standard limit. For Mn, except in Lahore and Faisalabad, its mean levels were within the recommended limits of WHO (437 µg/g) (Bawwab et al., 2022).

Heavy metal's deposition fluxes reported in the current study were generally align with the patterns observed in the earlier research conducted from various parts of the world, reinforcing the global nature of the issue. For instance, similar to findings in India (Ha et al., 2009), Nigeria (Isimekhai et al., 2017), China (Han et al., 2019 and Ghana (Teye et al., 2023), Pakistani cities exhibit elevated levels of Zn, Mn, Pb, Cu, Ni and Cr as these metals are often associated with electronic waste (Purchase et al., 2020). Zn is used in die-castings, batteries fluorescent lights and X-ray screens in EEE while Mn as an alloy with Pb, Al or Cu is a major constituent of batteries, sensors, and super capacitors. Similarly, Al, Pb Cr, and Cu are major components of printed circuit boards, smart card chips, electrical wiring, and various other EEE. During recycling process (mostly informal) including dismantling, repairing, burning and acids treatment to recover precious metals may

possibly initiate heavy metal contamination at the e-waste recycling facilities (Li et al., 2011). In addition to the overall concentration of the metals, the extent of contamination is determined by the fraction of their movable and bioavailable forms, which in general controlled by the organic matter, pH and other properties in soil (Tang et al., 2010).

3.2. *Spatial trends*

An intra-city relationship of HMs in the air samples near e-waste facilities is illustrated in Figure 2 and S1 while descriptive statistics is given in Table S11.. Karachi, Gujranwala, Lahore, and Faisalabad were found to be the leading cities with higher level of HMs. Out of studied HMs, four metals were found higher in Karachi (μg/m^2.day) i.e., Cr (with annual mean 38.4), Mn (231), Zn (1410) and Pb (410) whereas higher fluxes of Ni (157) and Cu (255) were noted for Gujranwala. In Karachi, the maximum fluxes (μg/m^2.day) of Cr (131), Mn (1520), Ni (276), Cu (931), Zn (8105), Cd (23.4), and Pb (2993) were observed at Sher shah (J18) or its adjacent Lyari area (J20). Sher Shah market is the biggest junkyard of Pakistan for used electronic and locomotive parts storage, dismantling, and recycling reported in earlier studies (Hameed et al., 2020; Rafeeq et al., 2021). Recently, Kazim et al. (2023) also reported the higher levels of gaseous elemental mercury at this site which they had associated with e-waste dismantling and recycling processes in Pakistan. Among the sampling cities, HMs concentrations in Rawalpindi, Multan, Quetta, and Peshawar were lower but still well above background HM levels. High variability of Zn, Cu, Pb, Mn and Ni in major cities in comparison to background concentrations (Table 1) may also be due to different sources in addition to contribution from e-waste recycling sites. These sources could also be attributed to the industrial emissions and vehicular traffic (Zhou et al., 2014) as most of the studied sites are densely populated and industrial hubs of some scale. Vehicles emit HMs into the atmosphere mainly via exhaust (fossil fuel emissions) and non-exhaust emissions which include

wearing and tearing of different vehicular sections like tires, brake pads, and corrosion of metallic parts. Therefore, USEPA highlights 21 hazardous elements that can mostly be appointed to road traffic (Gupta, 2020); and the five dominant HMs (Zn, Cu, Pb, Mn and Ni) as observed in our study are among them.

Descriptive statistics for the HMs concentrations found in samples collected from soil at contaminated sites in nine cities are summarized in Table S12 and their variations are shown in Figure 3, while spatial distribution maps are given at Figure S2. Higher mean concentrations (µg/g dw) of Cr (112), Ni (79), Cu (457) and Cd (1.32) were found in Gujranwala. Among four selected study sites in Gujranwala, the industrial zone site (J26 Site) had major contribution in elevating the mean concentration of HMs. Maximum concentrations of Cr, Ni, Cu, and Cd at this site were observed up to 716, 542, 2565 and 4.53 (in µg/g dw) respectively in different seasons which were nearly 1 ~ 2 orders of magnitude higher than the background site (Table 1). The city of Gujranwala's industrial zone is a center for buying/selling of e-waste with informal recycling of printed circuit boards (PCBs) by using include acid baths, open burning etc. Highest mean concentrations of Mn (477 µg/g), Cu (514 µg/g) and Pb (172 µg/g) in soil were detected in Faisalabad, Lahore, and Karachi. The dominant site in Faisalabad, which had particularly contributed to mean concentrations of HMs was Motor Market (J9) where the concentration of Mn (1415 µg/g dw), Cr (142 µg/g dw), Zn (1014 µg/g dw) and Pb (221 µg/g dw) were higher than other three selected sites within the city. In Lahore and Karachi, the elevated inter-city concentrations of mostly HMs were observed in soil samples collected from Misri Shah (J15) and Sher Shah (J18) respectively. The description of all these sites is summarized in Table S1.

3.3. *Seasonal trends*

Previous studies have reported substantial influence of meteorological conditions (temperature, wind speed, rainfall) on levels of air pollutants (Nasir et al., 2019) and physicochemical characteristics of soils (Aydın et al., 2023; Isimekhai et al., 2017). Pakistan enjoys four seasons i.e., dry autumn between September and November, dry and cold winter during December and February, spring from March to May, warm and rainy summer which generally lasts from June to August. Since the present study was carried out for one year covering all four seasons, seasonal comparisons of concentrations of studied HMs (Cr, Mn, Ni, Cu, Zn, Cd, Pb) in air and soil was undertaken for each city. Seasonal mean deposition fluxes of HMs in air are presented in Table S13 and their seasonal variations are illustrated in Figure 4a. On average, higher mean fluxes of all HMs in air were recorded either in Winter or Autumn whereas lowest fluxes were observed during the summer except for Zn,. Mean deposition fluxes (μg/m^2.day) of Cu (122), Cd (2.4), and Pb (159) were observed in Autumn whereas those for Cr (25.3), Mn (167) and Ni (67.8) were found during winter. For Zn, higher mean deposition fluxes were inversely found during the spring season. As presented in Table S13, this rise in fluxes was only contributed by elevated levels of Zn in Karachi and Lahore. This might be due to the contribution by some additional sources at sites and/or extraordinary dismantling / recycling.

Average seasonal concentrations of selected HMs in soil at sampling sites are given in Table S14. Similarly, higher concentrations of HMs in soil were associated with the dry season *i.e.,* autumn followed by winter, spring, and summer. Mean concentrations (in μg/g dw) of Cr, Mn, Ni, Cu, Cd, Pb were recorded as 59.6, 415.2, 47.8, 188.2, 0.8, 118.5 respectively except for Zn for which the mean elevated levels were observed in spring likewise levels in air. In contrast to variability pattern as observed for air levels, high seasonal variability was observed for most of the HMs (Figure 4b). The spatial spread of HMs could be attributed to magnitude of e-waste dismantling / recycling

activities in addition to variable rainfall patterns in different cities, surface runoffs, human activities across the sites and soil characteristics (Isimekhai et al., 2017). The range of HMs concentrations at e-waste processing sites depends on nature of activities. Isimekhai et al., 2017 had associated clustering of Cd, Cu, Cr, Pb and Zn with recycling activities, whereas the presence of Ni and Mn indicates dismantling activities. Nevertheless, no significant difference ($p > 0.05$) in HMs concentrations were found in different seasons despite different meteorological conditions which shows that sampling sites are hot spots of HMs throughout the year. While positive correlation ($p<0.05$) of most of the heavy metals in soil and in Particulate matter suggest common source of contamination (Table S15).

3.4. *Geo-accumulation index (I_{geo})*

Geo-accumulation index (I_{geo}) was determined based on the comparison between concentration level of heavy metals at e-waste recycling facilities with the background site (Islamabad). The calculated I_{geo} values for given recycling sites at sampling cities have been presented in supplementary information (Table S16) and assessed with the given criteria for determining the scale of contamination. The I_{geo} values for each heavy metal was interpreted as follows: ≤ 0 (uncontaminated); $0 - \leq 1$ (uncontaminated – moderately contaminated); $1 - \leq 2$ (moderately contaminated); $2 - \leq 3$ (moderately – heavily contaminated); $3 - \leq 4$ (heavily contaminated); $4 - \leq 5$ (heavily to extremely contaminated) and $5 <$ (extremely contaminated).

Among sampling cities, Lahore, Gujranwala, Karachi, and Faisalabad were the most contaminated ones based on soil residues of the heavy metals. More specifically, Lahore and Gujranwala were extremely contaminated by Cu, while heavily to extremely contaminated by Pb and Cd, respectively. Similarly, Karachi and Faisalabad were marked for moderately to extremely contaminated for all target heavy metals especially Cu, Pb, Ni and Zn. Rawalpindi and Multan

have high Cu contamination levels, whereas these sites were moderately to heavily contaminated with Zn, Cd, and Pb. Interestingly, Peshawar and Hyderabad showed moderate contamination of all metals except Cd in Hyderabad. However, the I_{geo} levels for studied sites are elevated while comparing I_{geo} levels of some other regional e-waste recycling locations of the world, e.g., informal e-waste recycling shops in Dhaka, Bangladesh (Mowla et al., 2021) and inside the dumping area of e-waste recycling facility at Korle Lagoon, Ghana for Ni, Pb and Cu (Fosu-Mensah et al., 2017). The present study found moderate to extreme levels of contamination of soil around e-waste recycling locations (primarily in Pakistan's megacities) by most e-waste oriented heavy metals, raising concerns about their possible exposure to workers and the surrounding environment.

3.5. *Enrichment Factor (EF)*

Table S17 presents the contamination factor results of HMs in soil and particulate samples of 40 e-waste recycling facilities throughout the country. The mean EF value of Pb>Zn>Cd>Cu>Ni>Cr>Mn was 12.7>8.27>7.83>7.16>1.29>1.28>1.00 in soil, while Zn>Cd>Pb>Cu>Ni>Cr>Mn was 78.9>78.5>64.6>15.2>4.89>1.49>1.00 for particulate samples respectively. Among cities, higher EF values were calculated for Gujranwala, Lahore, Karachi, Peshawar, and Quetta, being the most populated and industrialized cities depicting higher contamination levels. In both sampling matrices, EF values indicate elevated contamination between e-waste recycling facilities for most of the metals studied except for Cr whereas Ni shows a considerable contamination level. Pb, Cd, Zn and Cu levels were found to be elevated in comparison with levels reported from e-waste recycling sites in India (Pradhan & Kumar, 2014). Similarly, higher levels of Cu, Pb, Cd and Zn and low levels of Mn and Ni were calculated from another study conducted in India (Arya et al., 2021). The contamination levels at e-waste recycling

sites ranged from substantial or moderate contamination to extremely high contamination in Pakistan's megacities.

3.6. *Human health risk assessment*

In soil $ADI_{soil\text{-}ing}$ was the main exposure pathway to the workers in proximity of e-waste recycling sites in all sampling cities (Table S18). Higher $ADI_{soil\text{-}ing}$ was calculated for Zn, Mn, Pb and Cu, while $ADI_{soil\text{-}inh}$ was the least exposure pathway in all sampling cities. Non-CRs exposure of HQ through different exposure routes suggests that $HQ_{soil\text{-}der}$ was the major route followed by $HQ_{soil\text{-}ing}$ and $HQ_{soil\text{-}inh}$ in all sampling cities (Table S20). All elements did not pose any threat with value of (HI < 1) for workers residing near e-waste recycling sites. Our results trends were similar to the previous studies on non-CRs health risk assessment (Dutta et al., 2022; Han et al., 2018; Singh et al., 2018). $HQ_{soil\text{-}der}$ was observed to be the main exposure route of heavy metals with high values of HI were calculated for Gujranwala, Lahore, Karachi, and Faisalabad with a value of 3.27×10^{-1}, 2.63×10^{-1}, 2.43×10^{-1} and 2.22×10^{-1}, respectively.

Estimation model of daily intake of particulate matter through different route suggest that inhalation exposure (ECPM-inh) was the major pathway for all studied metals in all sampling cities followed by ingestion, while dermal exposure have least observed values (Table S19). High inhalation exposure ($\mu g/m^3$) was observed for Zn, Pb and Cu for Karachi (4.21×10^{-1}, 1.22×10^{-1}, 5.34×10^{-2}), Lahore (4.16×10^{-1}, 2.51×10^{-2}, 2.64×10^{-2}) and Gujranwala (3.62×10^{-1}, 5.64×10^{-2}, 7.62×10^{-2}), respectively. Non-CRs model provide evidence that the value of $HQ_{PM\text{-}inh}$ was >1 for Ni at Gujranwala (3.35) and Hyderabad (1.20), Mn at Karachi (1.38) and Faisalabad (~1) (Table S21) indicate the potential for adverse health effects (USEPA, 2001). While HI values of $HQ_{PM\text{-}inh}$ was >1 for Gujranwala (4.63), Karachi (3.37), Hyderabad (2.21), Faisalabad (1.76), Lahore (1.54) Multan (1.25) and Quetta (1.17) suggest the chance of occurrence of non-CRs effects to the

workers and public living near e-waste recycling facilities in these cities. Previously, Aziz et al. (2022) also reported high level heavy metal (particulate) exposure via inhalation route among ingestion and dermal at Makkah city in Saudia Arabia. To summarize, non-CRs assessments indicate that dermal exposure is the major route of exposure to the contaminated soil, while inhalation for particulate matter. Karachi, Gujranwala, Lahore, Faisalabad were the most contaminated cities while Pb, Zn, Cu, Cd are key elements (exposure and contamination) in soil and particulate matter to the workers and general population residing near e-waste recycling sites in Pakistan.

3.7. *Lifetime Cancer Risk*

Among all investigated heavy metals, IARC-2024 (The International Agency for Research on Cancer) has categorized Ni Cr, and Cd as group-1 carcinogens while Pb lies in group-2A carcinogens. Cancer risk over lifetime (ILCR) of Cr, Ni, Cd and Pb via ingestion, inhalation and dermal contact were determined and shown in Table 2. Moderate to very low ILCR was observed for metals through different exposure routes in all cities ($ILCR_{\Sigma inh}$: 9.76×10^{-8} at Peshawar to $ILCR_{\Sigma der}$:1.02×10^{-4} at Lahore for Pb). $ILCR_{\Sigma inh}$ and $ILCR_{\Sigma Der}$ were most common exposure pathways for Cr being the major contributor of ΣILCR in all e-waste recycling sites. The accumulative ILCR ranged: high from Multan (1.08×10^{-4}) and low at Peshawar (9.53×10^{-5}) for inhalation pathways. Overall, cumulative ILCR model suggest that inhalation and dermal contact are main exposure route depicting moderate to low CRs for workers at e-waste recycling facilities in Pakistan. The children living near and adults working in the proximity of these e-waste recycling sites may encounter acute as well as chronic health effects due to continuous exposures to HMs (Wu et al., 2019). Moreover, local population living nearby these uncontrolled informal e-waste recycling practices (through secondary exposure) in the studied cities especially Karachi, Lahore

512 and Faisalabad could potentially suffer health related problems by e.g., liver, and vascular system
513 disorders, chronic kidney damage, irritation of upper respiratory tract due to chronic HMs exposure
514 (Grant et al., 2013). In conclusion, soil and particulate matter contamination from e-waste
515 recycling operations poses a potentially alarming risk of cancer and other health issues in Pakistan.

516 **4.** *Conclusions*

517 In the last decade, there has been a rise in e-waste recycling activities in Pakistan to recover
518 valuable metals by open burning, dismantling without safety measures, treatment with acid bath,
519 and refurbishment without safety measures. However, very few studies have been carried out
520 where hazardous emissions in environmental media are conducted to assess their magnitude and
521 health effects. This is the first comprehensive study, where passive air samplers were deployed at
522 40 e-waste recycling facilities across multiple cities (n=9) for a year-long continuous sampling of
523 heavy metals (Cr, Ni, Pb, Cu, Mn, Zn and Cd). Simultaneously, the soil samples were also taken
524 from the same sites, once in each season. Comparatively, the major urban centers i.e., Karachi,
525 Lahore, Gujranwala, and Faisalabad showed the highest levels of HMs in air and soil.
526 Concentrations of HM in air and soil were generally higher during dry and cold seasons as
527 compared to wet and hot seasons. E-waste facilities were shown to be hotspots of HMs leading to
528 concern for workers at these facilities as well as nearby populations. The results of I_{geo} index depict
529 that Gujranwala, Lahore, Karachi, and Faisalabad are in range of moderately to extremely
530 contaminated for most the HMs studied especially Cu, Ni, Pb and Cd. The present informal e-
531 waste dismantling and recycling practices in Pakistan highlights the need for stricter regulatory
532 frameworks around e-waste and improvements to recycling practices and technologies.
533 Given the large-scale informal e-waste recycling operations in the country, further research
534 involving more detailed sampling, especially reference sites in every studied city would help in

forming a clearer spatial trend. Transect studies for air and soil would also help to delineate the extent to which these hot spots of HMs exert effects on local populations and the environment. In addition, exploring bioavailability of HMs in air (particulate matter) and soil may also be an additional source to determine the potential harmful risks caused by synergistic effects by presence of HMs in several environmental matrices. For instance, oxidative potential of airborne particulate matter, has been shown to be driven by high levels of certain HMs such as Fe and Cu, among others. HMs associated with particulate matter, when inhaled, lead to the formation of reactive oxygen species, which negatively impact health through damage of cardiovascular and respiratory tissues (Cohen et al., 2015; Shahpoury et al., 2021).

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5. References

Abdulaziz, M., Alshehri, A., Badri, H., Summan, A., & Sayqal, A. (2022). Concentration Level and Health Risk Assessment of Heavy Metals in PM 2.5 in Ambient Air of Makkah City, Saudi Arabia. Polish Journal of Environmental Studies, 31(5).

559 Ahmad, W., Alharthy, R.D., Zubair, M., Ahmed, M., Hameed, A., Rafique, S., 2021. Toxic and
560 heavy metals contamination assessment in soil and water to evaluate human health risk.
561 Sci. Rep. 11, 17006. https://doi.org/10.1038/s41598-021-94616-4
562 Ajani, M. B., Maleka, P. P., Penabei, S., & Usman, I. T. (2022). Health risk assessment of heavy
563 metals concentration from soil; a case study of the Mayo-Dallah in Southern area of Chad.
564 Journal of Radiation Research and Applied Sciences, 15(1), 130-138.
565 Al-Khashman, O. A. (2013). Assessment of heavy metals contamination in deposited street dusts
566 in different urbanized areas in the city of Ma'an, Jordan. Environmental earth sciences, 70,
567 2603-2612.
568 Arya, S., Rautela, R., Chavan, D., Kumar, S., 2021. Evaluation of soil contamination due to crude
569 E-waste recycling activities in the capital city of India. Process Saf. Environ. Prot. 152,
570 641–653. https://doi.org/10.1016/j.psep.2021.07.001
571 ASTM, 2000. 2974 standard test methods for moisture, ash, and organic matter of peat and other
572 organic soils. Am. Soc. Test. Mater. Doi 10, D2974.
573 Aydın, H., Tepe, Y., & Ustaoğlu, F. (2023). A holistic approach to the eco-geochemical risk
574 assessment of trace elements in the estuarine sediments of the Southeastern Black Sea.
575 Marine Pollution Bulletin, 189, 114732.
576
577 Bawwab, M., Qutob, A., Khatib, M.A., Malassa, H., Shawahna, A., Qutob, M., 2022. Evaluation
578 of Heavy Metal Concentrations in Soil and Edible Vegetables Grown in Compost from
579 Unknown Sources in Al-Jiftlik, Palestine. J. Environ. Prot. 13, 112–125.
580 https://doi.org/10.4236/jep.2022.131007
581 Bellinger, D.C., 2011. The Protean Toxicities of Lead: New Chapters in a Familiar Story. Int. J.
582 Environ. Res. Public. Health 8, 2593–2628. https://doi.org/10.3390/ijerph8072593
583 Birgül, A., & Kurt-Karakuş, P. B. (2024). Air monitoring of organochlorine pesticides (OCPs) in
584 Bursa Türkiye: Levels, temporal trends and risk assessment. Science of The Total
585 Environment, 912, 169397
586 Brindhadevi, K., Barceló, D., Lan Chi, N.T., Rene, E.R., 2023. E-waste management, treatment
587 options and the impact of heavy metal extraction from e-waste on human health: Scenario
588 in Vietnam and other countries. Environ. Res. 217, 114926.
589 https://doi.org/10.1016/j.envres.2022.114926
590 Chalvatzaki, E., Chatoutsidou, S. E., Lehtomäki, H., Almeida, S. M., Eleftheriadis, K., Hänninen,
591 O., & Lazaridis, M. (2019). Characterization of human health risks from particulate air
592 pollution in selected European cities. Atmosphere, 10(2), 96.
593 Cohen, A. J., Brauer, M., Burnett, R., Anderson, H. R., Frostad, J., Estep, K., ... & Forouzanfar, M.
594 H. (2017). Estimates and 25-year trends of the global burden of disease attributable to
595 ambient air pollution: an analysis of data from the Global Burden of Diseases Study 2015.
596 The lancet, 389(10082), 1907-1918.
597 Duan, X., Yan, Y., Li, R., Deng, M., Hu, D., & Peng, L. (2021). Seasonal variations, source
598 apportionment, and health risk assessment of trace metals in PM2. 5 in the typical industrial
599 city of changzhi, China. Atmospheric Pollution Research, 12(1), 365-374.
600 Dutta, D., Goel, S., Kumar, S., 2022. Health risk assessment for exposure to heavy metals in soils
601 in and around E-waste dumping site. J. Environ. Chem. Eng. 10, 107269.
602 https://doi.org/10.1016/j.jece.2022.107269
603 El-Naggar, A., Ahmed, N., Mosa, A., Niazi, N.K., Yousaf, B., Sharma, A., Sarkar, B., Cai, Y.,
604 Chang, S.X., 2021. Nickel in soil and water: Sources, biogeochemistry, and remediation

using biochar. J. Hazard. Mater. 419, 126421. https://doi.org/10.1016/j.jhazmat.2021.126421
Eng, A., Harner, T., Pozo, K., 2014. A Prototype Passive Air Sampler for Measuring Dry Deposition of Polycyclic Aromatic Hydrocarbons. Environ. Sci. Technol. Lett. 1, 77–81. https://doi.org/10.1021/ez400044z
Fabretti, J. F., Sauret, N., Gal, J. F., Maria, P. C., & Schärer, U. (2009). Elemental characterization and source identification of PM2. 5 using Positive Matrix Factorization: The Malraux road tunnel, Nice, France. Atmospheric Research, 94(2), 320-329.
Forti, V., Baldé, C.P., Kuehr, R., Bel, G., 2020. The global e-waste monitor 2020. U. N. Univ. UNU Int. Telecommun. Union ITU Int. Solid Waste Assoc. ISWA BonnGenevaRotterdam 120.
Fosu-Mensah, B. Y., Addae, E., Yirenya-Tawiah, D., & Nyame, F. (2017). Heavy metals concentration and distribution in soils and vegetation at Korle Lagoon area in Accra, Ghana. Cogent Environmental Science, 3(1), 1405887-1405901.
Gaga, E.O., Harner, T., Dabek-Zlotorzynska, E., Celo, V., Evans, G., Jeong, C.-H., Halappanavar, S., Jariyasopit, N., Su, Y., 2019. Polyurethane Foam (PUF) Disk Samplers for Measuring Trace Metals in Ambient Air. Environ. Sci. Technol. Lett. 6, 545–550. https://doi.org/10.1021/acs.estlett.9b00420
Grant, K., Goldizen, F. C., Sly, P. D., Brune, M. N., Neira, M., van den Berg, M., & Norman, R. E. (2013). Health consequences of exposure to e-waste: a systematic review. The lancet global health, 1(6), e350-e361.
Gupta, V., 2020. Vehicle-Generated Heavy Metal Pollution in an Urban Environment and Its Distribution into Various Environmental Components, in: Shukla, V., Kumar, N. (Eds.), Environmental Concerns and Sustainable Development: Volume 1: Air, Water and Energy Resources. Springer, Singapore, pp. 113–127. https://doi.org/10.1007/978-981-13-5889-0_5
Ha, N.N., Agusa, T., Ramu, K., Tu, N.P.C., Murata, S., Bulbule, K.A., Parthasaraty, P., Takahashi, S., Subramanian, A., Tanabe, S., 2009. Contamination by trace elements at e-waste recycling sites in Bangalore, India. Chemosphere 76, 9–15. https://doi.org/10.1016/j.chemosphere.2009.02.056.
Hameed, H. B., Ali, Y., & Petrillo, A. (2020). Environmental risk assessment of E-waste in developing countries by using the modified-SIRA method. Science of The Total Environment, 733, 138525.
Han, Y., Ni, Z., Li, S., Qu, M., Tang, F., Mo, R., ... & Liu, Y. (2018). Distribution, relationship, and risk assessment of toxic heavy metals in walnuts and growth soil. Environmental Science and Pollution Research, 25, 17434-17443.
Han, Y., Tang, Z., Sun, J., Xing, X., Zhang, M., Cheng, J., 2019. Heavy metals in soil contaminated through e-waste processing activities in a recycling area: Implications for risk management. Process Saf. Environ. Prot. 125, 189–196.
Hashmi, M.Z., Chen, K., Khalid, F., Yu, C., Tang, X., Li, A., Shen, C., 2022. Forty years studies on polychlorinated biphenyls pollution, food safety, health risk, and human health in an e-waste recycling area from Taizhou city, China: a review. Environ. Sci. Pollut. Res. 29, 4991–5005. https://doi.org/10.1007/s11356-021-17516-0
Houessionon, M.G.K., Ouendo, E.-M.D., Bouland, C., Takyi, S.A., Kedote, N.M., Fayomi, B., Fobil, J.N., Basu, N., 2021. Environmental Heavy Metal Contamination from Electronic Waste (E-Waste) Recycling Activities Worldwide: A Systematic Review from 2005 to 2017. Int. J. Environ. Res. Public. Health 18, 3517. https://doi.org/10.3390/ijerph18073517

IARC, 2023. Monographs on the identification of carcinogenic hazards to humans. Available online: https://monographs.iarc.who.int/list-of-classifications/ (Accessed on 16 January 2023).

Imran, M., Haydar, S., Kim, J., Awan, M.R., Bhatti, A.A., 2017. E-waste flows, resource recovery and improvement of legal framework in Pakistan. Resour. Conserv. Recycl. 125, 131–138. https://doi.org/10.1016/j.resconrec.2017.06.015

Iqbal, M., Breivik, K., Syed, J.H., Malik, R.N., Li, J., Zhang, G., Jones, K.C., 2015. Emerging issue of e-waste in Pakistan: A review of status, research needs and data gaps. Environ. Pollut. 207, 308–318. https://doi.org/10.1016/j.envpol.2015.09.002

Iqbal, M., Syed, J.H., Breivik, K., Chaudhry, M.J.I., Li, J., Zhang, G., Malik, R.N., 2017. E-Waste Driven Pollution in Pakistan: The First Evidence of Environmental and Human Exposure to Flame Retardants (FRs) in Karachi City. Environ. Sci. Technol. 51, 13895–13905. https://doi.org/10.1021/acs.est.7b03159

Isimekhai, K.A., Garelick, H., Watt, J., Purchase, D., 2017. Heavy metals distribution and risk assessment in soil from an informal E-waste recycling site in Lagos State, Nigeria. Environ. Sci. Pollut. Res. 24, 17206–17219. https://doi.org/10.1007/s11356-017-8877-9

Kazim, M., Hussain Syed, J., Binnur Kurt-Karakus, P., Ozkaleli Akcetin, M., Akram, S., Birgul, A., Kara, M., Dumanoglu, Y., Odabasi, M., Saqib, Z., Nasir, J., Mahar, A., Shah, T., Gulzar Burq, M., Jones, K.C., Wania, F., 2023. Gaseous elemental mercury emissions from informal E-Waste recycling facilities in Pakistan. Waste Manag. 170, 261–269. https://doi.org/10.1016/j.wasman.2023.09.014

Khan, M.U., Malik, R.N., Muhammad, S., Ullah, F., Qadir, A., 2015. Health Risk Assessment of Consumption of Heavy Metals in Market Food Crops from Sialkot and Gujranwala Districts, Pakistan. Hum. Ecol. Risk Assess. Int. J. 21, 327–337. https://doi.org/10.1080/10807039.2014.913445

Khan, S. A., Muhammad, S., Nazir, S., & Shah, F. A. (2020). Heavy metals bounded to particulate matter in the residential and industrial sites of Islamabad, Pakistan: Implications for non-cancer and cancer risks. Environmental Technology & Innovation, 19, 100822.

Kodat, M., & Tepe, Y. (2023). A holistic approach to the assessment of heavy metal levels and associated risks in the coastal sediment of Giresun, southeast Black Sea. Heliyon, 9(6).

Kumar, A., Holuszko, M., & Espinosa, D. C. R. (2017). E-waste: An overview on generation, collection, legislation and recycling practices. Resources, Conservation and Recycling, 122, 32-42.

Kumar, P., Fulekar, M.H., 2019. Multivariate and statistical approaches for the evaluation of heavy metals pollution at e-waste dumping sites. SN Appl. Sci. 1, 1506. https://doi.org/10.1007/s42452-019-1559-0

Kuntawee, C., Tantrakarnapa, K., Limpanont, Y., Lawpoolsri, S., Phetrak, A., Mingkhwan, R., Worakhunpiset, S., 2020. Exposure to Heavy Metals in Electronic Waste Recycling in Thailand. Int. J. Environ. Res. Public. Health 17, 2996. https://doi.org/10.3390/ijerph17092996

Li J, Duan H, Shi P (2011) Heavy metal contamination of surface soil in electronic waste dismantling area: site investigation and source-apportionment analysis. Waste Manag Res 29(7):727– 738. https ://doi.org/10.1177/07342 42X10 397580

Li, Q., Yang, K., Li, J., Zeng, X., Yu, Z., & Zhang, G. (2018). An assessment of polyurethane foam passive samplers for atmospheric metals compared with active samplers. Environmental Pollution, 236, 498-504.

Mahmood, A., Malik, R.N., 2014. Human health risk assessment of heavy metals via consumption of contaminated vegetables collected from different irrigation sources in Lahore, Pakistan. Arab. J. Chem., Special Issue: Environmental Chemistry 7, 91–99. https://doi.org/10.1016/j.arabjc.2013.07.002

Mowla, M., Rahman, E., Islam, N., & Aich, N. (2021). Assessment of heavy metal contamination and health risk from indoor dust and air of informal E-waste recycling shops in Dhaka, Bangladesh. Journal of Hazardous Materials Advances, 4, 100025-100034.

Muller, G. (1981). Heavy metals and other pollutants in the environment: a chronology based on the analysis of dated sediments. In Heavy metals in the Environment. International Conference, Amsterdam- September (p. 1981).

Muller, G. M. M. G. M. G. M. G. P. (1969). Index of geoaccumulation in sediments of the Rhine River. Geojournal, 2, 108-118.

Murthy, V., & Ramakrishna, S. (2022). A review on global E-waste management: urban mining towards a sustainable future and circular economy. Sustainability, 14(2), 647.

Nasir, J., Zeb, B., Sorooshian, A., Mansha, M., Alam, K., Ahmad, I., Rizvi, H.H., Shafiq, M., 2019. Spatio-temporal variations of absorbing aerosols and their relationship with meteorology over four high altitude sites in glaciated region of Pakistan. J. Atmospheric Sol.-Terr. Phys. 190, 84–95. https://doi.org/10.1016/j.jastp.2019.05.010

Obeng-Gyasi, E., 2018. Lead Exposure and Oxidative Stress—A Life Course Approach in U.S. Adults. Toxics 6, 42. https://doi.org/10.3390/toxics6030042

Osobamiro, T., Awolesi, O., Alabi, O., Oshinowo, A., Idris, M., Busari, F., 2019. Heavy Metal Levels of Soil Samples Collected From a Major Industrial Area in Abeokuta, Southwestern Nigeria. Indian J. Sci. Res. 9, 914–919. https://doi.org/10.29322/IJSRP.9.08.2019.p92125

Pasha, M. J., & Alharbi, B. H. (2015). Characterization of size–fractionated PM10 and associated heavy metals at two semi–arid holy sites during Hajj in Saudi Arabia. Atmospheric Pollution Research, 6(1), 162-172.

Perkins, D.N., Brune Drisse, M.-N., Nxele, T., Sly, P.D., 2014. E-Waste: A Global Hazard. Ann. Glob. Health 80, 286–295. https://doi.org/10.1016/j.aogh.2014.10.001

Pradhan, J.K., Kumar, S., 2014. Informal e-waste recycling: environmental risk assessment of heavy metal contamination in Mandoli industrial area, Delhi, India. Environ. Sci. Pollut. Res. 21, 7913–7928. https://doi.org/10.1007/s11356-014-2713-2

Purchase, D., Abbasi, G., Bisschop, L., Chatterjee, D., Ekberg, C., Ermolin, M., ... & Wong, M. H. (2020). Global occurrence, chemical properties, and ecological impacts of e-wastes (IUPAC Technical Report). Pure and Applied Chemistry, 92(11), 1733-1767.

Qin, Y., Liu, Y., Wang, J., Lu, Y., Xu, Z., 2022. Emission of PAHs, PCBs, PBDEs and heavy metals in air, water and soil around a waste plastic recycling factory in an industrial park, Eastern China. Chemosphere 294, 133734. https://doi.org/10.1016/j.chemosphere.2022.133734

Rafeeq, A., Ali, S. A., Tanoli, A. K., Akhter, N., & Raza, G. (2021). Analytical Profiling of Heavy Metals Contamination in soils, Dismantling Dust, and Rubber Samples in Karachi City Using AAS, WD-XRF, and SEM Technique. Indonesian Journal of Social and Environmental Issues (IJSEI), 2(3), 242-257.

Rahman, M. S., Jolly, Y. N., Akter, S., Kamal, N. A., Rahman, R., Choudhury, T. R., & Begum, B. A. (2021). Sources of toxic elements in indoor dust sample at export processing zone (EPZ) area: Dhaka, Bangladesh; and their impact on human health. Environmental Science and Pollution Research, 28, 39540-39557.

Sajid, M., Syed, J.H., Iqbal, M., Abbas, Z., Hussain, I., Baig, M.A., 2019. Assessing the generation, recycling and disposal practices of electronic/electrical-waste (E-Waste) from major cities in Pakistan. Waste Manag. 84, 394–401.

Sakata, M., & Asakura, K. (2011). Atmospheric dry deposition of trace elements at a site on Asian-continent side of Japan. Atmospheric Environment, 45(5), 1075-1083.

Saleem, M., Iqbal, J., Akhter, G., Shah, M.H., 2018. Fractionation, bioavailability, contamination and environmental risk of heavy metals in the sediments from a freshwater reservoir, Pakistan. J. Geochem. Explor. 184, 199–208.

Shahpoury, P., Zhang, Z. W., Arangio, A., Celo, V., Dabek-Zlotorzynska, E., Harner, T., & Nenes, A. (2021). The influence of chemical composition, aerosol acidity, and metal dissolution on the oxidative potential of fine particulate matter and redox potential of the lung lining fluid. Environment international, 148, 106343.

Shaikh, S., 2021. A critical analysis of e-waste management and recycling in Pakistan: a life cycle assessment (PhD Thesis). Victoria University.

Shakil, S., Nawaz, K., Sadef, Y., 2023. Evaluation and environmental risk assessment of heavy metals in the soil released from e-waste management activities in Lahore, Pakistan. Environ. Monit. Assess. 195, 89.

Singh, M., Thind, P.S., John, S., 2018. Health risk assessment of the workers exposed to the heavy metals in e-waste recycling sites of Chandigarh and Ludhiana, Punjab, India. Chemosphere 203, 426–433. https://doi.org/10.1016/j.chemosphere.2018.03.138.

Song, Q., Li, J., 2014. Environmental effects of heavy metals derived from the e-waste recycling activities in China: a systematic review. Waste Manag. 34, 2587–2594.

Sthiannopkao, S., & Wong, M. H. (2013). Handling e-waste in developed and developing countries: Initiatives, practices, and consequences. Science of the Total Environment, 463, 1147-1153.

Sun, S., Zheng, N., Wang, S., Li, Y., Hou, S., Song, X., ... & Dong, D. (2021). Source analysis and human health risk assessment based on entropy weight method modification of PM2. 5 heavy metal in an industrial area in the Northeast of China. Atmosphere, 12(7), 852.

Tang X, Shen C, Chen L, Xiao X, Wu J, Khan MI et al (2010) Inorganic and organic pollution in agricultural soil from an emerging e-waste recycling town in Taizhou area. China. Journal of Soils and Sediments 10(5):895–906. https ://doi.org/10.1007/ s1136 8-010-0252-0

Taylor, S. R. (1964). Abundance of chemical elements in the continental crust: a new table. Geochimica et cosmochimica acta, 28(8), 1273-1285.

Tchounwou, P.B., Yedjou, C.G., Patlolla, A.K., Sutton, D.J., 2012. Heavy Metals Toxicity and the Environment. EXS 101, 133–164. https://doi.org/10.1007/978-3-7643-8340-4_6

Tepe, Y., Şimşek, A., Ustaoğlu, F., & Taş, B. (2022). Spatial–temporal distribution and pollution indices of heavy metals in the Turnasuyu Stream sediment, Turkey. Environmental monitoring and assessment, 194(11), 818

Teye, A.K., Tetteh, I.K., 2023. Heavy Metal Pollution Resulting from Informal E-Waste Recycling in the Greater Accra Region of Ghana, in: Heavy Metals-Recent Advances. IntechOpen.

Umair, S., Anderberg, S., Potting, J., 2016. Informal Electronic Waste Recycling in Pakistan. J. Solid Waste Technol. Manag. 42, 222–235. https://doi.org/10.5276/JSWTM.2016.222

United States. Environmental Protection Agency. Office of Emergency & Remedial Response. (1989). Risk Assessment Guidance for Superfund: pt. A. Human health evaluation manual (Vol. 1). Office of Emergency and Remedial Response, US Environmental Protection Agency.

US EPA (Ed.), 2001. Risk Assessment guidance for superfund: volume III-part a.
US EPA, 2000. Assigning values to non-detected/non-quantified pesticide residues in human health food exposure assessments. Office of Pesticide Programs, US Environmental Protection Agency.
US EPA, 2004. Method 9045D Soil and Waste pH; USEPA: Washinghton, DC, USA,2004.
USEPA (United States Environmental Protection Agency). (1997). Exposure factors handbook. Washington, DC: US Environmental Protection Agency [EPA/600/P-95/ 002F a-c].
Wu, K. G., Chang, C. Y., Yen, C. Y., & Lai, C. C. (2019). Associations between environmental heavy metal exposure and childhood asthma: A population-based study. Journal of Microbiology, Immunology and Infection, 52(2), 352-362.
Wu, W., Wu, P., Yang, F., Sun, D. L., Zhang, D. X., & Zhou, Y. K. (2018). Assessment of heavy metal pollution and human health risks in urban soils around an electronics manufacturing facility. Science of the Total Environment, 630, 53-61.
Zhang, X., Eto, Y., & Aikawa, M. (2021). Risk assessment and management of PM2. 5-bound heavy metals in the urban area of Kitakyushu, Japan. Science of The Total Environment, 795, 148748.
Zhou, S., Yuan, Q., Li, W., Lu, Y., Zhang, Y., Wang, W., 2014. Trace metals in atmospheric fine particles in one industrial urban city: Spatial variations, sources, and health implications. J. Environ. Sci. 26, 205–213. https://doi.org/10.1016/S1001-0742(13)60399-X

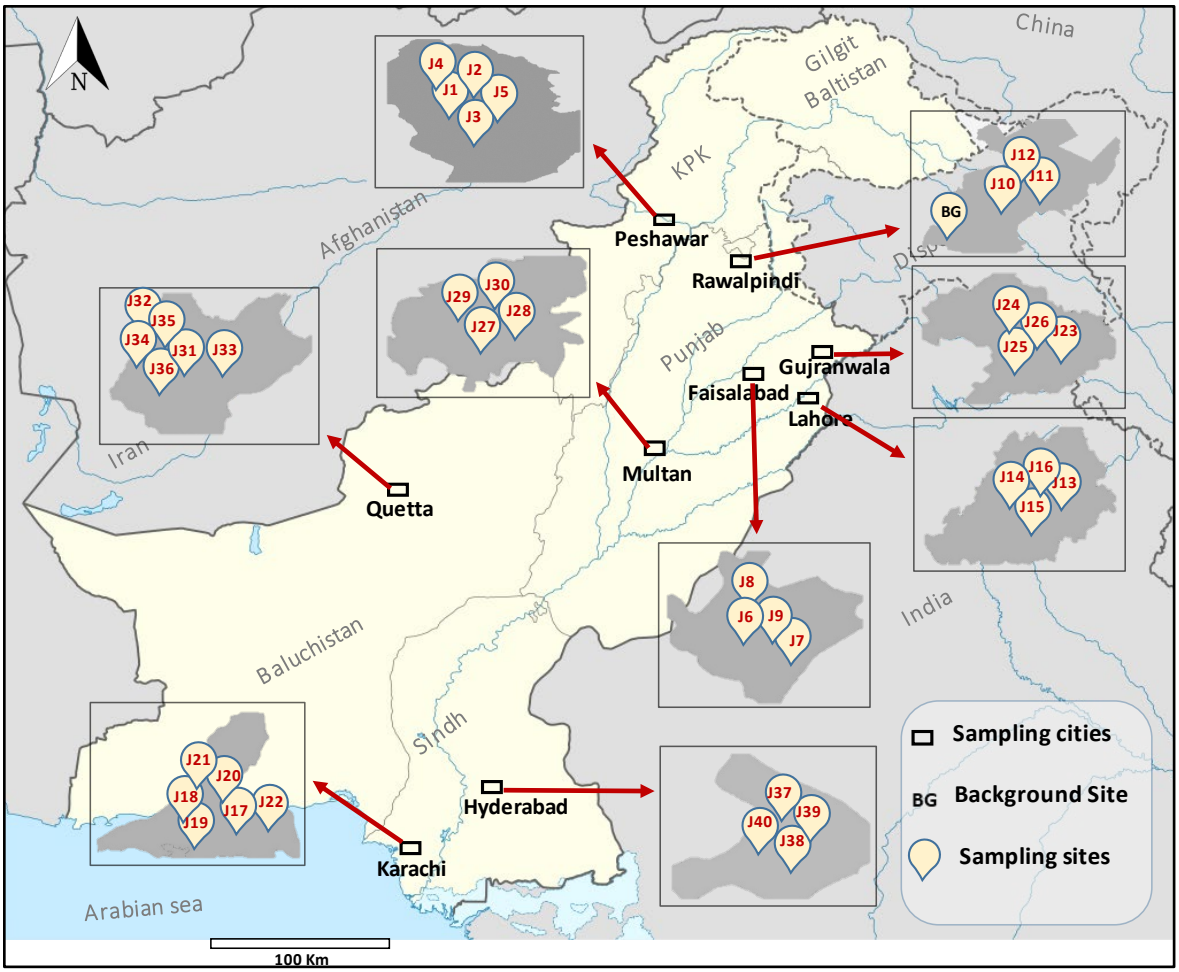


Figure 1: Geographical locations of cities in focus and sampling network (refer to Table S1 for which sampling site code (J1, J2 etc.) belongs to which sampling site)

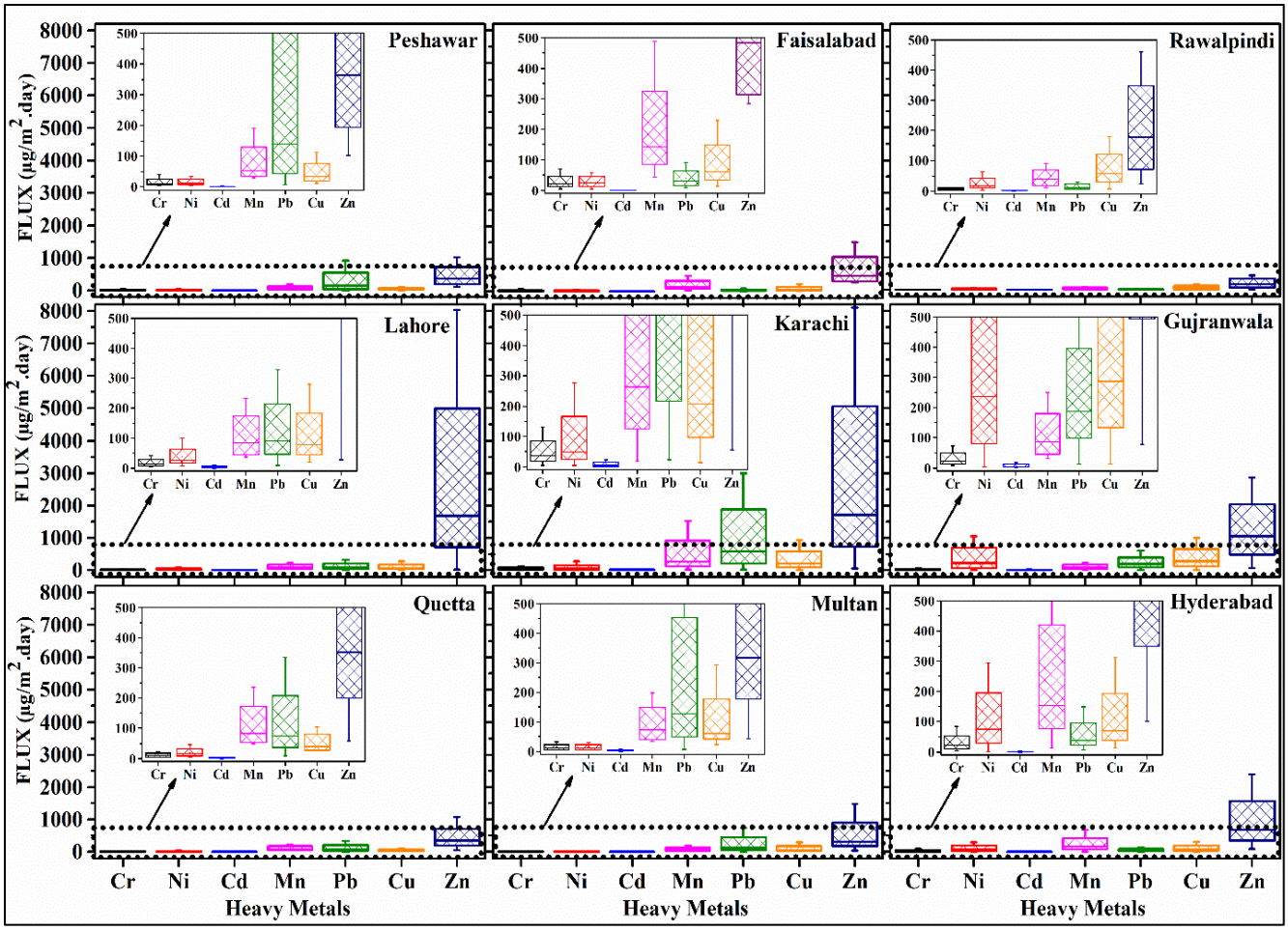


Figure 2: An intra-city comparison of heavy metals (flux (µg/m^2.day) in air

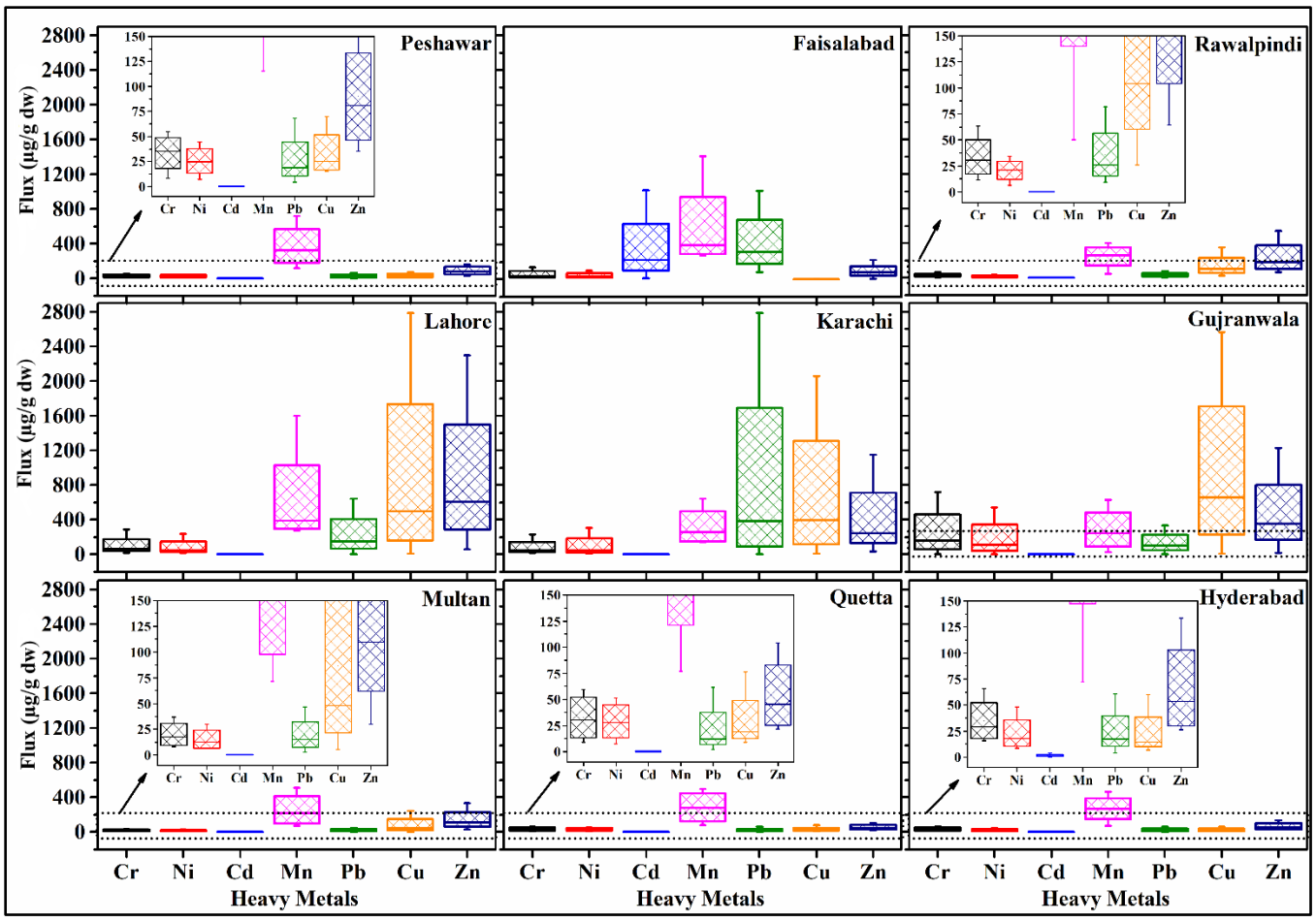
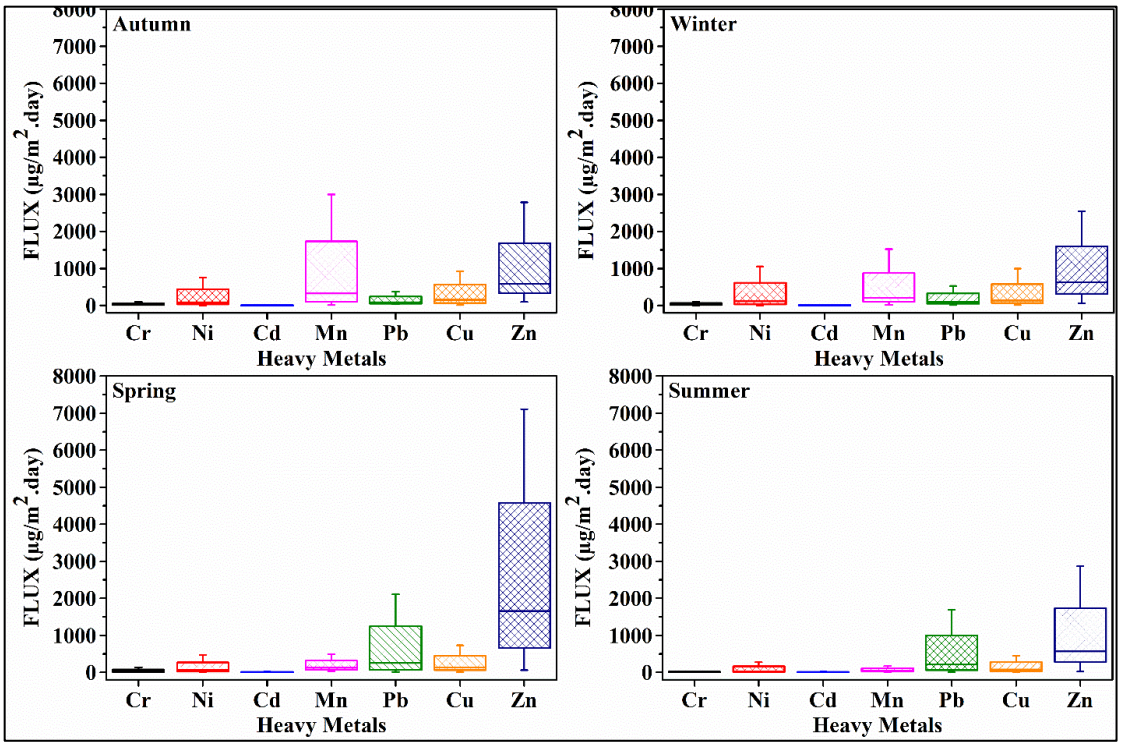
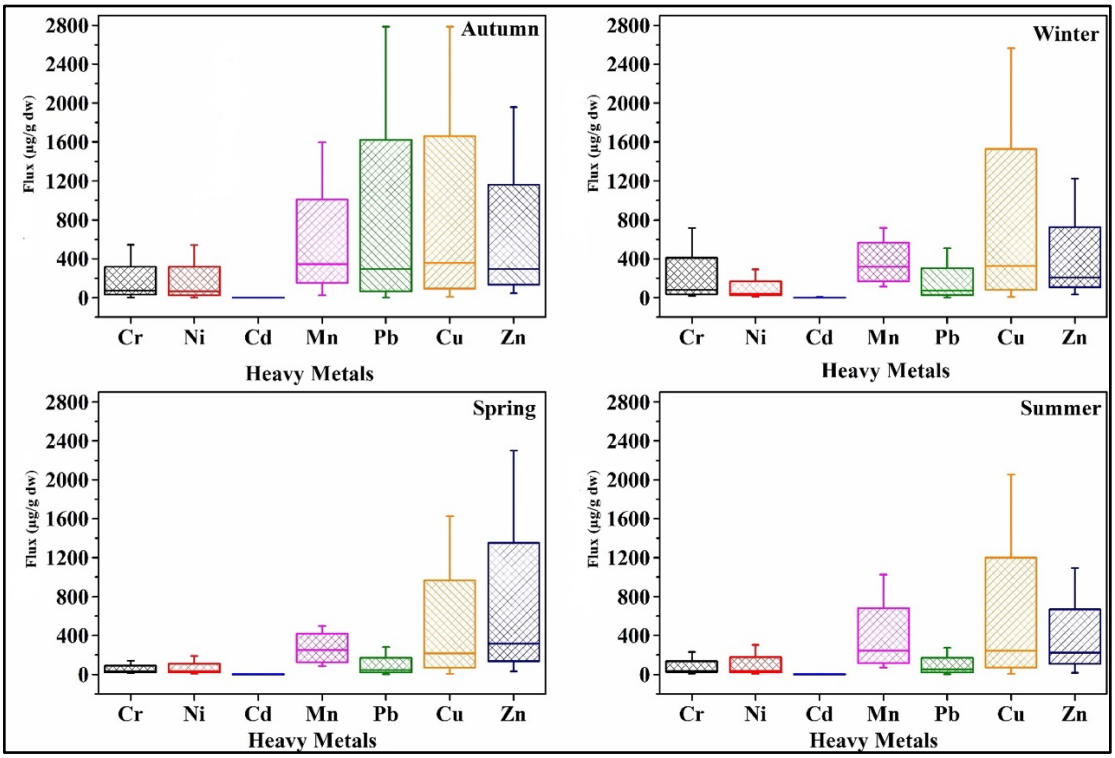


Figure 3: An intra-city comparison of targeted metals in soils (µg/g dw)



(a).



(b).

Figure 4: Seasonal variations of studied heavy metals in (a). air & (b). soil

Table 1: Descriptive statistics of heavy metals at e-waste recycling facilities (all studied sites) and background site (Islamabad) in Pakistan

HMs	Concentration level in air (Flux, µg/m^2.day)				Concentration level in soils (µg/g dw)			
	E-waste Recycling sites		Background site		E-waste Recycling sites		Background site	
	Mean+S.D	Range	Mean+S.D	Range	Mean+S.D	Range	Mean+S.D	Range
Cr	18.9±13.8	2.3-131	5.34±2.58	2.23-7.66	49.4±35.5	3.6-716	19.5±3.10	16.5-23.8
Mn	116±95.8	10.5-1520	33.47±13.3	15.8-45.8	372±152	25.1-1599	210±59.9	161-296
Ni	42.2±61.8	1.1-1057	10.0±7.33	3.08-20.2	37.3±26.1	2.9-542	12.2±4.60	8.3-18.8
Cu	97±52.4	7.5-1000	24.7±17.0	5.67-40.8	155±85.3	5.4-2784	11.9±2.06	8.9-13.63
Zn	733±273	23.6-8105	169±110	67.1-323	220±96.9	15.8-2301	45.5±3.81	41.6-49.3
Cd	1.8±0.3	0.1-23.4	0.18±0.09	0.03-0.24	0.6±0.3	0.03-6.3	0.11±0.02	0.10-0.14
Pb	112±74.5	5.9-2992	49.9±86.1	2.35-179	63.1±46.3	2.2-2786	7.97±2.07	4.87-9.16
Avg.	161±111	0.1-8105	41.8 + 33.8	0.03-179	128±63.2	0.03-2786	43.8±10.8	0.10-296

Table 2: ILCRing, ILCRinh, ILCRdermal and Cumulative ILCR for Cd, Cr, Ni and Pb (carcinogenic heavy metals)

Sampling cities	Exposure Risk	Cr	Ni	Cd	Pb	ΣILCR
Peshawer	$ILCR_{\Sigma ing}$	8.55×10^{-6}	1.05×10^{-5}	1.51×10^{-6}	7.18×10^{-8}	2.06×10^{-5}
	$ILCR_{\Sigma inh}$	9.47×10^{-5}	3.91×10^{-7}	1.75×10^{-7}	9.76×10^{-8}	9.53×10^{-5}
	$ILCR_{\Sigma Der}$	1.70×10^{-4}	2.55×10^{-6}	1.51×10^{-7}	1.81×10^{-5}	1.91×10^{-4}
Faislabad	$ILCR_{\Sigma ing}$	9.32×10^{-6}	1.38×10^{-5}	6.66×10^{-6}	3.06×10^{-7}	3.01×10^{-5}
	$ILCR_{\Sigma inh}$	1.91×10^{-4}	7.51×10^{-7}	9.49×10^{-8}	5.33×10^{-8}	1.92×10^{-4}
	$ILCR_{\Sigma Der}$	1.86×10^{-4}	3.35×10^{-6}	6.79×10^{-7}	7.48×10^{-5}	2.64×10^{-4}
Rawalpindi	$ILCR_{\Sigma ing}$	7.47×10^{-6}	8.41×10^{-6}	2.05×10^{-6}	1.06×10^{-7}	1.80×10^{-5}
	$ILCR_{\Sigma inh}$	6.68×10^{-5}	4.31×10^{-7}	7.27×10^{-8}	2.13×10^{-8}	6.73×10^{-5}
	$ILCR_{\Sigma Der}$	1.49×10^{-4}	2.04×10^{-6}	2.08×10^{-7}	2.58×10^{-5}	1.77×10^{-4}
Lahore	$ILCR_{\Sigma ing}$	1.22×10^{-5}	1.52×10^{-5}	4.52×10^{-6}	4.18×10^{-7}	3.23×10^{-5}
	$ILCR_{\Sigma inh}$	1.38×10^{-4}	6.16×10^{-7}	5.96×10^{-7}	1.14×10^{-7}	1.39×10^{-4}
	$ILCR_{\Sigma Der}$	2.43×10^{-4}	3.69×10^{-6}	4.52×10^{-7}	1.02×10^{-4}	3.49×10^{-4}
Karachi	$ILCR_{\Sigma ing}$	8.00×10^{-6}	1.27×10^{-5}	5.35×10^{-6}	5.97×10^{-7}	2.66×10^{-5}
	$ILCR_{\Sigma inh}$	3.15×10^{-4}	1.08×10^{-6}	6.80×10^{-7}	5.19×10^{-7}	3.17×10^{-4}
	$ILCR_{\Sigma Der}$	1.59×10^{-4}	3.07×10^{-6}	5.35×10^{-7}	1.48×10^{-4}	3.11×10^{-4}
Gujranwala	$ILCR_{\Sigma ing}$	2.25×10^{-5}	2.71×10^{-5}	8.13×10^{-6}	2.98×10^{-7}	5.80×10^{-5}
	$ILCR_{\Sigma inh}$	2.01×10^{-4}	3.86×10^{-6}	9.61×10^{-7}	2.40×10^{-7}	2.06×10^{-4}
	$ILCR_{\Sigma Der}$	$4.48\times10\text{-}^{4}$	6.54×10^{-6}	8.15×10^{-7}	7.40×10^{-5}	5.30×10^{-4}
Multan	$ILCR_{\Sigma ing}$	4.97×10^{-6}	6.39×10^{-6}	1.98×10^{-6}	6.06×10^{-8}	1.34×10^{-5}
	$ILCR_{\Sigma inh}$	1.07×10^{-4}	4.64×10^{-7}	1.09×10^{-7}	8.15×10^{-8}	1.08×10^{-4}
	$ILCR_{\Sigma Der}$	9.89×10^{-5}	1.55×10^{-6}	2.01×10^{-7}	1.52×10^{-5}	1.16×10^{-4}
Quetta	$ILCR_{\Sigma ing}$	8.94×10^{-6}	1.26×10^{-5}	9.13×10^{-7}	4.14×10^{-8}	2.25×10^{-5}
	$ILCR_{\Sigma inh}$	1.16×10^{-4}	3.75×10^{-7}	1.61×10^{-7}	1.05×10^{-7}	1.16×10^{-4}
	$ILCR_{\Sigma Der}$	1.78×10^{-4}	3.06×10^{-6}	9.05×10^{-8}	1.07×10^{-5}	1.92×10^{-4}
Hyderabad	$ILCR_{\Sigma ing}$	7.76×10^{-6}	8.14×10^{-6}	4.25×10^{-6}	6.25×10^{-8}	2.02×10^{-5}
	$ILCR_{\Sigma inh}$	1.62×10^{-4}	1.39×10^{-6}	1.21×10^{-7}	5.15×10^{-8}	1.63×10^{-4}
	$ILCR_{\Sigma Der}$	1.55×10^{-4}	1.96×10^{-6}	4.33×10^{-7}	1.55×10^{-5}	1.72×10^{-4}