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Research Paper

Volatile organic compounds and metals/metalloids exposure in children after e-waste control: Implications for priority control pollutants and exposure mitigation measures

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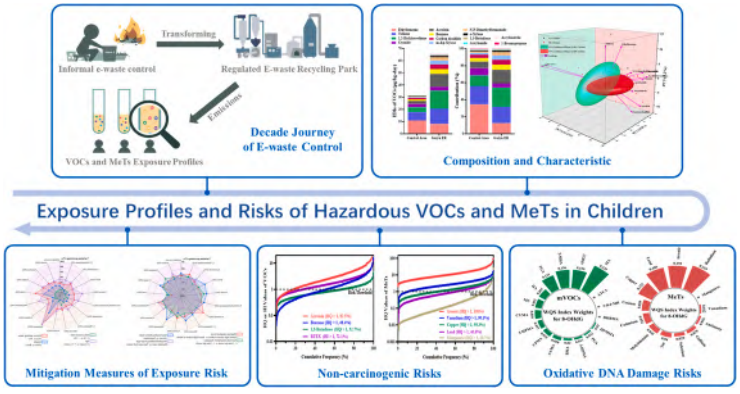
HIGHLIGHTS

- We profile the exposure risk of VOCs and MeTs in children from a regulated ER.
- Exposure to VOCs and MeTs is significantly associated with oxidative DNA damage.
- Exposure to six VOCs and four MeTs poses considerable health risks to ER children.
- Enhancing physical exercise facilitates mitigating pollutant exposure risks.
- TGA/PGA ratio and TGA may be used as diagnostic indexes to identify e-waste pollution.

GRAPHICAL ABSTRACT



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ABSTRACT

The decade-long effort to control e-waste in China has made significant progress from haphazard disposal to organized recycling, but environmental research suggests that exposure to volatile organic compounds (VOCs) and metals/metalloids (MeTs) still poses plausible health risks. To investigate the exposure risk faced by children and identify corresponding priority control chemicals, we evaluated the carcinogenic risk (CR), non-CR, and oxidative DNA damage risks of VOCs and MeTs exposure in 673 children from an e-waste recycling area (ER) by measuring urinary exposure biomarker levels. The ER children were generally exposed to high levels of VOCs and MeTs. We observed distinctive VOCs exposure profiles in ER children. In particular, the 1,2-dichloroethane/ethylbenzene ratio and 1,2-dichloroethane were promising diagnostic indexes for identifying e-waste pollution due to their high accuracy (91.4%) in predicting e-waste exposure. Exposure to acrolein, benzene, 1,3-butadiene, 1,2-dichloroethane, acrylamide, acrylonitrile, arsenic, vanadium, copper, and lead posed considerable CR or/and non-CR and oxidative DNA damage risks to children, while changing personal lifestyles, especially enhancing daily physical exercise, may facilitate mitigating these chemical exposure risks. These findings highlight that the exposure risk of some VOCs and MeTs is still non-negligible in regulated ER, and these hazardous chemicals should be controlled as priorities.

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1. Introduction

Informal recycling of electronic waste (e-waste) releases numerous chemicals into the surrounding environment, including heavy metals and persistent organic pollutants. These chemicals have been linked to DNA lesions, telomere attrition, oxidative stress, and immune function in populations living in e-waste-exposed regions [33]. Many countries have enacted e-waste legislation to mitigate environmental pollution and population health risks associated with informal e-waste recycling [10]. China, which produces the most e-waste annually, has implemented strict laws and regulations on e-waste management and tightened the e-waste import policies over the past decade [1,42]. The closure of informal workshops and the construction of organized e-waste recycling parks have contributed to significant environmental, economic, and social benefits [32,42], which provides a valuable reference for e-waste recycling management in other developing countries. Longitudinal population exposure monitoring studies have revealed a significant decline in the exposure levels of polycyclic aromatic hydrocarbons, polychlorinated biphenyls, polybrominated diphenyl ethers, and traditional organophosphorus flame retardants in residents after e-waste control [21,45]. However, our recent studies found that the exposure levels of volatile organic compounds (VOCs) and metals/metalloids (MeTs) in children did not decrease as expected after e-waste control, and adults traveling from the reference area (RA) to a regulated e-waste recycling area (ER) exhibited elevated exposure levels of most VOCs and MeTs [21,24]. The profound associations of exposure to VOCs and MeTs with oxidative DNA damage and adverse health outcomes identified previously [24,33] underpin that exposure to VOCs and MeTs may still pose significant health risks to the ER populations despite e-waste control.

VOCs are byproducts inevitably produced during the heat treatment and incineration of e-waste [28,30]. Although regulated e-waste recycling parks are equipped with environmental protection facilities to deal with the gas pollutants produced during the production process [32], VOCs with low dielectric constants are difficult to eliminate [28] The removal rate of some low-concentrated and highly toxic aromatic hydrocarbons, such as benzene, is extremely low and accumulates intensively in the outlet air [29]. Our recent unpublished longitudinal study also manifested that the exposure levels of two-thirds of the toxic VOCs investigated in children exhibited a significant increasing trend after e-waste control. This may be ascribed to the dramatic shift in e-waste dismantling patterns and VOC elimination technologies. In contrast to organic pollutants, as a component of e-waste, MeTs will persist in the environment and can diffuse between different environmental media, such as air, water, soil, and sediment, once discharged. The emissions of MeTs from e-waste sources into the surrounding environment have significantly decreased after strict e-waste control was implemented [32]. However, due to past e-waste recycling activity, unless thorough remediation measures are taken to minimize environmental MeT levels, the ER populations will continue to be exposed to high levels of MeTs through dietary and respiratory pathways [49]. Furthermore, airborne toxic MeTs from formal e-waste recycling parks can metastasize to surrounding residential areas, aggravating the exposure risk of residents [50]. Hence, it is necessary to verify the exposure risks of VOCs and MeTs in ER to prioritize the control of high-risk chemicals to mitigate population health risks.

The exposure risks of VOCs and MeTs in populations of ER after e-waste control are rarely investigated. Two recent studies estimated the inhalation risks of atmospheric VOCs and MeTs. They found that the non-carcinogenic risks (CR) of exposure to 25 VOCs and 6 MeTs were below the threshold, while exposure to 4 VOCs and arsenic posed unacceptable CR in residential areas [6,50]. However, relying solely on atmospheric VOCs and MeTs tends to underestimate the overall exposure risks since dermal contact and dietary intake are profound routes of exposure, especially for MeTs. Furthermore, the uncertainty of respiratory coefficients that vary significantly under different exposure levels may affect the accuracy of atmospheric VOCs exposure assessment [16]. In comparison, internal exposure risk analysis is superior to external exposure risk assessment as it can circumvent the exposure assessment bias caused by individual differences, unknown pollution sources, and unclear exposure routes. However, there is a lack of studies assessing the internal exposure risk of VOCs and MeTs in the general population of ER after e-waste control [33]. To fill this knowledge gap, this study utilized the internal exposure risk assessment method to evaluate the non-CR and CR of a large panel of VOCs and MeTs and examined their associations with oxidative DNA damage risk to profile the corresponding exposure risks faced by the susceptible population (*i.e.*, children) living in ER. This will facilitate determining which hazardous VOCs and MeTs should be prioritized for control. One step further, we attempted to identify the key factors that affect the exposure risk of VOCs and MeTs through a questionnaire survey to propose intervening measures to mitigate the exposure risks of ER children.

2. Methods

2.1. Population recruitment and sample collection

This research is an ongoing regional cohort study initiated in 2021 by the South China Institute of Environmental Sciences to investigate the potential health impacts of e-waste recycling. In November 2021, we recruited 673 children from a typical ER located in Guiyu Town, China. Children aged 5–13 years and settled in ER since birth were given priority for recruitment. These children were mainly from six villages in Guiyu, including Hua-Mei, Bei-Lin, Xian-Peng, Long-Gang, Nan-An, and Du-Tou villages, which are former e-waste recycling sites located < 3 kilometers away from the newly established e-waste recycling industrial park. All enrolled children provided signed informed consent and completed a questionnaire, including basic information (*e.g.*, gender, age, weight, height, and years of residency), family information (*e.g.*, maternal education level, maternal occupation, and paternal occupation), diet and lifestyle (*e.g.*, passive smoking status, daily exercise time, frequency of handwashing before meals and seafood consumption, household drinking water type, and fuel for cooking), and personal disease history. First-morning urine samples were collected from each child and stored at -20 °C until chemical analysis. In addition, to intuitively characterize the differences in VOCs exposure levels and compositions between ER and RA children, a previously reported population of children recruited from the suburbs of Guangzhou, China, was enrolled as controls [22]. This research was approved by the Research Ethics Committee of the South China Institute of Environmental Sciences, Ministry of Ecology and Environment.

2.2. Sample preparation and chemical analysis

Detailed measurement method for urinary trans-3′-hydroxy-cotinine (OH-Cot, a urinary biomarker of smoking or passive smoking) [19], 8-hydroxy-2′-deoxyguanosine (8-OHdG, a urinary biomarker of oxidative DNA damage), and metabolites of VOCs (mVOCs) had been described previously [20]. Briefly, urine samples thawed from –20 ℃ were centrifuged at 12000 rpm for 5 min. One mL of urine supernatant was transferred to a plastic pipe and mixed with formic acid buffer, isotope-internal standards, and β-glucuronidase/arylsulfatase enzyme. The mixed solutions were enzymolized at 37 °C overnight. The target analytes were then extracted by the solid phase extraction method using a polar-enhanced polymer cartridge (Cleanert, 60 mg/3 mL; Bonna-Agela Technologies, China). The analytes were eluted using a mixture of formic acid and acetonitrile (2:98, v-v) with a volume of 2 mL. After drying and redissolving, 100 μL of supernatant was used for instrumental analysis. The levels of urinary OH-Cot, 8-OHdG, and 25 mVOCs, including N-acetyl-S-(3,4-dihydroxybutyl)-L-cysteine (DHBMA), trans,trans-muconic acid (MU), N-acetyl-S-(2-carbamoylethyl)-L-cysteine (AAMA), N-acetyl-S-(N-methylcarbamoyl)-L-cysteine (AMCC), N-acetyl-S-(2-cyanoethyl)-L-cysteine

(CYMA), N-acetyl-S-propyl-L-cysteine (BPMA), N-acetyl-S-(2-hydroxypropyl)-L-cysteine (2-HPMA), 2,2′-thiodiacetic acid (TGA), N-acetyl--S-(2-hydroxyethyl)-L-cysteine (HEMA), phenylglyoxylic acid (PGA), s-benzylmercapturic acid (BMA), N-acetyl-S-(3-hydroxypropyl)-L-cysteine (3-HPMA), N-acetyl-S-(3-hydroxypropyl-1-methyl)-L-cysteine (HPMMA), hippuric acid (HA), 2-methylhippuric acid (2-MHA), 3-methylhippuric acid (3-MHA), 4-methylhippuric acid (4-MHA), N-acetyl-S-(2-carboxypropyl)-L-cysteine (CPMA), (R)−2-thioxothiazolidine-4-carboxylic acid (TTCA), rac 2-aminothiazoline-4--carboxylic acid (ATCA), mandelic acid (MA), cis-N-acetyl-S-(4-hydroxy-2-buten-1-yl)-L-cysteine (MHBMA3), N-acetyl-S-(1,2-dichloroethenyl)-L-cysteine (1,2-DCVMA), N-acetyl-S-(trichlorovinyl)-L-cysteine (TCVMA), and N-acetyl-S-(4-hydroxy-2-methyl-2-trans-buten-1-yl)-L-cysteine (IPMA3), were simultaneously determined using an ultra-high-performance liquid chromatography system (Vanquish Autosampler, Thermo Fisher Scientific, USA) coupled with electro spray tandem mass spectrometry (TSQ Quantis Triple Quadrupole system, Thermo Fisher Scientific, USA). All chemicals and reagents are listed in Text S1. The levels of 17 urinary MeTs (*i.e.*, vanadium, manganese, nickel, rubidium, molybdenum, cadmium, tin, antimony, copper, strontium, selenium, tellurium, cesium, lead, gallium, arsenic, and cobalt) were pretreated using the direct dilution method and determined using an inductively coupled plasma-mass spectrometer (Agilent 7800, USA) [26]. Urine samples were centrifugated at 13,000 rpm for 15 min and filtrated using a 0.45 μm Millipore filter (Welch Technology Company, Zhejiang Province, China). One mL of filtrate was diluted ten times with 2% nitric acid containing internal standards (*i.e.*, scandium, germanium, yttrium, indium, terbium, and bismuth) and subsequently used for instrumental analysis. The main detection parameters are provided in Text S2. In addition, the creatinine level in urine was determined using the Jaffe method [46] to adjust the effects of urine dilution.

2.3. Quality control and quality assurance

Internal standards were utilized to correct for potential matrix effects and process losses of the target analytes. Each analysis batch included water, matrix, and method blank samples to investigate potential background contamination from water, internal standards, and chemical reagents. No apparent background contamination interference was found for all analytes. The limit of detection (LOD) ranged 0.15–30.0 μg/L for mVOCs and from 0.002 to 0.363 μg/L for MeTs (Table S1). The linear correlation coefficients of the standard curve of all analytes were > 0.998 (Table S2). Two levels of spiked samples were used to evaluate the recovery and precision of the method. The average relative recoveries for all mVOCs and MeTs were within the range of 80.0–120%, except for PGA (70.6%) in low spiked samples. The relative standard deviations (RSD) for all mVOCs and MeTs were $< 15.0\%$ except for MA (15.4%), TGA (15.7%), TTCA (15.9%), and HEMA (16.1%). Eight percent of random samples were selected in each batch of samples to analyze in duplicate, and the results indicated that the coefficients of variation for the target analytes were $< 15\%$, except for HEMA.

2.4. Exposure risk assessment

The hazard quotient (HQ) and hazard index (HI) were employed to assess the non-CR of exposure to VOCs and MeTs. The HQ represents the health risk of a chemical, while the HI represents the total risk of multiple pollutants. A value of HQ or HI < 1.0 indicate no significant non-CR, and *vice versa* [27,36]. Biomonitoring equivalent (BE) values refer to the concentrations of chemicals or their metabolites in a biological medium that are consistent with existing health-based exposure guidance values [11]. If the BE values were established, the HQs were calculated as the ratio of the chemical concentration to the corresponding BE values (Eq. 1) [24]. When BE values were not available, the HQs were calculated as the ratio of the estimated daily intakes (EDIs, μg/kg-day) of pollutants to the reference doses (RfDs) or minimal risk levels (MRLs) [38]. The EDIs of MeTs and VOCs were calculated using Eq. 2 and Eq. 3, respectively [23].

The HI for benzene, toluene, ethylbenzene, xylene, and styrene (collectively referred to as BTEX) was calculated due to their similar mode of action and non-CR endpoints (Eq. 4) [30]. The CR of pollutants was calculated by multiplying the EDI of each chemical with its oral slope factor (SF) (Eq. 5). In addition to deterministic risk assessment, we also conducted the probabilistic risk assessment to avoid underestimating or overestimating the health risks. Deterministic risk assessment is a mathematical model in which outcomes are precisely determined without room for random variation, while probabilistic risk assessment includes both a deterministic component and a random error component. The parameters of our probabilistic risk assessment were as follows: Urinary mVOCs and MeTs were assumed to follow a lognormal distribution. N was modeled as a normally distributed variable. BE values, RfDs, MRLs, M_1, M_2, and SF were set as fixed values. Fue was set as a decision variable. We used Monte Carlo simulation with 10,000 iterations to estimate the risk using Crystal Ball.

$$HQ = \frac{C}{BE} \quad or \quad \frac{EDI}{\text{RfDs or MRLs}} \tag{1}$$

$$EDI_{metal} = \frac{C \times N}{F_{ue}} \tag{2}$$

$$EDI_{VOCs} = \frac{C \times N \times M_1}{M_2 \times F_{ue}} \tag{3}$$

$$HI_{BTEX} = HQ_{benzene} + HQ_{toleune} + HQ_{ethylbenzene} + HQ_{xylene} + HQ_{styrene} \tag{4}$$

$$CR = EDI \times SF \tag{5}$$

C represents the urinary concentration of pollutants (μg/g creatinine). N is the daily excretion of creatinine *via* urine and was assumed to be 0.015 g/kg-day for children [48]. F_{ue} is the urinary excretion factor of the pollutants. M_1 and M_2 are the molecular weights of the parent compounds and their corresponding metabolites (g/mol), respectively. The BE values were established to be 8.3, 2.0, 206, 26, and 123 μg/g creatinine for urinary As [11], Cd [14], Mo [13], Sn [35], and Se (when assuming a ratio of 0.73 g creatinine/L urine) [12,14], respectively. The Fue (Table S3), SF (Table S4), RfD (Table S5), and MRL values of VOCs and MeTs are presented in the Supplementary Materials. The non-CR of exposure to some VOCs and MeTs was not evaluated due to missing F_{ue} or RfD values. The urinary concentrations of MU, DHBMA, AAMA, AMCC, CYMA, BPMA, TGA, PGA, MA, BMA, 3-HPMA, MHA, ATCA, and TTCA were used to evaluate the EDIs of benzene, 1,3-butadiene, acrylamide, N,N-dimethylformamide, acrylonitrile, 1-bromopropane, 1,2-dichloroethane, ethylbenzene, styrene, toluene, acrolein, xylene, cyanide, and carbon disulfide, respectively.

2.5. Statistical analysis

All statistical analyses were performed using R version 3.6.0 (R Foundation for Statistical Computing, Vienna, Austria). We excluded 1,2-DCVMA, TCVMA, MHBMA3, and IPMA3 from statistical analysis due to their low detection frequencies. Measured values of urinary mVOCs and MeTs below the LOD were substituted with LOD/2 for statistical analyses. Due to the skewed distribution of all mVOC and MeT concentrations, the differences in chemical exposure levels between two subgroups and between three subgroups were examined by the Mann Whitney U and Kruskal-Wallis H tests, respectively. Spearman correlation analysis was employed to investigate the associations between urinary mVOCs and MeTs to reflect their similarities in exposure sources. We then investigated and visualized the differences in VOCs composition between ER and RA children using principal component analysis. Based on a machine learning method, we constructed a prediction model for distinguishing between ER and RA children using 14

VOCs and a support vector machine (SVM) classifier. We established the linear associations between exposure to VOCs and MeTs and oxidative DNA damage levels using a multiple linear regression model. Age, body mass index (BMI), sex, urinary creatinine, and OH-Cot were included in the model because previous studies have shown significant associations between these factors and VOCs or MeTs exposure [19,21,47,52]. To test for potential interaction effects between VOCs and MeTs exposure on oxidative DNA damage levels, we added a cross-product term of VOC × MeT to the multiple linear regression model. All original data were $\log_{10}$-transformed and standardized before being included in the model. Subsequently, we used the quantile g-computation (QG-C) and weighted quantile sum (WQS) regression models to evaluate the joint effects of exposure to VOCs and MeTs mixtures on oxidative DNA damage levels to address the effects of exposure mixtures. QG-C and WQS regression models are commonly used to reduce dimensionality and address collinearity issues raised by single pollutant models [18]. Additionally, we divided the children into passive smoking (OH-Cot > or = 2.00 μg/g creatinine) and non-passive smoking (OH-Cot < 2.00 μg/g creatinine) groups according to the urinary OH-Cot level established in our previous passive smoking simulation research [19].

3. Results and discussions

3.1. Exposure levels of VOCs and MeTs

Except for HEMA, TTCA, manganese, nickel, and antimony, the detection frequencies of all urinary mVOCs and MeTs were higher than 80%, indicating the widespread exposure of VOCs and MeTs in ER children (Table 1). The geometric mean (GM) concentrations of mVOCs varied from as low as 0.965 μg/g creatinine for HEMA to as high as 139026 μg/g creatinine for HA. Except for PGA, MA and BPMA, the median levels of the other 18 mVOCs in our ER children were approximately 1.29–9.76 times higher than those of children recruited from the suburbs of Guangzhou, China [22]. The GM concentrations of urinary TGA (101 *vs.* 762 μg/L), 3-&4-MHA (54.5 *vs.* 161 μg/L), BMA (3.11 *vs.* 7.69 μg/L), TTCA (8.8 *vs.* 18.5 μg/L), 3-HPMA (297 *vs.* 602 μg/L), AAMA (26.4 *vs.* 43.7 μg/L), CYMA (0.85 *vs.* 1.33 μg/L), and 2-MHA (23.7 *vs.* 35.7 μg/L) in the general children recruited from Wuhan and Shenzhen, China were lower than our results [41]. Similarly, the GM concentrations of urinary TTCA (24.5 *vs.* < LOD μg/g creatinine), 3-HPMA (799 *vs.* 463 μg/g creatinine), 2-MHA (47.4 *vs.* 18.5 μg/g creatinine), and 3-&4-MHA (213 *vs.* 180 μg/g creatinine) in our ER children were higher than those of general children aged 6 − 11 from the 2015 − 2016 National Health and Nutrition Examination Surveys (NHANES) in the United States [5]. Remarkably, the GM levels of MU (73.8 μg/g creatinine) and BMA (10.2 μg/g creatinine) in our study

Table 1
Urinary creatinine-adjusted (μg/g creatinine) and -unadjusted concentrations (μg/L) of oxidative DNA damage, volatile organic compound metabolites (mVOCs) and metals/metalloids (MeTs) in children from the e-waste recycling area (ER) and other reference areas (RA).

Analytes	DF	ER Children (Creatinine-adjusted)				ER Children (Creatinine-unadjusted)				RA Children
	(%)	GM	P5th	Median	P95th	GM	P5th	Median	P95th	Median
8-OHdG	100	4.56	2.39	4.48	8.96	3.44	1.22	3.55	8.82	2.66[a]
DHBMA	100	177	87.4	173	360	133	38.3	138	379	81.3[a]
MU	98.8	73.8	19.0	71.0	355	55.6	8.99	55.3	314	24.3[a]
AAMA	100	57.9	22.3	54.2	180	43.7	11.0	43.1	191	20.3[a]
AMCC	100	65.7	31.9	61.7	163	49.5	15.4	49.5	158	29.3[a]
CYMA	96.4	1.76	0.732	1.66	5.50	1.33	0.368	1.37	4.68	0.700[a]
BPMA	95.1	3.33	0.351	3.25	26.9	2.51	0.204	2.56	21.9	3.11[a]
2-HPMA	98.1	33.1	14.5	35.1	84.1	24.9	6.33	27.0	90.1	11.7[a]
TGA	100	1011	474	1002	2116	762	246	769	2069	276[a]
HEMA	79.0	0.965	0.143	1.10	4.73	0.729	0.0837	0.841	4.46	0.270[a]
PGA	99.9	167	60.8	160	515	124	33.0	122	445	243[a]
BMA	100	10.2	3.25	9.46	40.2	7.69	1.54	7.75	40.1	4.18[a]
3-HPMA	100	799	229	790	2864	602	118	625	2967	186[a]
HPMMA	100	244	127	236	501	184	57.2	187	541	150[a]
HA	100	139026	31304	132839	748186	104787	20405	102007	536369	68420[a]
2-MHA	100	47.4	14.8	47.7	141	35.7	8.46	35.3	157	12.2[a]
3-&4-MHA	100	213	65.3	196	969	161	32.3	151	918	46.2[a]
CPMA	100	282	122	265	797	212	58.2	210	718	138[a]
TTCA	78.2	24.5	2.52	24.4	271	18.5	1.47	18.9	221	2.50[a]
ATCA	99.3	240	84.0	243	685	181	56.4	184	522	188[a]
MA	99.7	360	96.7	297	1817	271	59.2	230	1842	1585[a]
Vanadium	100	28.9	9.88	28.8	87.1	21.7	8.88	22.5	46.7	0.630[b]
Manganese	72.3	1.55	0.247	1.41	13.2	1.17	0.368	1.20	5.35	1.25[b]
Cobalt	97.4	0.476	0.112	0.512	1.52	0.357	0.0610	0.385	1.28	0.340[b]
Nickel	78.8	5.32	1.35	5.44	18.9	4.00	1.04	4.53	15.2	2.91[b]
Copper	99.8	33.4	15.4	31.0	96.2	25.1	11.7	25.3	50.8	9.73[b]
Gallium	100	56.2	18.6	57.7	163	42.2	11.5	43.6	147	
Arsenic	100	93.2	36.9	89.5	239	70.0	26.2	66.8	193	21.2[b]
Selenium	99.8	12.1	6.17	11.4	24.6	9.09	2.96	9.43	23.6	25.5[b]
Rubidium	100	1208	547	1131	3407	908	335	868	2626	1703[b]
Strontium	99.8	162	35.0	183	474	121	25.0	128	488	101[b]
Molybdenum	100	86.0	32.5	84.6	224	64.6	17.7	67.5	223	79.6[c]
Cadmium	99.7	0.686	0.295	0.660	1.65	0.515	0.150	0.535	1.63	0.310[b]
Tin	100	1.19	0.403	1.06	4.42	0.892	0.275	0.880	3.18	1.22[c]
Antimony	58.6	0.109	0.0300	0.100	0.591	0.0821	0.0321	0.0800	0.379	0.0840[c]
Tellurium	83.7	0.137	0.0400	0.143	0.427	0.103	0.0258	0.115	0.320	
Cesium	100	8.90	4.65	8.73	18.4	6.69	2.63	7.00	15.4	6.52[b]
Lead	91.7	1.82	0.432	1.85	8.61	1.37	0.238	1.39	5.15	1.48[b]

Abbreviations: DF, detection frequency; GM, geometric mean; P, percentile. The median concentrations of mVOCs and MeTs in RA children from Guangzhou, China a [22], Hubei province, China b [51], and the United States c (children aged 6–11 in 2015–2016 NHANES) (CDC, 2021) are shown as μg/g creatinine.

were slightly higher than those of children recruited in Guiyu ER from 2016 to 2019 (MU: 61.6 μg/g creatinine; BMA: 5.35 μg/g creatinine) [21], suggesting that exposure levels of some VOCs in children have not significantly diminished despite prolonged e-waste control. A recent study [30] found that the median levels of all eight investigated urinary mVOCs in occupational workers in Guiyu e-waste recycling park were significantly higher than those in our ER children, including MU (224 *vs.* 71.0 μg/g creatinine), BMA (41.2 *vs.* 9.46 μg/g creatinine), HA (2.03×10^5 *vs.* 1.33×10^5 μg/g creatinine), PGA (1570 *vs.* 160 μg/g creatinine), MA (2760 *vs.* 297 μg/g creatinine), 2-MHA (304 *vs.* 47.7 μg/g creatinine), and 3-&4-MHA (902 *vs.* 196 μg/g creatinine), suggesting that the e-waste recycling park continues to release large amounts of VOCs. The results of the atmospheric VOCs monitoring study were in line with our findings. The average levels of atmospheric benzene (3.75 *vs.* 1.6 and 0.62 μg/m^3), 1,2-dichloroethane (2.5 *vs.* 0.95 and 0.93 μg/m^3), toluene (11.81 *vs.* 1.72 and 0.86 μg/m^3), o-xylene (2.73 *vs.* 0.37 and 0.26 μg/m^3), and m-&p-xylene (3.59 *vs.* 0.48 and 0.02 μg/m^3) in Guiyu ER were significantly higher than those in RAs, such as Nanjing and Xi'an, China [8,43]. Hence, children in ER are generally exposed to higher VOCs level than children in RA.

Urinary rubidium, strontium, arsenic, and molybdenum were found to be the most abundant MeTs in ER children. Compared with a previous study on urinary MeTs in Guiyu ER children recruited from 2016 to 2019, the GM levels of molybdenum (131 *vs.* 86.0 μg/g creatinine), nickel (6.34 *vs.* 5.32 μg/g creatinine), lead (5.23 *vs.* 1.82 μg/g creatinine), manganese (3.01 *vs.* 1.55 μg/g creatinine), tin (3.50 *vs.* 1.19 μg/g creatinine), and cobalt (0.598 *vs.* 0.476 μg/g creatinine) in our ER children were generally lower [21]. Similarly, the median concentrations of urinary lead (6.89 *vs.* 1.39 μg/L) and nickel (9.07 *vs.* 4.53 μg/L) in our ER children were approximately 80% and 50% lower than those in e-waste recyclers from an unregulated e-waste recycling site in Agbogbloshie, Ghana, respectively [44]. The measures on strengthening environmental infrastructure construction and environmental restoration in the e-waste recycling park greatly reduced the occurrences of exposure to MeTs in children [32]. Formal e-waste control appears to prevent the diffusion of MeTs from the centralized industrial park to the surrounding residential areas [50]. However, compared with children from other RAs, the levels of some MeTs such as vanadium, arsenic, copper, cadmium, nickel, strontium, and cobalt in our ER children were still higher [5,51]. These results jointly manifest that ER children still have relatively high exposure levels of VOCs and MeTs in spite of e-waste control.

3.2. Sources, composition characteristics, and potential diagnostic indexes

The correlations between VOCs and MeTs exposure are shown in Fig. S1. Significant positive correlations were observed in 169 out of 190 pairs of urinary mVOC ($r = 0.080 - 0.590$, $p < 0.05$, with the highest correlation observed between MA and PGA), 131 out of 136 pairs of urinary MeTs ($r = 0.078 - 0.711$, $p < 0.05$, with the highest correlation observed between copper and vanadium), and 225 out of 340 pairs of urinary mVOCs and MeTs ($r = 0.078 - 0.486$, $p < 0.05$, with the highest correlation observed between ATCA and cesium). Similar correlations have been reported previously [21,24]. These results indicate that ER children may have similar sources of exposure to VOCs and MeTs, which may be explained by the fact that emissions of VOCs and MeTs from the e-waste recycling processes into the atmosphere significantly influence the surrounding residential areas despite implementing e-waste control [30,50]. Large amounts of BTEX can be generated during the recycling processes of e-waste [28]. A recent study showed that the composition profiles of atmospheric BTEX followed the order of toluene (42.9%) > m-&p-xylene (22.7%) > benzene (14.4%) > ethylbenzene (10.0%) > o-xylene (9.93%) in the e-waste dismantling park [30], which is similar with the exposure patterns of our ER children (toluene (43.2%) > ethylbenzene (27.6%) > benzene (13.9%) > m-&p-xylene (11.3%) > o-xylene (4.03%)) (Fig. 1 A). Hence, atmospheric VOCs discharged from the e-waste recycling park may be an important source of VOCs exposure in ER children.

The sum EDIs of 14 VOCs in ER children ($\sum_{14VOCs} = 65.5$ μg/kg-day) were approximately twice as high as those in RA children ($\sum_{14VOCs} = 31.7$ μg/kg-day) (Fig. 1 A). Except for ethylbenzene, the exposure levels of all VOCs were higher in ER children than in RA children. Furthermore, the composition profiles of VOCs exposure were distinctive between ER and RA children. Ethylbenzene was the predominant component in RA children, accounting for 34.6% of VOCs intake, followed by toluene (21.5%), 1,2-dichloroethane (12.3%), cyanide (8.74%), and acrolein (7.77%). In comparison, we observed a decreasing order of exposure to 1,2-dichloroethane (22.5%), toluene (19.3%), acrolein (16.0%), ethylbenzene (12.3%), and benzene (6.20%) in ER children. Principal component analysis also revealed significant differences in VOC exposure profiles between ER and RA children (Fig. 1B). PC1, PC2, and PC3 accounted for 19.8%, 12.4%, and 9.22% of the total variances, respectively. The VOC exposure profiles between ER and RA children exhibited the largest difference in the direction of PC2, which was mainly contributed by m-&p-xylene and o-xylene (Table S6).

Given the striking discrepancies in exposure level and composition of VOCs between two groups of children, we further explored the feasibility of developing diagnostic indexes for distinguishing e-waste pollution (ER children) from non-e-waste pollution (RA children). Based on the SVM classifier, we found that including 14 VOCs simultaneously in the model resulted in excellent classification accuracy (ACC) in distinguishing between ER and RA children. The model accurately predicted 94.2% of RA children and 99.7% of ER children, corresponding to a total ACC of 99.2% (Fig. 1 C). One step further, we screened out specific VOCs or VOC ratios that accurately differentiated ER and RA children. The distribution differences of 1,2-dichloroethane (Median: 14.2 *vs.* 4.02 μg/kg-day) and the ratio of 1,2-dichloroethane/ethylbenzene (Median: 1.71 *vs.* 0.402) were most pronounced between ER and RA children (Fig. 1D). The P5th values of 1,2-dichloroethane and the 1,2-dichloroethane/ethylbenzene ratio in ER children were 6.86 μg/kg-day and 0.574, respectively, approaching to the P80th values of RA children (6.39 μg/kg-day for 1,2-dichloroethane and 0.535 for 1,2-dichloroethane/ethylbenzene ratio). The low overlap between ER and RA children manifests that the combined use of 1,2-dichloroethane and the 1,2-dichloroethane/ethylbenzene ratio can effectively differentiate e-waste pollution from non-e-waste pollution (Fig. 1E). Subsequently, we verified whether these two diagnostic indexes alone could still distinguish between ER and RA children. Children who meet both conditions, where 1,2-dichloroethane/ethylbenzene > 0.574 (*i.e.*, TGA/PGA > 1.90) and 1,2-dichloroethane > 6.86 μg/kg-day (*i.e.*, TGA > 471 μg/g creatinine), would be predicted as ER children. Otherwise, the results would be RA children. Consequently, using these two diagnostic indexes could accurately classify 97.1% of RA children and 90.5% of ER children, corresponding to a total ACC of 91.4% (Fig. 1 F). Hence, 1,2-dichloroethane and the 1,2-dichloroethane/ethylbenzene ratio may be used as diagnostic indexes to identify e-waste pollution. A previous study on source profiles in the e-waste recycling park indicated that 1,2-dichloroethane and BTEX discharged massively during e-waste dismantling are characteristic factors for identifying the e-waste pollution source [6,28]. The biomonitoring results of urinary mVOCs in two populations of children recruited in RA supported our findings. For example, the GM concentrations of urinary TGA and PGA were 337 and 67.3 μg/g creatinine, respectively, in asthmatic children in Guangzhou (N = 252) [22], and 101 and 93.9 μg/g creatinine, respectively, in general children in China (N = 390) [41]. The calculation results manifested that asthmatic children (TGA/PGA > 1.90 but TGA < 471 μg/g creatinine) and general children (TGA/PGA < 1.90 and TGA < 471 μg/g creatinine) did not meet both conditions simultaneously, *i.e.*, TGA/PGA > 1.90 and TGA > 471 μg/g creatinine, and thus should be classified as RA children. Note that urinary TGA levels may be significantly affected by passive smoking in children [22]. Children exposed to secondhand smoke may have urinary TGA levels several times higher than those

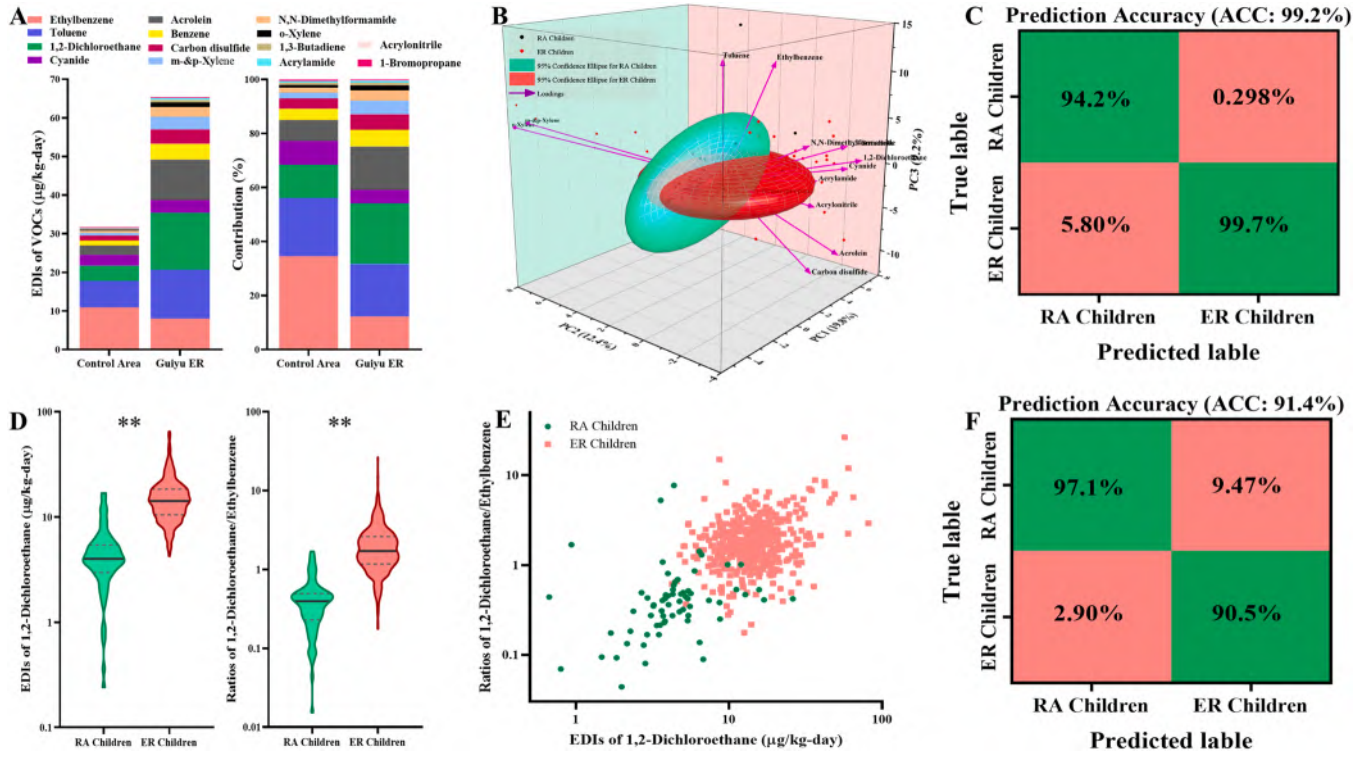

Fig. 1. The exposure composition characteristic of VOCs in ER children and its application in predicting the presence of e-waste pollution. We compared the composition characteristic of VOCs in ER children with our previously reported children recruited in RA [22]. Fig. 1 A shows the composition profiles of estimated daily intakes (EDIs) of 14 VOCs in ER and RA children. The differences in VOCs composition between ER and RA children were investigated and visualized using principal component analysis (Fig. 1B). Fig. 1 C shows the prediction accuracies (ACC) of the 14 VOCs in distinguishing between ER and RA children based on a support vector machine classifier. Fig. 1D and Fig. 1E show the distribution differences between ER and RA children in 1,2-dichloroethane/ethylbenzene ratios and 1, 2-dichloroethane levels. * * $p < 0.01$. Fig. 1 F shows the prediction accuracies of distinguishing ER and RA children using 1,2-dichloroethane/ethylbenzene ratios and 1,2-dichloroethane.

without exposure [19,22]. The use of children with severe secondhand smoke exposure may pose a high risk of misdiagnosis. Therefore, when using 1,2-dichloroethane and the 1,2-dichloroethane/ethylbenzene ratio as diagnostic indexes to identify e-waste pollution, the simultaneous determination of OH-Cot to rule out the influence of secondhand smoke is of great necessity.

3.3. Associations of VOCs and MeTs exposure with oxidative DNA damage

After adjusting for confounders, the urinary levels of 18 mVOCs and 13 MeTs were significantly associated with 8-OHdG in the multiple linear regression model ($p < 0.05$) (Fig. 2 A). The estimated z-score changes in 8-OHdG level for one SD increase in $\log_{10}$-transformed mVOC or MeT concentrations ranged from 0.076 to 0.221 (Table S7). Since co-exposure to VOC and MeT mixtures frequently occurs in daily life, we further used QG-C and WQS regression analyses to address the effects of exposure mixtures. Each additional joint exposure quantile increases in the 18 mVOC mixtures and 13 MeT mixtures was significantly associated with a 0.779 and 0.582 increase in 8-OHdG level in the QG-C analysis, respectively (Fig. 2B). In line with the QG-C analysis, each one-unit increase in the mVOC mixtures and MeT mixtures was significantly associated with a 0.472 (95% CIs: 0.290, 0.654) and 0.221 (95% CIs: 0.045, 0.398) increase in 8-OHdG levels in the WQS regression analysis, respectively. Hence, exposing to VOC or MeT mixtures may pose high risks of oxidative DNA damage. For MeT mixtures alone, rubidium (21.9%), arsenic (19.2%), lead (18.6%), copper (12.1%), cesium (5.92%), and cadmium (5.16%) contributed to approximately 82.9% of the joint effect on 8-OHdG (Fig. 2 C). For mVOC mixtures alone, approximately 79.3% of the joint effect on 8-OHdG was ascribed to MA (23.9%), AMCC (15.9%), 2-MHA (15.6), TGA (13.1%), HA (5.74%), and MU (5.13%) (Fig. 2D). Thus, exposure to rubidium, arsenic, styrene, and N,N-dimethylformamide appears to be the main contributor to increased oxidative DNA damage levels in children. Interestingly, rubidium and arsenic were the most abundant urinary MeTs in ER children (Table 1). Arsenic has been considered a carcinogen, and its possible toxic mechanism to cause the development of carcinogenicity may involve the induction of oxidative stress and DNA damage [3]. Rubidium is a rare earth element found in e-waste [10]. However, previous studies have seldom reported on the risk of oxidative DNA damage from rubidium exposure. Styrene and N,N-Dimethylformamide have been classified as group 2 A carcinogens by the International Agency for Research on Cancer [17]. The metabolism of styrene involves the cytochrome P450 enzyme-mediated conversion of styrene to its active metabolite styrene-7,8-oxide through oxidation, while styrene-7,8-oxide can covalently bind to biomacromolecules such as DNA and is thought to be directly responsible for the genotoxic effects of styrene [37]. The median level of MA, a urinary metabolite of exposure to styrene, was found to be as high as 2760 μg/g creatinine in occupational e-waste recycling workers [30], which is far higher than the median level found in our ER children (297 μg/g creatinine) and the general population (150 μg/g creatinine for children aged 6 −11 and 134 μg/g creatinine for adults) from the NHANES 2015 − 2016 research [5]. Thus, the risk of oxidative DNA damage from exposure to styrene in ER populations, especially occupational e-waste recycling workers, should be of significant concern.

In the QG-C analysis, each additional joint exposure quantile increase in the mVOCs & MeTs mixtures was only associated with a 0.980 increase in 8-OHdG levels, which is lower than the total additive effect (1.361) of the mVOC mixtures (0.779) and MeT mixtures (0.582). This

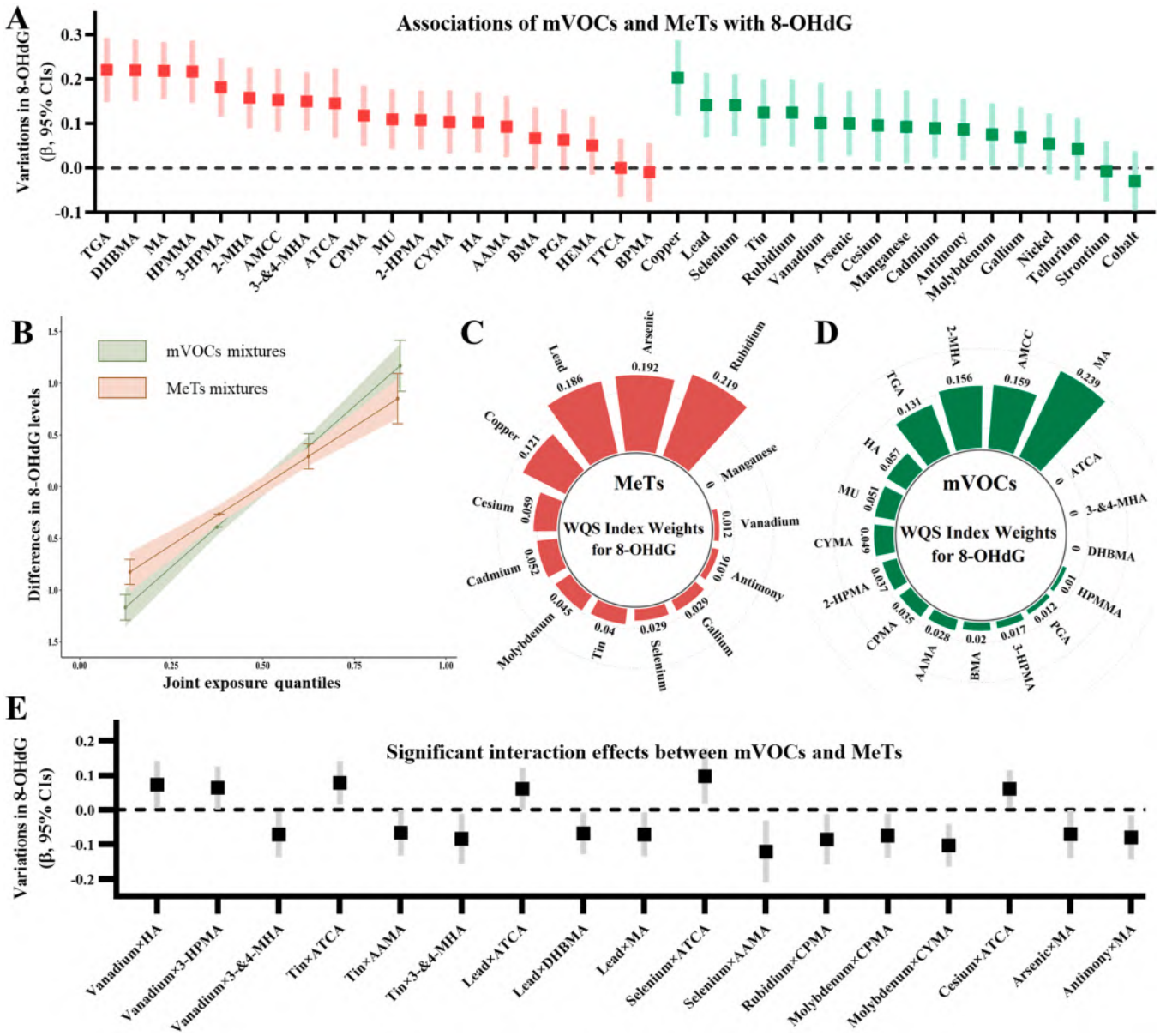

Fig. 2. The associations of VOCs and MeTs exposure with oxidative DNA damage levels. All original data were log_{10}-transformed and standardized before including in the model. In Fig. 2 A, estimates were calculated using a multiple linear regression model and adjusted for age, BMI, sex, urinary creatinine, and OH-Cot. The results were shown as variations (β, 95% CIs) in 8-OHdG associated with each one SD increase in log_{10}-transformed urinary pollutant levels. In addition, we used the quantile g-computation (QG-C) (Fig. 2B) and weighted quantile sum (WQS) (Fig. 2 C; Fig. 2D) regression models to evaluate the joint effects of exposure to VOC mixtures or MeT mixtures on oxidative DNA damage levels. The red and green bars represent the joint effects of exposure to multiple MeTs and VOCs on 8-OHdG, respectively. All significant interactive effects between mVOCs and MeTs on 8-OHdG are shown in Fig. 1E.

phenomenon may be explained by the interactive effects between VOCs and MeTs in reducing oxidative DNA damage risks. As shown in Fig. 2E, we found significant antagonistic effects between MeTs and VOCs on 8-OHdG levels ($p < 0.05$), such as antimony × MA (β: −0.079; 95% CIs: −0.144, −0.015), molybdenum × CYMA (β: −0.102; 95% CIs: −0.164, −0.041), molybdenum × CPMA (β: −0.075; 95% CIs: −0.138, −0.011), rubidium × CPMA (β: −0.085; 95% CIs: −0.159, −0.012), selenium × AAMA (β: −0.121; 95% CIs: −0.211, −0.030), lead × MA (β: −0.071; 95% CIs: −0.135, −0.007), lead × DHBMA (β: −0.068; 95% CIs: −0.128, −0.008), tin × 3-&4-MHA (β: −0.084; 95% CIs: −0.156, −0.011), and vanadium × 3-&4-MHA (β: −0.071; 95% CIs: −0.137, −0.006). Furthermore, we also observed significant synergistic effects of cesium × ATCA, selenium × ATCA, lead × ATCA, tin × ATCA, vanadium × 3-HPMA, and vanadium × HA on 8-OHdG levels ($p < 0.05$). Therefore, accurately assessing the population risks posed by combined exposure to VOCs and MeTs is challenging due to their joint exposure effect and interactive effect.

3.4. Carcinogenic and non-carcinogenic risks of exposure to VOCs and MeTs

Summary statistics for the probabilistic risk assessment of exposure to VOCs and MeTs in ER children are presented in Table 2. Of the 15 VOCs and 12 MeTs assessed, three VOCs and four MeTs had HQ values greater than or slightly lower than one at the median, namely acrolein (2.35), benzene (0.960), 1,3-butadiene (0.792), arsenic (9.78), vanadium (2.48), copper (1.19), and lead (0.788). The HI for BTEX (1.48) also exceeded one at the median. Exceeding 30.0% of our ER children may be faced with non-CR due to exposure to acrolein (82.5%), benzene (48.4%), 1,3-butadiene (32.7%), BTEX (72.1%), arsenic (100%),

Table 2
Probabilistic non-carcinogenic (non-CR) and carcinogenic risk (CR) assessment for VOCs and MeTs exposure using Monte-Carlo simulation.

Risk Types	Parent Chemicals	Biomarkers	Non-carcinogenic and Carcinogenic Risk Levels		
			Mean	Median	95% CIs
HQ	Acrolein	3-HPMA	3.64	2.35	0.419 – 14.4
HQ	Benzene	MU	1.55	0.960	0.151 – 6.58
HQ	1,3-Butadiene	DHBMA	0.918	0.792	0.279 – 2.30
HQ	Acrylamide	AAMA	0.334	0.234	0.0660 – 1.22
HQ	N,N-Dimethylformamide	AMCC	0.270	0.190	0.0567 – 0.992
HQ	Toluene	BMA	0.219	0.135	0.0298 – 0.931
HQ	Styrene	MA	0.173	0.0909	0.0100 – 0.838
HQ	1,2-Dichloroethane	TGA	0.0827	0.0708	0.0259 – 0.211
HQ	Cyanide	ATCA	0.0814	0.0630	0.0149 – 0.250
HQ	Acrylonitrile	CYMA	0.0537	0.0338	0.00548 – 0.216
HQ	Ethylbenzene	PGA	0.0515	0.0380	0.00798 – 0.173
HQ	Carbon disulfide	TTCA	0.110	0.0477	0.00465 – 0.589
HQ	*m*-&*p*-Xylene	3-&4-MHA	0.0263	0.0127	0.00168 – 0.136
HQ ($\times10^{-2}$)	*o*-Xylene	2-MHA	0.865	0.313	0.0518 – 5.23
HQ ($\times10^{-2}$)	1-Bromopropane	BPMA	0.307	0.138	0.0120 – 1.58
HI	BTEX	HI-BTEX	2.00	1.48	0.453 – 6.55
HQ	Arsenic		13.6	9.78	3.36 – 46.6
HQ	Vanadium		3.17	2.48	0.615 – 9.64
HQ	Copper		1.66	1.19	0.293 – 5.73
HQ	Lead		1.26	0.788	0.139 – 5.22
HQ	Molybdenum		0.487	0.402	0.157 – 1.33
HQ	Cadmium		0.391	0.322	0.113 – 1.09
HQ	Manganese		0.535	0.118	0.0137 – 3.34
HQ	Nickel		0.142	0.0675	0.0139 – 0.705
HQ	Selenium		0.108	0.0895	0.0396 – 0.281
HQ	Tin		0.0624	0.0383	0.0105 – 0.261
HQ	Cobalt		0.0481	0.0370	0.00825 – 0.155
HQ ($\times10^{-2}$)	Antimony		1.21	0.628	0.128 – 5.93
CR ($\times10^{-3}$)	Arsenic		4.22	2.90	0.810 – 15.7
CR ($\times10^{-3}$)	1,2-Dichloroethane	TGA	1.49	1.27	0.458 – 3.73
CR ($\times10^{-4}$)	Acrylamide	AAMA	3.36	2.37	0.655 – 12.1
CR ($\times10^{-5}$)	Benzene	MU	9.45	5.83	0.909 – 39.8
CR ($\times10^{-5}$)	Acrylonitrile	CYMA	2.08	1.34	0.208 – 8.33

vanadium (90.1%), copper (59.3%), and lead (40.6%) (Fig. 3A). Slight exceedances of non-CR were also observed in ER children due to exposure to acrylamide (4.09%), N,N-dimethylformamide (2.40%), toluene (2.07%), styrene (1.71%), carbon disulfide (0.83%), acrylonitrile (0.04%), manganese (10.7%), molybdenum (6.14%), cadmium (3.25%), nickel (1.24%), and tin (0.04%). Exposure to other VOCs and MeTs was not likely to cause non-CR. In addition, the CR of arsenic (2.90×10^{-3}), 1,2-dichloroethane (1.27×10^{-3}), acrylamide (2.37×10^{-4}), benzene (5.83×10^{-5}), and acrylonitrile (1.34×10^{-5}) at the median were above 1.0×10^{-6} considered to be a non-negligible risk [9]. Nearly 100% of ER children may be at risk of cancer due to exposure to these five chemicals (Fig. 3B). These findings were in alignment with the results of deterministic risk assessment (Table S8), indicating the robustness of our assessment for chemical exposure risks. Taken together, exposure to some environmental pollutants may continue to pose considerable health concerns for ER children even after e-waste control are implemented, including acrolein, benzene, 1,3-butadiene, arsenic, 1,2-dichloroethane, acrylamide, acrylonitrile, vanadium, copper, lead, and BTEX. It is important to prioritize further investigation of these hazardous chemicals, including their levels, sources, and pathways of exposure, and to manage them effectively at e-waste recycling sites in order to mitigate corresponding exposure risks.The non-CR and CR assessment results of VOCs in our ER children are partly consistent with previous studies evaluated using atmospheric VOCs data. Inhalation is the

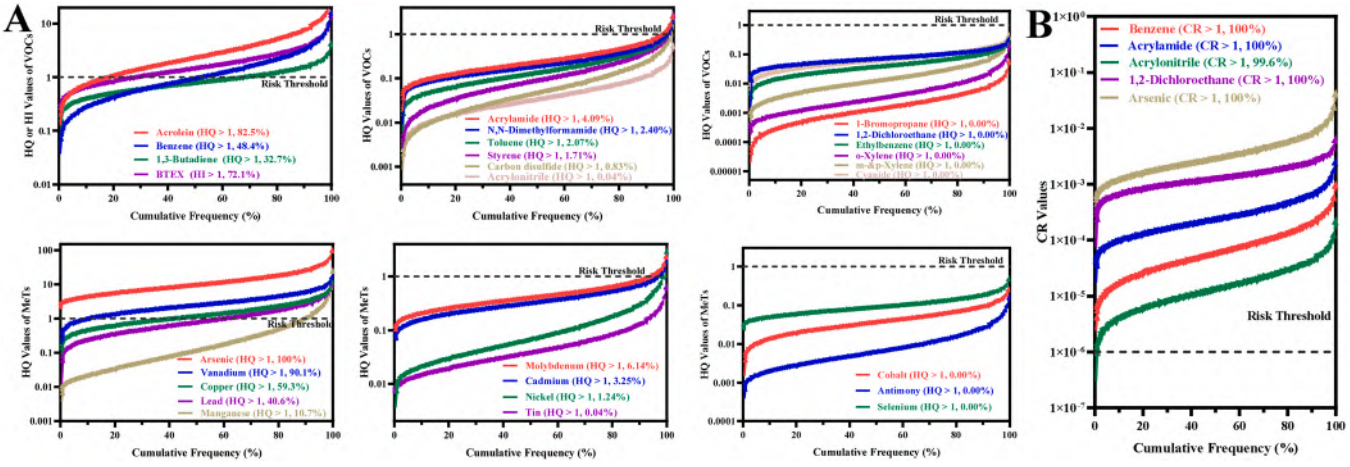

Fig. 3. Probabilistic distribution of non-CR (A) and CR (B) of VOCs and MeTs exposure using Monte-Carlo simulation. The risk threshold for HQ/HI and CR are 1.0 and 1×10^{-6}, respectively.

predominant route for human exposure to VOCs. A study conducted in Guiyu ER estimated the CR through inhalation of ambient VOCs and found that the lifetime CR of benzene, 1,2-dichloroethane, and 1,3-butadiene in the e-waste recycling park and residential areas were larger than 1.0×10^{-6}, posing possible CR to the surrounding populations [6]. These external exposure assessment findings coincide with our internal exposure assessment results. However, the HQ values of all 24 VOCs investigated, including benzene, 1,3-butadiene, and BTEX that exhibited high non-CR to our ER children, were below the risk threshold in their research. A possible explanation of such inconsistency is that relying on inhalation exposure pathway alone to estimate the VOC exposure risks may cause underestimation because ingestion and dermal uptake are also known pathways of VOCs exposure in addition to inhalation intake. Recently, Liu et al. [30] conducted a probabilistic risk assessment for BTEX in Guiyu ER and indicated that the inhalation of atmospheric benzene exhibited an unacceptable CR (Median: 1.22×10^{-6}) [30]. Their internal exposure risk assessment based on a benzene exposure biomarker (N-acetyl-S-phenyl-L-cysteine) also implied that 13.7% and 99.9% of the occupational e-waste recycling workers exceeded the definite CR (1.0×10^{-4}) and the possible CR (1.0×10^{-6}), respectively, which is relatively lower than the CR of benzene in our ER children (CR: 29.3% > 1.0×10^{-4}; 100% > 1.0×10^{-6}). This may be explained by two reasons. First, MU may result from ingesting sorbic acid, a common food preservative [34]. Our study used urinary MU to estimate the exposures and CR of benzene, which is less specific than N-acetyl--S-phenyl-L-cysteine in evaluating benzene exposures. Thus, the CR of benzene in our ER children may be overestimated. Second, the metabolic rate of children is higher than that of adults due to their larger surface-to-volume ratio, resulting in higher oxygen consumption and exposure to air pollutants [4].

Interestingly, exposure to arsenic consistently posed the highest non-CR and CR to our ER children. The urinary concentrations of total arsenic (GM: 93.2 μg/g creatinine or 70.0 μg/L; median: 89.5 μg/g creatinine or 66.8 μg/L) in our ER children were significantly higher than those in populations from RAs, such as the general children in China (median: 21.2 μg/g creatinine) [51], the national populations in Korea (median of adults aged 20 −29: 37.2 μg/L) [25], the general children in Mexico (median: 13.2 μg/L) [2], the national populations in Canada (median: 4.8 μg/L) [9] and the United States (median: 5.56 μg/g creatinine) [5], and are comparable to those found in an adult population in Bangladesh affected by arsenic in drinking water (GM of males: 79.8 μg/L) [39]. Note that the high exposure to arsenic in our ER children may not be solely a result of e-waste recycling activities. Because the background exposure to arsenic in the Chinese population is generally higher than in developed countries [5,31]. E-waste recycling sites worldwide are commonly contaminated with arsenic, and large amounts of arsenic have been detected in soil, water, and sediment around these sites [15]. A previous study found that the residues of MeTs in an abandoned e-waste recycling site still posed ecological risks [49] The use of pond water for irrigation resulted in considerable MeTs contamination to the paddy soil and crops, which could increase the risk of exposure to MeTs *via* dietary intakes [49]. In addition to dietary exposure, inhalation exposure to arsenic cannot be ignored. A recent study highlighted that the CR of lifetime exposure to arsenic through inhalation of atmospheric $PM_{2.5}$ in a formal e-waste recycling park and its surrounding residential areas still exceeded the acceptable risk level even after e-waste control [50]. Hence, remediating contaminated soil and water and controlling arsenic emissions into the atmosphere should be continuously implemented to prevent the dissemination of arsenic and corresponding population health risk.

3.5. Mitigation measures of exposure risks of VOCs and MeTs

In view of the probable non-CR and CR of exposure to arsenic, vanadium, copper, lead, molybdenum, cadmium, manganese, acrolein, benzene, 1,3-butadiene, acrylamide, N,N-dimethyl-formamide, toluene, BTEX, acrylonitrile, and 1,2-dichloroethane in ER children (Fig. 3), we attempted to propose measures to mitigate the risks of exposure to these 16 pollutants. To identify potential factors influencing the exposure risks of these hazardous chemicals, we established the associations of demographic characteristics, diets, and lifestyles with non-CR and CR of these 16 pollutants in ER children. The results showed that the effects of BMI, maternal education levels, handwashing before meals, seafood consumption frequency, household drinking water type, and fuel for cooking on the exposure risks of most VOCs and MeTs were not profound (Table S9; Fig. 4 A). In contrast, age, gender, daily exercise time, and passive smoking status were significantly associated with the exposure risks of most hazardous pollutants. Compared with children aged older (7 −11 years), children of younger ages (≤ 6 years) had significantly higher exposure risks of all 16 hazardous pollutants. Furthermore, the 8-

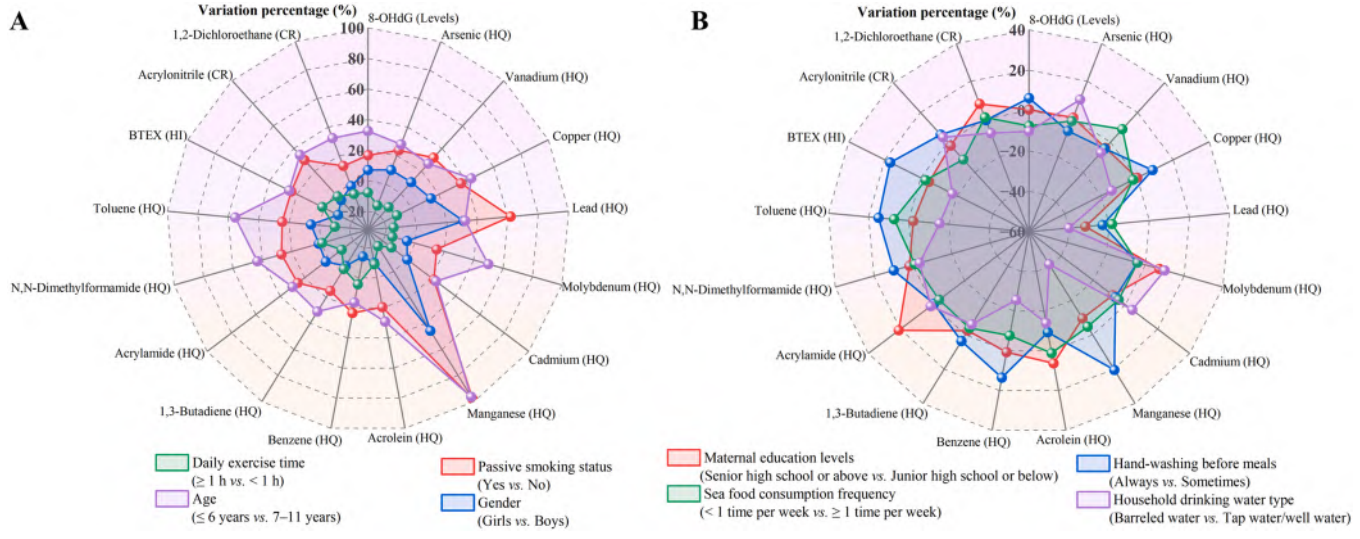

Fig. 4. Demographics-, diets-, and lifestyles-related differences in VOC and MeT exposure risks. Fig. 4 A shows the variation percentages of 8-OHdG level and non-CR or CR of exposure to 16 VOCs and MeTs in ER children between subgroups of age, daily exercise time, passive smoking status, and gender. Fig. 4B shows the variation percentages of 8-OHdG level and non-CR or CR of exposure to 16 VOCs and MeTs in ER children between subgroups of maternal education levels, sea food consumption frequency, hand-washing before meals, and household drinking water type. Taking passive smoking as an example, the variation percentages of exposure risk of each chemical in ER children were calculated as $\frac{GM_{passive\ smoking\ children} - GM_{non-passive\ smoking\ children}}{GM_{non-passive\ smoking\ children}} \times 100\%$.

OHdG level in children of younger ages was also approximately 32.6% higher, which is consistent with previous finding that younger children are more vulnerable to e-waste pollution [21]. The susceptibility of young children to e-waste pollutants is likely due to their high pollutant exposure amount per unit of body weight, immature metabolic detoxification system, and high tissue vulnerability [4]. The intensity of the influence of passive smoking status on the exposure risks of hazardous pollutants was similar to that of age. The exposure risks of all 16 hazardous pollutants in passive smoking children were significantly higher than in children without passive smoking, with elevated percentages ranging from 12.8% for 1,2-dichloroethane to 98.1% for manganese (Fig. 4 A). Moreover, the 8-OHdG level in passive smoking children was also approximately 16.7% higher. In accordance with a previous experiment simulating secondhand smoke exposure, urinary concentrations of exposure biomarkers of benzene, acrylonitrile, 1-bromopropane, toluene, and cyanide, and 8-OHdG levels in children significantly increased by 60.2%, 57.1%, 429%, 101%, 61.1%, and 24.6%, respectively, after one hour of passive smoking [19]. In addition, we observed gender-dependent differences in MeTs exposure risk in our ER children. The exposure risks of arsenic, vanadium, copper, lead, and manganese in girls were more pronounced than in boys, which may be due to the differences in the absorption and metabolism of MeTs between girls and boys. For example, according to the Exposure Factors Handbook of Chinese Population (6–17 years), the likelihood and average time of hand-mouth contact behavior, a known exposure route of MeTs, are generally higher in girls than in boys [54]. Animal studies have found that the expression of metallothionein, a metal-binding protein that protects against heavy metal toxicity, is higher in female mice than in male mice [53].

Interestingly, enhancing daily physical exercise may be beneficial for attenuating the exposure risks of VOCs and MeTs in ER children (Fig. 4 A). The non-CR or CR of exposure to acrylamide, arsenic, vanadium, copper, lead, molybdenum, and cadmium in children with daily physical exercise time ≥ 1 h diminished by 10.4%, 15.0%, 11.9%, 10.8%, 15.1%, 15.4%, and 12.8%, respectively, compared with children with daily physical exercise time < 1 h (Table S9). Furthermore, the 8-OHdG level in children with daily physical exercise time ≥ 1 h was also approximately 7.66% lower. Moderate exercise significantly increases individual water intake and oxygen consumption, and thus promoting the rapid metabolism and elimination of hazardous chemicals and reducing human exposure risks. Besides, physical exercise has acute and chronic effects on inflammatory balance, metabolic regulation, and redox status, which can be useful against disease conditions such as obesity, diabetes mellitus, and exposure to atmospheric pollutants [7]. We also found that children who drank barreled water, had higher maternal education levels (senior high school or above), higher frequencies of handwashing before meals (always), and fewer frequencies of seafood consumption ($<$ 1 time per week) had significantly lower non-CR of lead exposure (Fig. 4B). On the one hand, mothers with high education levels are likely to make a conscious effort to avoid behaviors that increase their children's exposure to chemicals, such as hand-mouth behavior. On the other hand, air, soil, groundwater, and seafood in ER may be contaminated by MeTs [49]. Thus, keeping handwashing before meals, substituting tap water/well water with barreled water, and reducing seafood consumption may mitigate lead exposure *via* dust and dietary intake. In general, strengthening daily physical exercise and controlling dietary habits facilitate mitigating the exposure risks and maximizing the health benefits of ER children. Furthermore, girls and young children are vulnerable to e-waste pollution and should be given priority protection or special preventive measures.

3.6. Strengths and limitations

This research presents four main strengths. First, our study is the first to reveal reference values for VOC and MeT exposure levels and risks in the susceptible population living around the e-waste recycling sites through biomonitoring a large panel of urinary exposure biomarkers and recruiting the largest group of ER children to date [33]. Second, we combinedly utilized the deterministic and probabilistic risk assessment models as well as the single and mixed pollutant exposure models to investigate the non-CR and CR of exposure to VOCs and MeTs and their associated oxidative DNA damage risk, presenting a comprehensive view of the exposure risks faced by ER children after e-waste control. The corresponding findings will underpin risk managers' decision-making on enacting priority control chemicals and taking targeted actions to mitigate the population exposure risks. Third, according to the characteristic profiles of VOCs exposure concentration and composition in ER children, we constructed models for distinguishing between ER and RA using the SVM classifiers, and two diagnostic indexes for identifying e-waste pollution were proposed for the first time. Fourth, our study not only revealed the chemical exposure risks faced by ER children, but also recognized key factors affecting VOCs and MeTs exposure risk based on a questionnaire survey, which provides some references for intervening measures to mitigate the exposure risks of VOCs and MeTs in ER children.

Some limitations of our study are inherent to the specificity of the VOC exposure biomarkers and some parameters for risk assessment. First, some urinary exposure biomarkers of VOCs, such as MU, may be affected by dietary intake [34]. Urinary BMA is a biomarker of both toluene and benzyl chloride [40]. Low biomarker specificity may result in overestimating the exposure risks. Additionally, Fue and RfD values may introduce some uncertainties in accurate risk assessment due to the lack of experimental data, pharmacological models, and understanding of the mode of action for some VOCs and MeTs when deriving these values [9]. To address these limitations, we utilized the associations between VOCs/MeTs exposure and oxidative DNA damage as supplement evidence to verify the exposure risks of these chemicals. Furthermore, we introduced the probabilistic risk assessment model and set the Fue values as decision variables, with confidence intervals ranging from $<$ 20% to $>$ 20% of the Fue values. In addition, the absence of Fue or RfD values for some chemicals (*e.g.*, strontium and cesium) rendered them impossible for risk assessment, which may cause an underestimation of the aggregate risks. This highlights the limited ability to monitor and interpret environmental chemicals in a health risk context. In this study, we found that it was plausible to use 8-OHdG levels to interpret the exposure risks posed by mixed pollutant exposure in ER, providing a promising way for aggregate risk assessment for mixed pollutant exposure.

4. Conclusions and Implications

The ER children are still being exposed to high levels of VOCs and MeTs despite a decade-long journey of e-waste control. Exposure to VOCs and MeTs is significantly and positively associated with oxidative DNA damage in single and mixed pollutant exposure models, while the antagonistic effect between VOCs and MeTs exposure may attenuate their associated oxidative DNA damage risks. Specifically, exposure to acrolein, benzene, 1,3-butadiene, 1,2-dichloroethane, acrylamide, acrylonitrile, arsenic, vanadium, copper, and lead poses considerable CR or non-CR to children, suggesting that these hazardous VOCs or MeTs should be controlled in priority in regulated e-waste recycling areas to maximize the health benefits of the populations. Furthermore, changing personal lifestyles, such as enhancing daily physical exercise, avoiding passive smoking, and keeping handwashing before meals, may mitigate the exposure risks of these hazardous chemicals. Notably, for the first time, our study manifests that the 1,2-dichloroethane/ethylbenzene ratio ($>$ 0.574; or TGA/PGA $>$ 1.90) and 1,2-dichloroethane ($>$ 6.86 μg/kg-day; or TGA $>$ 471 μg/g creatinine) can serve as diagnostic indexes for identifying e-waste pollution. In general, our study provides an overview of the exposure risks of VOCs and MeTs faced by children in a typical regulated ER. Targeted actions on mitigating or eliminating related exposure sources to reduce associated health risks are of great necessity.

Environmental implications

This study profiled the exposure composition and risk of VOCs and MeTs in children from a regulated ER by biomonitoring urinary exposure biomarkers. High exposure levels of VOCs and MeTs were found in children. 1,2-Dichloroethane/ethylbenzene ratio and 1,2-dichloroethane were promising diagnostic indexes for identifying e-waste pollution. Remarkably, exposure to six VOCs (e.g., acrolein and benzene) and four MeTs (e.g., arsenic and vanadium) posed considerable carcinogenic or non-carcinogenic risks to children, while changing personal lifestyles (e.g., enhancing physical exercise) may mitigate these chemical exposure risks. These findings signified the non-negligible risks of exposure to some VOCs and MeTs in regulated ER.

CRediT authorship contribution statement

Hong-Xuan Kuang: Writing – original draft. Hong-Xuan Kuang, and Meng-Yang Li: Formal analysis; Hong-Xuan Kuang, Meng-Yang Li, Yang Zhou, Zhen-Chi Li, and Ming-Deng Xiang: Investigation; Methodology. Hong-Xuan Kuang and Yun-Jiang Yu: Supervision; Review of experimental scheme & protocol; Writing - review & editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data Availability

Data will be made available on request.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.jhazmat.2023.131598.

References

[1] Ahirwar, R., Tripathi, A.K., 2021. E-waste management: a review of recycling process, environmental and occupational health hazards, and potential solutions. Environ Nanotechnol Monit Manag 15, 100409.
[2] Ashrap, P., Meeker, J.D., Sanchez, B.N., Basu, N., Tamayo-Ortiz, M., Solano-Gonzalez, M., Mercado-Garcia, A., Tellez-Rojo, M.M., Peterson, K.E., Watkins, D.J., 2020. In utero and peripubertal metals exposure in relation to reproductive hormones and sexual maturation and progression among boys in Mexico City. Environ Health 19, 124.
[3] Balali-Mood, M., Naseri, K., Tahergorabi, Z., Khazdair, M.R., Sadeghi, M., 2021. Toxic mechanisms of five heavy metals: Mercury, lead, chromium, cadmium, and arsenic. Front Pharmacol 12, 643972.
[4] Bearer, C.F., 1995. How are children different from adults? Environ Health Perspect 103, 7–12.
[5] CDC). Fourth national report on human exposure to environmental chemicals, updated tables, March 2021. Available from: https://www.cdc.gov/exposurereport/ (Accessed 12 December 2021).
[6] Chen, D., Liu, R., Lin, Q., Ma, S., Li, G., Yu, Y., Zhang, C., An, T., 2021. Volatile organic compounds in an e-waste dismantling region: From spatial-seasonal variation to human health impact. Chemosphere 275, 130022.
[7] Costa-Beber, L.C., Heck, T.G., Fiorin, P.B.G., Ludwig, M.S., 2021. HSP70 as a biomarker of the thin threshold between benefit and injury due to physical exercise when exposed to air pollution. Cell Stress Chaperon– 26, 889–915.
[8] Fan, M.Y., Zhang, Y.L., Lin, Y.C., Li, L., Xie, F., Hu, J., Mozaffar, A., Cao, F., 2021. Source apportionments of atmospheric volatile organic compounds in Nanjing, China during high ozone pollution season. Chemosphere 263, 128025.
[9] Faure, S., Noisel, N., Werry, K., Karthikeyan, S., Aylward, L.L., St-Amand, A., 2020. Evaluation of human biomonitoring data in a health risk based context: An updated analysis of population level data from the Canadian Health Measures Survey. Int J Hyg Environ Health 223, 267–280.
[10] Forti, V., Baldé, C.P., Kuehr, R., Bel, G., 2020. The global e-waste monitor 2020: Quantities, flows and the circular economy potential. United Nations University (UNU)/United Nations Institute for Training and Research (UNITAR) – co-hosted SCYCLE Programme, International Telecommunication Union (ITU) & International Solid Waste Association (ISWA),, Bonn/Geneva/Rotterdam.
[11] Hays, S.M., Aylward, L.L., Gagne, M., Nong, A., Krishnan, K., 2010. Biomonitoring equivalents for inorganic arsenic. Regul Toxicol Pharmacol 58, 1–9.
[12] Hays, S.M., Macey, K., Nong, A., Aylward, L.L., 2014. Biomonitoring equivalents for selenium. Regul Toxicol Pharmacol 70, 333–339.
[13] Hays, S.M., Macey, K., Poddalgoda, D., Lu, M., Nong, A., Aylward, L.L., 2016. Biomonitoring equivalents for molybdenum. Regul Toxicol Pharmacol 77, 223–229.
[14] Hays, S.M., Nordberg, M., Yager, J.W., Aylward, L.L., 2008. Biomonitoring Equivalents (BE) dossier for cadmium (Cd) (CAS No. 7440-43-9). Regul Toxicol Pharmacol 51, S49–S56.
[15] Houessionon, M.G.K., Ouendo, E.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.
[16] Huang, Z.-H., Zhang, Y.-L., Yan, Q., Wang, Z.-Y., Zhang, Z., Wang, X.-M., 2017. Decreased human respiratory absorption factors of aromatic hydrocarbons at lower exposure levels: The dual effect in reducing ambient air toxics. Environ Sci Technol Lett 4, 463–469.
[17] IARC monographss, volumes 1–122. Available from: https://monographs.iarc.fr/agents-classified-by-the-iarc/ (Accessed 12 December 2021).
[18] Keil, A.P., Buckley, J.P., O'Brien, K.M., Ferguson, K.K., Zhao, S., White, A.J., 2020. A Quantile-Based g-computation approach to addressing the effects of exposure mixtures. Environ Health Perspect 128, 47004.
[19] Kuang, H., Feng, J., Li, Z., Tan, J., Zhu, W., Lin, S., Pang, Q., Ye, Y., Fan, R., 2022. Volatile organic compounds from second-hand smoke may increase susceptibility of children through oxidative stress damage. Environ Res 207, 112227.
[20] Kuang, H., Li, Y., Jiang, W., Wu, P., Tan, J., Zhang, H., Pang, Q., Ma, S., An, T., Fan, R., 2019. Simultaneous determination of urinary 31 metabolites of VOCs, 8-hydroxy-2'-deoxyguanosine, and trans-3'-hydroxycotinine by UPLC-MS/MS: (13) C- and (15)N-labeled isotoped internal standards are more effective on reduction of matrix effect. Anal Bioanal Chem 411, 7841–7855.
[21] Kuang, H., Li, Y., Li, L., Ma, S., An, T., Fan, R., 2022. Four-year population exposure study: Implications for the effectiveness of e-waste control and biomarkers of e-waste pollution. Sci Total Environ 842, 156595.
[22] Kuang, H., Li, Z., Lv, X., Wu, P., Tan, J., Wu, Q., Li, Y., Jiang, W., Pang, Q., Wang, Y., Fan, R., 2021. Exposure to volatile organic compounds may be associated with oxidative DNA damage-mediated childhood asthma. Ecotoxicol Environ Saf 210, 111864.
[23] Kuang, H., Liu, J., Zeng, Y., Zhou, W., Wu, P., Tan, J., Li, Y., Pang, Q., Jiang, W., Fan, R., 2020. Co-exposure to polycyclic aromatic hydrocarbons, benzene and toluene may impair lung function by increasing oxidative damage and airway inflammation in asthmatic children. Environ Pollut 266, 115220.
[24] Kuang, H.X., Li, M.Y., Li, L.Z., Li, Z.C., Wang, C.H., Xiang, M.D., Yu, Y.J., 2023. Co-exposure levels of volatile organic compounds and metals/metalloids in children: Implications for E-waste recycling activity prediction. Sci Total Environ, 160911.
[25] Lee, J.W., Lee, C.K., Moon, C.S., Choi, I.J., Lee, K.J., Yi, S.M., Jang, B.K., Yoon, B.J., Kim, D.S., Peak, D., Sul, D., Oh, E., Im, H., Kang, H.S., Kim, J., Lee, J.T., Kim, K., Park, K.L., Ahn, R., Park, S.H., Kim, S.C., Park, C.H., Lee, J.H., 2012. Korea National Survey for Environmental Pollutants in the Human Body 2008: heavy metals in the blood or urine of the Korean population. Int J Hyg Environ Health 215, 449–457.
[26] Li, Y., Cha, C., Lv, X., Liu, J., He, J., Pang, Q., Meng, L., Kuang, H., Fan, R., 2020. Association between 10 urinary heavy metal exposure and attention deficit hyperactivity disorder for children. Environ Sci Pollut Res Int 27, 31233–31242.
[27] Lin, K., Lin, M., Ruan, J., 2022. Occupational threat of recycling spent lithium-ion batteries by vacuum reduction. ACS Sustain Chem Eng 10, 15297–15304.
[28] Liu, R., Chen, J., Li, G., An, T., 2017. Using an integrated decontamination technique to remove VOCs and attenuate health risks from an e-waste dismantling workshop. Chem Eng J 318, 57–63.
[29] Liu, R., Chen, J., Li, G., Wang, X., An, T., 2018. Cutting down on the ozone and SOA formation as well as health risks of VOCs emitted from e-waste dismantlement by integration technique. J Environ Manag 249. S0301-4797(0318)30786-30782.
[30] Liu, R., Ma, S., Chen, D., Li, G., Yu, Y., Fan, R., An, T., 2022. Human exposure to BTEX emitted from a typical e-waste recycling industrial park: External and internal exposure levels, sources, and probabilistic risk implications. J Hazard Mater 437, 129343.
[31] Lv, Y., Wei, Y., Zhou, J., Xue, K., Guo, Y., Liu, Y., Ju, A., Wu, B., Zhao, F., Chen, C., Xiong, J., Li, C., Gu, H., Cao, Z., Ji, J.S., Shi, X., 2021. Human biomonitoring of toxic and essential metals in younger elderly, octogenarians, nonagenarians and centenarians: analysis of the healthy ageing and biomarkers cohort Study (HABCS) in China. Environ Int 156, 106717.
[32] MEE). The transformation of "the e-waste capital of the world"– Practice of comprehensive remediation of "Scattered and Polluted" in Guiyu Town, Shantou City, Guangdong Province. Available from: https://www.mee.gov.cn/xxgk2018/xxgk/xxgk15/201908/t20190828_730337.html (Accessed 5 December 2022).
[33] Parvez, S.M., Jahan, F., Brune, M.-N., Gorman, J.F., Rahman, M.J., Carpenter, D., Islam, Z., Rahman, M., Aich, N., Knibbs, L.D., Sly, P.D., 2021. Health consequences

of exposure to e-waste: an updated systematic review. Lancet Planet Health 5, e905–e920.
[34] Pezzagno, G., Maestri, L., Fiorentino, M.L., 1999. Trans,trans-muconic acid, a biological indicator to low levels of environmental benzene: some aspects of its specificity. Am J Ind Med 35, 511–518.
[35] Poddalgoda, D., Macey, K., Jayawardene, I., Krishnan, K., 2016. Derivation of biomonitoring equivalent for inorganic tin for interpreting population-level urinary biomonitoring data. Regul Toxicol Pharmacol 81, 430–436.
[36] Qin, B., Yao, Z., Deng, K., Ruan, J., Xu, Z., 2021. Analysis of contaminants and their formation mechanism in the desiccation-dissociation process of organic impurity of waste glass. J Hazard Mater 416, 125881.
[37] Rueff, J., Teixeira, J.P., Santos, L.S., Gaspar, J.F., 2009. Genetic effects and biotoxicity monitoring of occupational styrene exposure. Clin Chim Acta 399, 8–23.
[38] Saez, C., Sanchez, A., Yusa, V., Dualde, P., Fernandez, S.F., Lopez, A., Corpas-Burgos, F., Aguirre, M.A., Coscolla, C., 2021. Health Risk Assessment of Exposure to 15 Essential and Toxic Elements in Spanish Women of Reproductive Age: A Case Study. Int J Environ Res Public Health 18, 13012.
[39] Sanchez, T.R., Slavkovich, V., LoIacono, N., van Geen, A., Ellis, T., Chillrud, S.N., Balac, O., Islam, T., Parvez, F., Ahsan, H., Graziano, J.H., Navas-Acien, A., 2018. Urinary metals and metal mixtures in Bangladesh: Exploring environmental sources in the Health Effects of Arsenic Longitudinal Study (HEALS). Environ Int 121, 852–860.
[40] Saxena, S., Abdel-Rahman, M.S., 1989. Pharmacodynamics of benzyl chloride in rats. Arch Environ Contam Toxicol 18, 669–677.
[41] Song, W., Han, Q., Wan, Y., Qian, X., Wei, M., Jiang, Y., Wang, Q., 2022. Repeated measurements of 21 urinary metabolites of volatile organic compounds and their associations with three selected oxidative stress biomarkers in 0-7-year-old healthy children from south and central China. Chemosphere 287, 132065.
[42] Sun, B., Schnoor, J.L., Zeng, E.Y., 2022. Decadal journey of e-waste recycling: What has it achieved? Environ Sci Technol 56, 12785–12792.
[43] Sun, J., Shen, Z., Zhang, Y., Zhang, Z., Zhang, Q., Zhang, T., Niu, X., Huang, Y., Cui, L., Xu, H., Liu, H., Cao, J., Li, X., 2019. Urban VOC profiles, possible sources, and its role in ozone formation for a summer campaign over Xi'an, China. Environ Sci Pollut Res Int 26, 27769–27782.
[44] Takyi, S.A., Basu, N., Arko-Mensah, J., Dwomoh, D., Houessionon, K.G., Fobil, J.N., 2021. Biomonitoring of metals in blood and urine of electronic waste (E-waste) recyclers at Agbogbloshie, Ghana. Chemosphere 280, 130677.
[45] Tang, B., Chen, S.J., Zheng, J., Xiong, S.M., Yan, X., Luo, W.K., Mai, B.X., Yu, Y.J., 2022. Changes in human hair levels of organic contaminants reflecting China's regulations on electronic waste recycling. Sci Total Environ 806, 150411.
[46] Taussky, H.H., Kurzmann, G., 1954. A microcolorimetric determination of creatine in urine by the Jaffe reaction. J Biol Chem 208, 853–861.
[47] Wahlang, B., Gao, H., Rai, S.N., Keith, R.J., McClain, C.J., Srivastava, S., Cave, M. C., Bhatnagar, A., 2023. Associations between residential volatile organic compound exposures and liver injury markers: The role of biological sex and race. Environ Res 221, 115228.
[48] Wang, W., Du, C., Lin, L., Chen, W., Tan, L., Shen, J., Pearce, E.N., Zhang, Y., Gao, M., Bian, J., Wang, X., Zhang, W., 2018. Anthropometry-based 24-h urinary creatinine excretion reference for Chinese children. PLoS One 13, e0197672.
[49] Wu, Q., Leung, J.Y., Geng, X., Chen, S., Huang, X., Li, H., Huang, Z., Zhu, L., Chen, J., Lu, Y., 2015. Heavy metal contamination of soil and water in the vicinity of an abandoned e-waste recycling site: implications for dissemination of heavy metals. Sci Total Environ 506–507, 217–225.
[50] Wu, Y., Li, G., An, T., 2022. Toxic metals in particulate matter and health risks in an e-waste dismantling park and its surrounding areas: Analysis of 3 PM size groups. Int J Environ Res Public Health 19, 15383.
[51] Xue, Q., Zhou, Y., Gu, H., Xie, X., Hou, F., Liu, Q., Wu, H., Zhu, K., Wan, Z., Song, R., 2020. Urine metals concentrations and dyslexia among children in China. Environ Int 139, 105707.
[52] Yu, Y.J., Li, Z.C., Zhou, Y., Dong, C.Y., Kuang, H.X., Zheng, T., Xiang, M.D., Chen, X.C., Li, H.Y., Zeng, X.W., Xu, S.L., Hu, L.W., Dong, G.H., 2023. Associations between trace level thallium and multiple health effects in rural areas: Chinese exposure and response mapping program (CERMP). Sci Total Environ 862, 160466.
[53] Zhang, D., Jin, T., Xu, Y.Q., Lu, Y.F., Wu, Q., Zhang, Y.K., Liu, J., 2012. Diurnal-and sex-related difference of metallothionein expression in mice. J Circadian Rhythms 10, 5.
[54] Zhao, X.G., Duan, X.L., 2016. Exposure Factors Handbook of Chinese Population (Children 6-17 years). China Environment Press.