

This is an Open Access document downloaded from ORCA, Cardiff University's institutional repository:<https://orca.cardiff.ac.uk/id/eprint/131964/>

This is the author's version of a work that was submitted to / accepted for publication.

Citation for final published version:

Feng, Xiaolei, Shao, Longyi, Xi, Chunxiu, Jones, Tim, Zhang, Daizhou and Berube, Kelly 2020. Particle-induced oxidative damage by indoor size-segregated particulate matter from coal-burning homes in the Xuanwei lung cancer epidemic area, Yunnan Province, China. *Chemosphere* 256, 127058. [10.1016/j.chemosphere.2020.127058](https://doi.org/10.1016/j.chemosphere.2020.127058)

Publishers page: [http://dx.doi.org/10.1016/j.chemosphere.2020.127058...](http://dx.doi.org/10.1016/j.chemosphere.2020.127058)

Please note:

Changes made as a result of publishing processes such as copy-editing, formatting and page numbers may not be reflected in this version. For the definitive version of this publication, please refer to the published source. You are advised to consult the publisher's version if you wish to cite this paper.

This version is being made available in accordance with publisher policies. See <http://orca.cf.ac.uk/policies.html> for usage policies. Copyright and moral rights for publications made available in ORCA are retained by the copyright holders.



1 **Particle-induced oxidative damage by indoor size-segregated particulate matter**
2 **from coal-burning homes in the Xuanwei lung cancer epidemic area, Yunnan**
3 **Province, China.**

4 Xiaolei Feng ¹, Longyi Shao ^{1,*}, Chunxiu Xi ¹, Tim Jones ², Daizhou Zhang ³, Kelly Bérubé ⁴

5 ¹ State Key Laboratory of Coal Resources and Safe Mining and College of Geoscience and Surveying
6 Engineering, China University of Mining and Technology, Beijing 100083, China

7 ² School of Earth and Ocean Sciences, Cardiff University, Museum Avenue, Cardiff, CF10, 3YE, UK

8 ³ Faculty of Environmental and Symbiotic Sciences, Prefectural University of Kumamoto,
9 Kumamoto 862-8502, Japan

10 ⁴ School of Biosciences, Cardiff University, Museum Avenue, Cardiff CF10, 3US, UK

11

12

13 * Corresponding author. State Key Laboratory of Coal Resources and Safe Mining and College of Geoscience and
14 Surveying Engineering, China University of Mining and Technology, Beijing 100083, PR China.

15 E-mail addresses: 506870566@qq.com (X.L. Feng), ShaoL@cumt.edu.cn (L.Y. Shao), 2558465218@qq.com

16 (C.X. Xi), JonesTP@cardiff.ac.uk (T. Jones), dzzhang@pu-kumamoto.ac.jp (D.Z. Zhang), Berube@cardiff.ac.uk
17 (K. Berube)

18

19 **Highlights:**

- 20 1. Indoor size-segregated particles were collected at the Hutou lung cancer epidemic village.
21 2. DNA damage assessed by plasmid scission assay was mainly caused by smaller particles.
22 3. DNA damage had a positive correlation with the water-soluble Zn, Cu, Cd, Rb, Cs, and Sb.
23 4. Water-soluble metals Zn, Cu, Cd, Rb, Cs, and Sb were concentrated in the smaller particles.
24 5. Indoor particles in the small sizes were a higher health risk than those in the large sizes.

25

26 **Abstract:** Size-segregated samples of airborne particulate matter were collected at the coal-burning
27 homes of the Hutou high lung cancer epidemic village and a comparison site Xize village of the
28 Xuanwei County, Yunnan Province, by an Anderson Cascade Impact Sampler in winter and spring to
29 study the toxicological characteristics of different-sized particles. The DNA damage caused by the
30 water-soluble fractions of these size-segregated particles was analyzed by the Plasmid Scission Assay,
31 and the trace element compositions were determined by Inductively Coupled Plasma Mass
32 Spectrometry. The DNA damage rate from the airborne particles in the high lung cancer incidence
33 area was higher than that in Xize village. The different-sized particles have highly varying DNA
34 damage rates, with the values being greater in the small size range than in the large size range. The
35 particle-induced DNA damage rates had a significantly positive correlation with total water-soluble
36 trace elements. Further analysis of the individual elements indicated that the water-soluble heavy
37 metals Zn, Cu, Cd, Rb, Cs, and Sb had a positive correlation with the particle-induced DNA damage,
38 implying that these water-soluble heavy metals played an important role in the DNA damage. The Sr
39 had a negative correlation with the particle-induced DNA damage, suggesting that the water-soluble
40 Sr might counter DNA damage. The mass concentrations of the total and individual water-soluble
41 trace elements were mostly enriched in the small particle size ranges, thus implying the indoor
42 airborne particles in the small size ranges would have a higher health risk.

43

44 **Keywords:** Xuanwei lung cancer epidemic, coal-burning emissions, indoor size-segregated
45 particulate matter, water-soluble trace elements, plasmid scission assay

46

47 **1. Introduction**

48 Xuanwei County has attracted considerable attention due to the unusually high incidence of lung
49 cancer levels and mortality (Zhou et al., 2006, Chen et al., 2015). A number of studies showed that
50 tobacco smoking is not sufficient to explain the high lung cancer incidence in Xuanwei, and instead,
51 the coal burning is closely related to this lung cancer (Barone-Adesi et al., 2012; Hosgood et al., 2013;

52 Shao et al., 2013; Seow et al. 2014; Lui et al., 2017; Finkelman et al., 2018). The incidence of lung
53 cancer in Xuanwei's rural areas is the highest in China at approximately five times the national
54 average and was highly associated with the domestic coal combustion (Mumford et al., 1987; He et
55 al., 1994). Yang et al., (2010) found that lung cancer incidence in Xuanwei females was higher than
56 that in Xuanwei males and non-Xuanwei females, implying that lung cancer incidence might be
57 related to the coal burning particles emitted during cooking. The main energy source for residents in
58 Xuanwei City is the C1 coal situated stratigraphically at the end-Permian mass extinction (Large et
59 al., 2009; Shao et al., 2015; Wang et al., 2018) which is characterized by medium to high ash yields
60 (averaged 31.0%), low to medium volatile contents (averaged 20.0%), low sulfur contents (averaged
61 0.17%), and mid-rank (with the vitrinite reflectance from 1.19% to 1.37%) (Shao et al., 2015).

62 The study by He and Yang, (1994) revealed that the main risk factor for the high lung cancer
63 mortality in Xuanwei was the exposure to the high levels of carcinogenic polycyclic aromatic
64 hydrocarbons (PAHs) (e.g., BaP) from indoor coal emissions. Tian et al., (2008) reported that the
65 exposure variable associated with the lung cancer risk in Xuanwei was the indoor air emissions of
66 crystalline silica. A study carried by Large et al., (2009) revealed that high silica content in coal from
67 Xuanwei may be interacting with toxic volatiles in the coal to cause unusually high rates of lung
68 cancer. Zhang et al., (2016) studied the particles emitted from residential coal fires and found that the
69 toxicity caused by the fine particles is higher. A study of the characteristics of individual particles
70 emitted by experimental combustion of the Xuanwei coals demonstrated that harmful minerals like
71 quartz and Si-rich particles comprise the main particulate matter (Wang et al., 2019).

72 There are extensive studies dealing with the health effects of airborne particulate matter (PM)
73 (Rückerl et al., 2011; Brauer et al., 2012; Balakrishnan et al., 2015; Hu et al., 2016; Wang et al., 2017;
74 Li et al., 2020; Xing et al., 2020). Long-term exposure to PM_{2.5} (i.e. PM ≤ 2.5 microns in aerodynamic

75 diameter) can lead to small gestational age (Hao et al., 2019), respiratory disease (Li et al., 2013a,
76 2013b; Conibear et al., 2018; Slama et al., 2019), ischemic heart disease (Pope et al., 2015; Thurston
77 et al., 2016; Shah et al., 2013), and a range of lung diseases (Chow, 2006; Araujo et al., 2011),
78 especially in residential areas that burn large amounts of fossil fuels. Even in the case of low mass
79 concentrations of airborne particles, pollutants still can increase exposure risk (Kim et al., 2015).
80 Numerous studies have shown that types and levels of the metal elements in airborne particulates
81 have important effects on health, causing lung damage in humans (Zelikoff et al., 2002; Gilli et al.,
82 2007). Costa and Dreher (1997) noticed that biologically active transition metal elements play a
83 greater role in lung damage, and they have suggested that the soluble metal elements in the particles
84 are even more harmful. A study by Xue et al., (2018) has shown that a positive correlation exists
85 between lung cancer and PM_{2.5}. The metal elements can be abundant in aerosols, especially in the
86 fine particles with size ranges below 2.5µm (Schneidemesser et al., 2010).

87 The plasmid scission assay is an *in vitro* particle toxicology method employed to study oxidative
88 damage of supercoiled DNA by free radicals (Greenwell et al., 2002; Moreno et al., 2004; Shao et al.,
89 2013), and has been widely used to assess the oxidative capacity of airborne PM (Chuang et al., 2015;
90 Shao et al., 2016; Shao et al., 2017). Several studies have noted that the DNA damage caused by
91 water-soluble components in airborne particulates is significant (Merolla et al., 2005; Shao et al.,
92 2013; Hou et al., 2016).

93 Trace elements and their concentrations can differ in particles of different sizes and can thus
94 have different impacts on human health (Cao et al., 2012; Chow et al., 2012; Duan et al., 2012; Sun
95 et al., 2014; Rohra et al., 2018). Particles with small sizes are of more concern due to their large
96 specific surface area and strong adsorption capacity, leading to the enrichment of heavy metals in the
97 smaller particles (Al-Dabbous et al., 2018; Olawoyin et al., 2018). Pan et al., (2015) have found that

98 the toxic trace element content in northern China were concentrated in the submicron ($<1.1\mu\text{m}$ in
99 aerodynamic diameter) particle size range. Although the association of the water-soluble elements
100 with the DNA damage in the particulate matter in a Xuanwei high lung cancer incidence area has
101 been demonstrated by Shao et al., (2013), the size-segregated particulate matter, the size distribution
102 of the water-soluble trace elements and the particle-induced DNA damages are poorly understood.

103 The human respiratory system (RS) is perpetually exposed to oxidants generated either
104 endogenously or exogenously (e.g. indoor and outdoor combustion-derived PM; CDPM). Reactive
105 oxygen species (ROS) and oxidative stress in the RS increases the production of inflammatory
106 mediators that promote carcinogenic mechanisms. The risk of developing lung cancer increases
107 substantially following exposure to CDPM due to the synergistic effects in the generation of ROS
108 and concomitant induction of oxidative stress and inflammation that culminates in an elevated DNA
109 damage potential (IARC, 2012). PM physico-chemistry (e.g. size, shape, solubility, transition metal
110 content, speciation, stable free radicals, etc.) plays an important role in its oxidative capacity (Hamra
111 et al., 2014). PM-induced ROS initiates the synthesis of mediators of inflammation in lung epithelial
112 cells and initiation of carcinogenic mechanisms (DEFRA, 2017).

113 In this study, we collected airborne particles of different sizes in a high lung cancer incidence
114 village and a comparison village in Xuanwei County. The mass concentration and oxidative damage
115 characteristics of PM in different particle size ranges were analyzed. The distribution of the water-
116 soluble trace elements in the different particle sizes were also investigated, and their relationships
117 with the plasmid DNA damage rates were studied in order to identify the major heavy metal elements
118 and the major particle size ranges which are more closely associated with the DNA damage.

119

120 **2. Materials and methods**

121 **2.1. Sampling**

122 The samples were collected from Hutou village in Xuanwei, an area with high-risk lung cancer,
123 and Xize village, a low-risk comparison site. The sampling campaign was divided into two periods.
124 One was in December 2016 during winter when the coal is extensively used, and the other was in
125 March 2019 during spring when the coal is used less, both of which had same sampling sites. The
126 sampler was placed in the kitchens of two villagers' apartments in Hutou village (Hutou I, Hutou II)
127 and one villager's apartment in Xize village. Coal was used as the fuel at all sampling sites. The
128 selected sampling point will only burn coal for cooking during the winter and did not use coal at other
129 times during the spring. Xize village only burns coal for cooking. The sampling equipment was an
130 eight-stage Anderson cascade impact sampler (TE-20-800 TISCH, Germany), with a sampling height
131 1.5m above the ground. Quartz filters (80mm diameter, Millipore, China) were used for sampling.
132 The sampling flow rate was 28.3L/min, and the sampler segregated particles in eight size ranges:
133 0.43-0.65 μm , 0.65-1.1 μm , 1.1-2.1 μm , 2.1-3.3 μm , 3.3-4.7 μm , 4.7-5.8 μm , 5.8-9.0 μm and 9.0-10 μm .

134 Before sampling, the quartz filters were heated at 450 $^{\circ}\text{C}$ for 4 hours and placed in a constant
135 temperature and humidity chamber (Hitachi, Japan; temperature: 20 $^{\circ}\text{C} \pm 5^{\circ}\text{C}$, relative humidity: 45%
136 $\pm 5\%$). After the samples were collected, they were placed in the constant temperature and humidity
137 chamber for 48 hours before weighing. The mass concentrations of the different size particles were
138 obtained using the gravimetric method. The blank filters and the sample-loaded filters were weighed
139 using an electronic balance (MS105DU, Switzerland) with an accuracy of 0.01mg. The mass
140 concentration of the sample was calculated according to the sampling flow rate as follows:

$$141 \quad C = \frac{W_2 - W_1}{V_s}$$
$$142 \quad V_s = \frac{P \times V \times 273}{1013.25 \times (273 + T)}$$

143 Where: C-sample mass concentration (g/m^3), V_s -standard sample volume (m^3), W_1 -weight of
144 quartz membrane before sampling (g), W_2 -weight of quartz membrane after sampling (g), V-actual

145 sampling volume (m³), P-average air pressure (hPa), T-sampling average temperature (°C).

146 Simultaneously, a portable meteorological instrument (Kestrel 5500 Weather LiNK, USA) was
147 used throughout the sampling process to monitor meteorological conditions including temperature,
148 relative humidity, and pressure. The sample information and the meteorological conditions during the
149 sampling period are shown in Table 1.

150

151 **2.2. Plasmid DNA scission assay**

152 The plasmid DNA assay is an *in vitro* method to measure oxidative damage to plasmid DNA
153 induced by free radicals on the particle surface. Oxidative damage initially causes the supercoiled
154 DNA to relaxed and further damage results in linearization (Figure 1). The sum of the percentage of
155 relaxed DNA and linearized DNA is the oxidative damage rate. Ultra-pure water (conductivity 18.2
156 MΩ, Millipore, China) was used as a procedure blank throughout the experiment. Ultra-pure water's
157 effect on DNA was subtracted to calculate the particle oxidative damage rate. Four samples were used
158 for each test to verify experiment accuracy. The methodological steps are as follows:

159 (1) Sample preparation process: Each filter (along with a corresponding blank filter) was cut into
160 half, and the mass of each sample was calculated. The sample was immersed in ultra-pure water. The
161 sample solution was gently shaken for 20 hours in a shaker (VORTEX-GENIE2, Scientific Industries,
162 USA) and then sonicated for 2 min. This released the particulate matter from the filter substrate into
163 the solution. After this step, the sample solution was centrifuged (D-37520 Osterode, Germany) to
164 deposit particles at the bottom of the tube, then, the mixture supernatant was taken as water-soluble
165 sample. The water-soluble sample was adjusted to a 250μg/mL dose level. 4μl of X174-RF DNA
166 (Promega, USA) was added.

167 (2) Gel preparation: Tris/Borate/EDTA (TBE) buffer solution (Thermo Scientific, China) diluted
168 10 times with agarose (molecular biology grade; Sigma-Aldrich, China) was mixed, and the solution

169 was heated to clarity and transparency. The solidified gel was placed in an electrophoresis cell
170 (DYCP-34C; LIUYI, China) containing ten-times diluted TBE buffer.

171 (3) Injection of DNA mixtures and sample into the gel: 14 μ L of bromophenol blue stain (Sigma-
172 Aldrich, China) was added to the sample. After rocking horizontally for 6 hours in a rocker (HX-3000,
173 YOUNING, China), a fixed volume pipette (Eppendorf, Germany) was used to load 20 μ L of the
174 solution into each gel well. Four parallel samples were made for each sample. 20 μ L of ethidium
175 bromide (EB; Sigma-Aldrich, China) was added to both sides of the electrophoresis tank. After the
176 EB was fully dissolved in the buffer, the laboratory electrophoresis power supply (DYY-6C, LIUYI,
177 China) was turned on and operated at 30 Volts for 16 hours.

178 (4) Gel imaging and quantitative analysis of oxidative DNA damage: After 16 hours, the optical
179 densities of three different DNA morphologies (i.e. super-coiled, relaxed and linear) in the gel were
180 captured using a gel documentation system (ChemiDoc, Bio-red, China) and the Genetools (version
181 4.0; Syngene, USA) image analysis software program was utilized to determine the damage rate of
182 DNA via a linear regression analysis. In the final calculation, the DNA damage of ultra-pure water
183 was subtracted from the DNA damage caused by particles.

184

185 **2.3. Inductively coupled plasma mass spectrometer (ICP-MS)**

186 The water-soluble samples obtained by the first step of Plasmid DNA scission assay were also
187 chemically analyzed by inductively coupled plasma mass spectrometry (ICP-MS, 7700x, Agilent
188 Ltd.). Chemical analysis of the water-soluble sample using ICP-MS identified 26 elements with
189 values above the detection limit. The trace element content (ng/mL), the solution volume (mL), and
190 the actual sample volume in the water-soluble sample were calculated according to the following
191 formula to obtain the mass concentration (ng/m³) of the water-soluble element.

192
$$C = \frac{c \times Vt}{V}$$

193 Where: C represents the mass concentration of water-soluble trace elements (ng/m^3), c represents
194 the trace element content in the water-soluble sample (ng/mL), Vt represents the volume of the
195 ~~measured~~ total solution (mL), V represents the actual volume at the time of sampling (m^3).

196

197 **3. Results and Discussion**

198 **3.1 Mass concentrations of size-segregated particles**

199 The mass concentrations of size-segregated particles are presented in Table 2. The PM mass
200 concentration in winter was greater than that in spring in the Hutou village, and the total PM mass
201 concentrations at Hutou in both winter and summer were all significantly higher than that in the Xize
202 village.

203 The mass concentrations of PM had a clear size distribution (Table 2) with the PM mass
204 concentration decreasing first and then increasing with increased particle size. Due to sampler effects,
205 particles in the large size range may fall into the small size collections, resulting in abnormally high
206 values of the particle mass concentration in the $0.65\text{-}1.1\mu\text{m}$ range.

207 For Hutou I, the winter sample showed that the PM mass concentration decreased first and then
208 increased with increased particle size, while the spring sample showed that the PM mass
209 concentration continuously increased with the increase of particle size. The winter sample had the
210 lowest mass concentration value ($27.10\mu\text{g}/\text{m}^3$) in the $2.1\text{-}3.3\mu\text{m}$ range and the maximum value
211 ($67.17\mu\text{g}/\text{m}^3$) in the $9.0\text{-}10\mu\text{m}$ range. The spring sample had the lowest mass value ($24.17\mu\text{g}/\text{m}^3$) in
212 the $0.43\text{-}0.65\mu\text{m}$ and maximum values ($62.78\mu\text{g}/\text{m}^3$) in the $9.0\text{-}10\mu\text{m}$ ranges, with the maximum value
213 being in the same size range as the winter sample (Figure 2).

214 For Hutou II, both winter and spring samples showed that the PM mass concentration decreased

215 first and then increased with increased particle size. The winter sample had the lowest value
216 ($24.86\mu\text{g}/\text{m}^3$) in the $2.1\text{-}3.3\mu\text{m}$ range and the maximum value ($44.56\mu\text{g}/\text{m}^3$) in the $0.43\text{-}0.65\mu\text{m}$ range.
217 The spring sample had the lowest value ($21.29\mu\text{g}/\text{m}^3$) in the $2.1\text{-}3.3\mu\text{m}$ range and the highest value
218 ($42.11\mu\text{g}/\text{m}^3$) in the $9.0\text{-}10\mu\text{m}$ range (Figure 2).

219 In Xize village, the PM mass concentration also decreased first and then increased with increased
220 particle size, the same as the Hutou II and the winter sample of Hutou I. It had the lowest value
221 ($3.51\mu\text{g}/\text{m}^3$) in the $2.1\text{-}3.3\mu\text{m}$ range and the highest value ($32.93\mu\text{g}/\text{m}^3$) in the $0.43\text{-}0.65\mu\text{m}$ range
222 (Figure 2).

223 A comparison of the PM mass-size distribution at the sampling sites in Hutou and Xize villages
224 revealed that the Hutou samples had a higher value in the $0.65\text{-}10\mu\text{m}$ range and a lower value in the
225 $0.43\text{-}0.65\mu\text{m}$ range than the Xize sample (Figure 2).

226 The large particles in the $5.8\text{-}10\mu\text{m}$ size range are deposited in the nasal cavity, and the small
227 particles in the $0.43\text{-}2.1\mu\text{m}$ size range enter the bronchi (Sridevi et al., 2017) and the particles in the
228 $2.5\text{-}5\mu\text{m}$ have a larger residence time than the particles in the $5\text{-}10\mu\text{m}$ (Schleicher et al., 2011).

229 Comparing the winter and spring samples of Hutou village, the winter sample mass concentration
230 contains significant smaller particles, indicating that coal combustion has a significant impact on
231 indoor air quality. One possible explanation could be that there is more ambient airborne during the
232 spring. For both the small and large particle in the winter sample at the three sampling sites, the PM
233 mass concentration in Hutou is higher than that in Xize (in Figure 3).

234

235 **3.2 Oxidative capacity of the size-segregated airborne particles**

236 Numerous studies have demonstrated that the oxidative capacity of the whole particle
237 suspensions were similar to those of the water-soluble fractions, indicating that the DNA damage

238 induced by particulate matter was mainly a result of the water-soluble fraction (Shao et al., 2013,
239 2016). Therefore, we analyzed the oxidative capacity of the water-soluble components of size-
240 segregated particles of the collected samples.

241 For the comparison study, the DNA damage rate at the particle dosage of 250 μ g/mL was taken
242 for the size-segregated particles (Table 3). It can be seen that the damage rates of the Hutou samples
243 at this dosage were generally higher than those of the Xize samples, and the damage rates of all the
244 winter samples were generally higher than those of the all spring samples in the Hutou village. It is
245 also clear that the damage rates of the samples in the small size ranges, mostly 0.43-1.1 μ m, were
246 higher than those in the 9-10 μ m size ranges. Although the damage rates increased slightly in the 5.8-
247 10 μ m size range, they were still lower than the damage rates caused by the small size ranges. Sun et
248 al., (2014) studied the haze and clear weather PM in Beijing, they found that the average damage rate
249 in the 0.32-1.8 μ m size ranges was higher than that in the 5.6-10 μ m size ranges. This is similar to the
250 case in Xuanwei where the DNA damage rate caused by small particles is also greater than that caused
251 by large particles.

252 It can also be seen in Table 3 that the DNA damage rates of the winter samples of Hutou village
253 were higher than those of Xize village, but those of the spring samples of Hutou I particles in the 4.7-
254 5.8 μ m size range and Hutou II in the 0.65-3.3 μ m size range were lower than the those of Xize particles
255 in the corresponding size ranges.

256

257 **3.3 Relationship between water-soluble trace element content and the particle-induced DNA** 258 **damage of the size-segregated particles**

259 To examine the most probable source of the particle-induced oxidative capacities of the PM
260 samples, the DNA damage rates from particle doses of 250 μ g/ml were correlated against the

261 corresponding contents of the total and individual water-soluble elements in the PM (Table 4).
262 According to the correlation analysis, the coefficient is higher than 0.393 at the 0.01 significance level
263 with a sample number $N = 40$, which indicates that these two factors are related.

264 It can be seen in Table 4 that there is a significant positive correlation between the DNA damage
265 rates and the contents of total water-soluble trace elements, with the correlation coefficient of 0.649,
266 being higher than the critical value of 0.393. This demonstrated that the particle-induced DNA
267 damage was mainly caused by the water-soluble elements.

268 The individual elements Zn, Cu, Cd, Rb, Tl, Cs, and Sb showed a significant correlation with
269 DNA damage rates, all of which had correlation coefficients higher than the critical value. Among
270 these elements, the correlation coefficients of Cu (0.639), Zn (0.679), Tl (0.681), and Rb (0.690) were
271 particularly high. Opposite to these elements, the Sr had a negative correlation with the particle-
272 induced DNA damage, being -0.428, implying that the water-soluble Sr might mediate DNA damage.

274 **3.4 Mass concentrations and health risk of total water-soluble elements for the size-segregated** 275 **particles**

276 The correlation between the water-soluble elements and the oxidative DNA damage (Table 4)
277 revealed that the total water-soluble elements, together with the individual elements Zn, Cu, Cd, Rb,
278 Tl, Cs, and Sb, were positively associated with the oxidative potential. Therefore, the mass
279 concentrations of the total water-soluble elements and these individual Positively Correlated Water-
280 Soluble Elements (PCWSE) could represent the health risk levels of PM exposure.

281 Table 5 presents the total water-soluble element mass concentration (ng/m^3) of size-segregated
282 particles. It can be seen that the total water-soluble elements mass concentration in winter was greater
283 than that in spring for the two sampling sites in the Hutou village. The value for the winter in the Xize

284 village was very low compared to that for the winter in Hutou village, and this is mainly due to less
285 coal being burnt during winter.

286 The mass concentrations of the total water-soluble elements had a clear size distribution (Table
287 5). For Hutou I, the mass-size distribution of the total water-soluble elements of the winter samples
288 had the highest mass value (333.83ng/m^3) in the $0.43\text{-}0.65\mu\text{m}$ range and the lowest mass value
289 (9.02ng/m^3) in the $4.7\text{-}5.8\mu\text{m}$ range. The spring samples showed the highest mass value of the total
290 water-soluble elements (15.44ng/m^3) in the $0.65\text{-}1.1\mu\text{m}$ range and the lowest value (0.53ng/m^3) in the
291 $9.0\text{-}10\mu\text{m}$ range. Both the winter and spring samples demonstrated that the water-soluble elements
292 were enriched in the fine particle size range (Table 5). This is an agreement with the mass-size
293 distribution of trace elements in Xuanwei (Lv et al., 2017).

294 For Hutou II, the mass-size distribution of the total water-soluble elements of the winter sample
295 showed the highest value (21.47ng/m^3) in the $1.1\text{-}2.1\mu\text{m}$ range and the lowest value (6.26ng/m^3) in
296 the $4.7\text{-}5.8\mu\text{m}$ range. The spring sample had the highest value (10.56ng/m^3) in the $1.1\text{-}2.1\mu\text{m}$ range
297 and the lowest value (0.45ng/m^3) in the $0.43\text{-}0.65\mu\text{m}$ range.

298 In Xize village, the mass-size distribution of the total water-soluble elements showed the highest
299 value (21.69ng/m^3) and the minimum value (2.93ng/m^3) were in the $0.65\text{-}1.1\mu\text{m}$ range and $2.1\text{-}3.3\mu\text{m}$
300 range, respectively.

301 In general, the mass-size distribution of the total water-soluble elements in Xuanwei area showed
302 a higher value in the smaller than $2.1\mu\text{m}$ size range, and a lower value in the larger than $4.7\mu\text{m}$ size
303 range, indicating that the water-soluble elements tended to be enriched in the smaller particles. This
304 indicates that the indoor airborne particles in the small size ranges would have a higher health risk
305 than those in the large size ranges.

306

307 **3.5 Mass concentrations and health risk of individual water-soluble elements for the size-**
308 **segregated particles**

309 Among the analyzed elements, Zn, Cu, Cd, Rb, Tl, Cs, and Sb showed a positive correlation, and
310 Sr showed a negative correlation with the particle-induced DNA damage. The mass-size distribution
311 of these individual water-soluble trace elements in particles for the five sets of samples at three
312 sampling sites were further analyzed.

313 Figure 4a and 4b showed the mass concentrations of Zn generally decreased with the increase of
314 the particle sizes. It can be seen that Zn was mostly concentrated in the particle size range less than
315 2.1 μm , although some variation existed in the Hutou I spring sample. Zn is a major trace element
316 contributing to DNA damage rate in Beijing's atmosphere (Shao et al., 2013, 2017). Rönkkö et al.,
317 (2018) found that the mass concentration of Zn in $\text{PM}_{2.5-1.0}$ is high in Nanjing's atmosphere. In the
318 United States of America, Zn was also found to have the highest concentration in the fine particle size
319 range ($\text{PM}_{\leq 2.5\mu\text{m}}$) (Olawoyin et al., 2018). In a study on the mass-size distribution of the individual
320 water-soluble trace elements in size-segregated airborne particles, Zn was found to be concentrated
321 in the range of 0.56–1.0 μm (Sun et al., 2014). These results were also in good agreement with
322 previous studies in Beijing by Xu et al., (2007) and Duan et al., (2012) and in Shenyang by Hong et
323 al., (2011).

324 Figure 5a and 5b showed the mass concentrations of Cu. In the Hutou I winter sample, Cu was
325 mostly concentrated in the smaller than 2.1 μm particle size range, and in all other samples, a general
326 decrease in mass concentration with the increase of the sizes, although some variation existed. The
327 Hutou II winter sample had an increase in the concentration in the larger than 5.8 μm particles. The
328 study on the mass-size distribution of the individual water-soluble trace elements in size-segregated
329 airborne particles has demonstrated that Cu was concentrated in the range of 0.56–1.0 μm (Sun et al.,

330 2014), which is in a good agreement with this study. Wang et al., (2016) studied the enrich of the
331 heavy elements in Jincheng dust, indicating that they were affected by human activities.

332 Figure 6a and 6b showed that concentrations of Cd generally decreased with the increase of the
333 particle sizes. In the Hutou I and Hutou II winter samples Cd was mostly concentrated in the particles
334 smaller than $4.7\mu\text{m}$, and in all other samples, Cd was concentrated in the particles smaller than $2.1\mu\text{m}$.
335 The mass-size distribution of Cd is similar to the distribution of trace metals in the British atmosphere
336 (Allen et al., 2001). The International Agency for Research on Cancer identified that Cd is in a class
337 of substances that are carcinogenic to humans (IARC, 2017). Rönkkö et al., (2018) also found the
338 mass concentration of Cd in $\text{PM}_{2.5}$ is high in Nanjing's atmosphere.

339 Figure 7 and Figure 8 showed variation in concentrations of Rb, Tl, Cs and Sb. It can be seen in
340 the five samples that the mass-size distributions of Rb, Tl and Cs generally decreased with the
341 increase of the particle sizes, and all samples showed that Rb and Tl were both enriched in the particles
342 smaller than $2.1\mu\text{m}$ and Cs was enriched in the particles smaller than $1.1\mu\text{m}$ (Figure 7a and 7b and
343 Figure 8a). Figure 8b showed that the mass concentration of Sb had an overall decrease with the
344 increase of the particle sizes, except the Hutou II spring sample, which showed some variation, and
345 the Xize sample, which had a small increase in the particle size range of $9\text{-}10\mu\text{m}$. Ma et al., (2019)
346 studied the water-soluble particulate elements in the coastal city of Marina, California, and found that
347 Rb exhibited concentration peaks in the $0.32\text{-}0.56\mu\text{m}$ size range. Lv et al., (2017) found that Tl
348 concentrates in $\text{PM}_{2.1}$ and Sb has a dominant peak in the particles smaller then $2.1\mu\text{m}$ in the
349 nonferrous metal smelting industry of Zhuzhou, Hunan province. Tan et al., (2016) found the mass
350 concentrations of Cs and Sb were enriched in the particles smaller than $1\mu\text{m}$ in a high polluted episode
351 during winter in Beijing.

352 Figure 9a and 9b showed the mass-size distribution of Sr. Opposite to all the other elements, Sr

353 showed an obvious increasing trend with the particle sizes and higher values were concentrated in the
354 large particle sizes. Tang et al., (2018) studied the office indoor air in Nanjing and found that Sr was
355 concentrated in the particles larger than 5.8 μm . Tan et al., (2016) also found the mass concentration
356 of Sr was enriched in the particles larger than 2.7 μm in a high pollution episode during winter in
357 Beijing.

358

359 **3.6 The comparison of the water-soluble trace elements in the airborne particles with other** 360 **regions**

361 The mass concentrations of the water-soluble trace elements in the airborne particles in the Hutou
362 lung cancer epidemic village were very high, especially those having positive correlations with the
363 DNA damage. We have compared the Xuanwei data with those of the megacities of Beijing,
364 Guangzhou and Harbin in Table 6. It can be seen that the mass concentrations of both total and most
365 individual water-soluble trace elements in the PM collected in Xuanwei were much higher than those
366 in other cities. In particular, the mass concentrations of the water-soluble heavy metals Zn, Cu, Cd
367 which potentially caused the particle-induced DNA damage were significantly higher than those in
368 Beijing, Guangzhou, and Harbin.

369

370 **4. Conclusions**

371 1) The PM mass concentration in winter was greater than that in spring in the Hutou lung cancer
372 village, and PM mass first and then increases with increased particle size.

373 2) The DNA damage rate of Hutou samples was higher than that of Xize samples. The damage
374 rate of the winter sample is higher than that of the less coal-burning spring sample in Hutou village.
375 The damage rate in the smaller than 2.1 μm size range was greater than the damage rate in the large

376 size range.

377 3) The total water-soluble elements, together with the individual elements Zn, Cu, Cd, Rb, Tl,
378 Cs, and Sb, were positively associated with the oxidative potential, suggesting that these elements
379 could be a main cause for the particle-induced DNA damage. Sr had a negative correlation with the
380 DNA damage rate, implying this element might inhibit the DNA damage.

381 4) The mass concentrations of the total and individual water-soluble trace elements were mostly
382 enriched in the small particle size ranges, thus implying the indoor CDPM in the small size ranges
383 would have a higher health risk.

384 5) A comparison analysis indicates that the mass concentrations of the water-soluble heavy
385 metals Zn, Cu, Cd which potentially caused the particle-induced DNA damage were significantly
386 higher in Xuanwei area than those in Beijing, Guangzhou, and Harbin.

387 6) Pulmonary cancer initiation and promotion has been directly linked to biochemical pathways
388 of oxidative stress and DNA oxidative damage that modulates gene expression and activation of
389 transcription factors with important roles in carcinogenesis.

390

391 **Author contribution statement**

392 Xiaolei Feng: Sample, Conceptualization, Methodology, Formal analysis, Writing - Original
393 Draft. Longyi Shao: Conceptualization, Supervision, Resources, Writing - Review & Editing, Project
394 administration, Resources, Funding acquisition. Chunxiu Xi: Project administration, Sample. Tim
395 Jones: Data Analysis, Geochemistry, Editing. Daizhou Zhang: Editing. Kelly Bérubé: Data Analysis,
396 Toxicology, Editing.

397

398 **Acknowledgments**

399 This study is supported by the National Natural Science Foundation of China (Grant No.
400 41572090), the Projects of International Cooperation and Exchanges NSFC (Grant No. 41571130031)
401 and the Yueqi Scholar fund of China University of Mining and Technology (Beijing).

402

403

404 **References**

405 Al-Dabbous, A.N., Kumar, P., 2014, Number size distribution of airborne nanoparticles during
406 summertime in Kuwait: First observations from the Middle East. *Environmental Science &*
407 *Technology* 48, 13634-43.

408 Allen, A.G., Nemitz, E., Shi, J.P., Harrison, R.M., Greenwood, J.C., 2001, Size distributions of trace
409 metals in atmospheric aerosols in the United Kingdom. *Atmospheric Environment* 35, 4581-4591.

410 Ambient air quality standards, 2012.

411 Araujo, J.A., 2011, Particulate air pollution, systemic oxidative stress, inflammation, and
412 atherosclerosis. *Air Quality Atmosphere Health* 4:79–93.

413 Balakrishnan, K., Sambandam, S., Ramaswamy, P., Ghosh, S., Venkatesan, V., Thangavel, G.,
414 Mukhopadhyay, K., Johnson, P., Paul, S., Puttaswamy, N., Dhaliwal, R.S., SRU-CAR Team, 2015,
415 Establishing integrated rural-urban cohorts to assess air pollution–related health effects in pregnant
416 women, children and adults in Southern India: An overview of objectives, design and methods in
417 the Tamil Nadu Air Pollution and Health Effects (TAPHE) study. *BMJ Open*.

418 Barone-adesi, F., Chapman, R.S., Silverman, D.T., He, X., Hu, W., Vermeulen, R., Ning, B.F., Joseph,
419 F.F., Nathaniel, R., Lan, Q., 2012, Risk of lung cancer associated with domestic use of coal in
420 Xuanwei, China: retrospective cohort study. *Bmj British Medical Journal*, 345(aug29 2), e5414.

421 Brauer, M., Amann, M., Burnett, R.T., Cohen, A., Dentener, F., Ezzati, M., Henderson, S.B.,
422 Krzyzanowski, M., Martin R.V., Dingenen, R.V., Donkelaar, A.V., Thurston, G.D., 2012, Exposure
423 assessment for estimation of the global burden of disease attributable to outdoor air pollution.
424 *Environmental Science & Technology* 46, 652-660.

425 Cao, J.J., Xu, H., Xu, Q., Chen, B., Kan, H., 2012, Fine particulate matter constituents and
426 cardiopulmonary mortality in a heavily polluted Chinese city. *Environmental Health Perspectives*
427 120, 373-378.

428 Chang, L.L., 2019, Study on the trace elements and toxicology of PM_{2.5} after the action for

429 comprehensive control of air pollution. China University of Mining and Technology, Beijing. (in
430 Chinese with English Abstract)

431 Chen, G., Sun, X., Ren, H., Wan, X., Huang, H., Ma, X., Ning, B.F., Zou, X.N., Hu, W.J., Yang, G.H.,
432 2015, The mortality patterns of lung cancer between 1990–2013 in Xuanwei, China. *Lung Cancer*
433 *90*, 155-160.

434 Chow, Judith C., 2006, Health effects of fine particulate air pollution: Lines that connect. *Journal of*
435 *the Air & Waste Management Association* *56(6)*:707–708.

436 Chow, Judith C., Cao, J.J., Li, S.C., Wang, X.L., Watson, J.G., 2012, A brief history of PM_{2.5}, its
437 measurement and adverse effects. *Journal of Earth Environment* *5*, 1019-1029. (in Chinese with
438 English abstract)

439 Chuang, H.C., Jones, T.P., Lung, S.C., Bérubé K.A., 2015, Soot-driven reactive oxygen species
440 formation from incense burning. *Science of the Total Environment* *409*, 4781-4787.

441 Conibear, L., Butt, E.W., Knote C.J., Arnold, S., Spracklen, D.V., 2018, Residential energy use
442 emissions dominate health impacts from exposure to ambient particulate matter in India. *Nature*
443 *Communications*, *9*.

444 Costa, D.L., Dreher, K.L., 1997, Bioavailable transition metals in particulate matter mediate
445 cardiopulmonary injury in healthy and compromised animal models. *Environmental Health*
446 *Perspectives* *105*: 1053–1060.

447 Department for Environment, Food and Rural Affairs (DEFRA). Air Pollution in the UK (2017)

448 Duan, J., Tan, J., Wang, S., Hao, J., Chai, F.H., 2012, Size distributions and sources of elements in
449 particulate matter at curbside, urban and rural sites in Beijing. *Journal of Environmental Sciences*,
450 *24*, 87–94.

451 Fan, J.S., Shao, L.Y., Li, Z.X., Hu, Y., Hou, C., 2013, Mass concentration distribution of inhalable
452 particulates in different villages Xuanwei county, China. *Applied Mechanics and Materials* *295-*
453 *298*, 539-542.

454 Finkelman R.B., Tian, L.W., 2018, The health impacts of coal use in China. *International Geology*

455 *Review, 60, 579-589.*

456 Gilli, G., Traversi, D., Rovere, R., Pignata, C., Schilirò T., 2007, Chemical characteristics and
457 mutagenic activity of PM₁₀ in Torino, a Northern Italian City. *Science of the Total Environment*
458 385, 97-107.

459 Greenwell, L. L., Moreno, T., Jones, T. P., Richards, R. J., 2002, Particle-induced oxidative damage
460 is ameliorated by pulmonary antioxidants. *Free Radical Biology and Medicine* 32, 898-905.

461 Hamra, G.B., Guha, N., Cohen, A., Laden, F., Raaschou-Nielsen, O., Samet, J.M., Vineis, P.,
462 Forastiere, F., Saldiva, P., Yorifuji, T., Loomis, D., 2014, Outdoor particulate matter exposure and
463 lung cancer: a systematic review and meta-analysis. *Environmental health perspectives* 122, 906-
464 911

465 Hao, J., Zhang, F., Chen, D., Liu, Y., Liao, L., Shen, C., Liu, T.Y., Liao, J.L., Ma, L., 2019, Association
466 between ambient air pollution exposure and infants small for gestational age in Huangshi, China:
467 a cross-sectional study. *Environmental Science and Pollution Research* 26, 32029-32039

468 He X.Z., Yang R.D., 1994, Yunnan Science and Technology Press.

469 He, X., Chapman, R.S., Yang, R., Cao, S., Mumford, J.L., Liang, C., 1987, Lung cancer and indoor
470 air pollution in Xuanwei, China: current progress. *Science* 235, 217-220.

471 He, X.Z., Yang, R.D., 1994, Lung Cancer and Indoor Air Pollution from Coal Burning. Yunnan
472 Science and Technology Publishing House, Kunming (in Chinese).

473 Hong Y, Ma Y J, Li C L, Liu N W, Gao S P, Zhang Y H, 2011. Elemental size distribution
474 characteristics of atmospheric particles on hazy days during winter in Shenyang. *Research of*
475 *Environmental Sciences, 24(6), 637–643.* (in Chinese with English abstract)

476 Hosgood, H.D., Chapman, R.S., He, X.Z., Hu, W., Tian, L.W., Liu, L.Z., 2013, History of lung disease
477 and risk of lung cancer in a population with high household fuel combustion exposures in rural
478 China. *Lung Cancer* 81: 343–346.

479 Hou, C., Shao, L.Y., Wang, J., Liu, J.X., Zhao, C.M., Geng, C.M., 2016, Distribution of trace elements
480 in inhalable particulate matter emitted from coal burning. *Journal of China Coal Society* 41:760 -

481 768. (in Chinese with English abstract)

482 Hu Y., 2016, Domestic Coal Combustion Emissions and the Lung Cancer Epidemic in Xuanwei,
483 China. China University of Mining and Technology, Beijing. (in Chinese with English Abstract)

484 International Agency for Research on Cancer (IARC). Air pollution and Cancer. Vol 161, 2012.

485 International Agency for Research on Cancer (IARC). World Health Organization, 2017.

486 Kim, K. H., Kabir, E., Kabir, S., 2015, A review on the human health impact of airborne particulate
487 matter. *Environment International* 74, 136-143.

488 Large, D.J., Kelly, S., Spiro, B., Tian, L.W., Shao, L.Y., Finkelman, R., Zhang, M.Q., Somerfield, C.,
489 Plint, S., Ali, Y., Zhou, Y.P., 2009, Silica-volatile interaction and the geological cause of the
490 Xuanwei lung cancer epidemic. *Environmental Science and Technology* 43, 9016-9021.

491 Li, P., Xin, J.Y., Wang, Y.S., Wang, S.G., Shang, K.Z., Liu, Z.R., Li, G.X., Pan, X.C., Wei, L.B., Wang,
492 M.Z., 2013, Time-series analysis of mortality effects from airborne particulate matter size fractions
493 in Beijing. *Atmospheric Environment* 81, 253-262.

494 Li, P., Xin, J.Y., Wang, Y.S., Wang, S.G., Li, G.X., Pan, X.C., Liu, Z.R., Wang, L.L., 2013, The acute
495 effects of fine particles on respiratory mortality and morbidity in Beijing, 2004-2009.
496 *Environmental Science and Pollution Research* 20, 6433-6444.

497 Li, Y.W., Shao, L.Y., Wang, W.H., Zhang, M.Y., Feng, X.L., Li, W.J., Zhang, D.Z., 2020. Airborne
498 fiber particles: Types, size and concentration observed in Beijing, *Science of The Total Environment*
499 705, 135967.

500 Liu, Y.F., 2010, PM₁₀ and PM_{2.5} in Harbin Air: Physicochemistry and Bioreactivity. China University
501 of Mining and Technology, Beijing. (in Chinese with English Abstract)

502 Lui, K.H., Bandowe, B.A.M., Tian, L.W., Chan, C.S., Cao, J.J., Ning, Z., Lee, S.C., Ho, K.F., 2017,
503 Cancer risk from polycyclic aromatic compounds in fine particulate matter generated from
504 household coal combustion in Xuanwei, China. *Chemosphere* 169, 660–668.

505 Lv, Y., Zhang, K., Chai, F.H., Cheng, T.T., Yang, Q., Zheng, Z.L., Li, X., 2017, Atmospheric size-
506 resolved trace elements in a city affected by nonferrous metal smelting: Indications of respiratory

507 deposition and health risk. *Environmental Pollution* 224, 559-571.

508 Ma, L., Dadashazar, H., Braun, R.A., MacDonald, A.B. Aghdam, M.A., Maudlin, L.C., Sorooshian,
509 A., 2019, Size-resolved characteristics of water-soluble particulate elements in a coastal area:
510 Source identification, influence of wildfires, and diurnal variability. *Atmospheric Environment* 206,
511 72-84.

512 Merolla, L., Richards, R. J., 2005, In vitro effects of water-soluble metals present in UK particulate
513 matter. *Experimental Lung Research* 31, 671-683

514 Moreno, T., Merolla, L., Gibbons, W., Greenwell, L., Jones, T., Richards, R., 2004, Variations in the
515 source, metal content and bioreactivity of technogenic aerosols: a case study from Port Talbot,
516 Wales, UK. *Science of the Total Environment* 333, 59-73.

517 Olawoyin, R., Schweitzer, L., Zhang, K., Okareh, O., Slates, K., 2018, Index analysis and human
518 health risk model application for evaluating ambient air–heavy metal contamination in Chemical
519 Valley Sarnia. *Ecotoxicology and Environmental Safety* 148, 72-81.

520 Pan, Y.P., Tian, S.L., Li, X.R., Sun, Y., Li, Y., Wentworth, G.R., Wang, Y.S, 2015, Trace elements in
521 particulate matter from metropolitan regions of northern China: Sources, concentrations and size
522 distributions. *Science of The Total Environment* 537, 9-22.

523 Pope, C.A., Turner, M.C., Burnett, R.T., Jerrett, M., Gapstur, S.M., Diver, W.R., Krewski, D., Brook,
524 R. D., 2015, Relationships between fine particulate air pollution, cardiometabolic disorders, and
525 cardiovascular mortality novelty and significance. *Circulation Research* 116, 108.

526 Qiao, Y.S., 2011, Study on the physicochemistry and toxicity of inhalable parriculates in the Central
527 China City Group. China University of Mining and Technology, Beijing. (in Chinese with English
528 Abstract)

529 Rohra, H., Tiwari, R., Khare, P., Taneja, A., 2018, Indoor-outdoor association of particulate matter
530 and bounded elemental composition within coarse, quasi-accumulation and quasi-ultrafine ranges
531 in residential areas of northern India. *Science of the Total Environment* 631-632, 1383-1397.

532 Rönkkö T. J., Jalava, P. I., Happonen, M. S., Stefanie, K., Olli, S., Ari, L., et al., 2018, Emissions and

533 atmospheric processes influence the chemical composition and toxicological properties of urban
534 air particulate matter in Nanjing, China. *Science of The Total Environment* 639, 1290-1310.

535 R ckerl, Regina, Schneider, A., Breitner, S., Cyrys, J., Peters, A., 2011, Health effects of particulate
536 air pollution: A review of epidemiological evidence. *Inhalation Toxicology* 23, 555-592.

537 Schneidmesser, E.V., Stone, E.A., Quraishi, T.A., Shafer, M. M., Schauer, J.J., 2010, Toxic metals
538 in the atmosphere in Lahore, Pakistan. *Science of the Total Environment* 408, 1640-1648.

539 Schleicher, N., Norra, S., Dietze, V., Yu, Y., Fricker, M., Kaminski, U., Chen, Y., Cen, K., 2011, The
540 effect of mitigation measures on size distributed mass concentrations of atmospheric particles and
541 black carbon concentrations during the Olympic Summer Games 2008 in Beijing. *Science of the*
542 *Total Environment* 412-413, 185-193.

543 Seow, W.J., Hu, W., Vermeulen, R., Iii, H. H., Sdownward, G., Schapman, R., He, X.Z., ABassig, B.,
544 Kim, C., Wen, C.J., Rothman, N., Lan, Q., 2014, Household air pollution and lung cancer in China:
545 a review of studies in Xuanwei. *Chinese Journal of Cancer* 33, 471-475.

546 Shah ASV, Langrish J.P., Nair H., McAllister D.A., Hunter A.L., Donaldson K., Newby, D.E., Mills,
547 N.M., 2013, Global association of air pollution and heart failure: a systematic review and meta-
548 analysis. *The Lancet* 382, 1039-48.

549 Shao, L.Y., Hu, Y., Wang, J., Hou, C., Yang, Y., Wu, M., 2013, Particle-induced oxidative damage of
550 indoor PM₁₀ from coal burning homes in the lung cancer area of Xuanwei, China. *Atmospheric*
551 *Environment* 77, 959-967.

552 Shao, L.Y., Wang, J., Hou, H.H., Zhang, M.Q., Wang, H., Spiro, B., Large, D., Zhou, Y.P., 2015,
553 Geochemistry of the C₁ coal of the latest Permian during mass extinction in Xuanwei, Yunnan.
554 *Acta Geological sinica* 89, 163-179. (in Chinese with English abstract)

555 Shao, L.Y., Hou, C., Geng, C.M., Liu, J.X., Hu, Y., Wang, J., Jones, Tim, Zhao, C.M., B ruB , K.,
556 2016, The oxidative potential of PM₁₀ from coal, briquettes and wood charcoal burnt in an
557 experimental domestic stove. *Atmospheric Environment* 127, 372-381.

558 Shao, L.Y., Hu, Y., Shen, R.R., Sch fer, K., Wang, J., Wang, J.Y., Schnelle-Kreis, J., Zimmermann,

559 R., Bérubé, K., Suppan, P., 2017, Seasonal variation of particle-induced oxidative potential of
560 airborne particulate matter in Beijing. *Science of The Total Environment* 579, 1152-1160.

561 Slama, A., Sliwczynski, A., Woznica, J., Zdrolik, M., Wisnicki, B., Kubajek, J., Thurzańska-
562 Wieczorek, O., Gozdowski, D., Wierzba, W., Franek, E., 2019, Impact of air pollution on hospital
563 admissions with a focus on respiratory diseases: a time-series multi-city analysis. *Environmental*
564 *Science and Pollution Research* 26, 16998-17009

565 Sridevi, J., Gurdeep, S., 2017, Human health risk assessment of airborne trace elements in Dhanbad,
566 India. *Atmospheric Pollution Research* 8, 490-502.

567 Sun, Z.Q., Shao, L.Y., Mu, Y., Hu, Y., 2014, Oxidative capacities of size-segregated haze particles in
568 a residential area of Beijing. *Journal of Environmental Sciences* 26, 167-174.

569 Tan, J.H., Duan, J.C., Zhen, N.J., He, K.B., Hao, J.M., 2016, Chemical characteristics and source of
570 size-fractionated atmospheric particle in haze episode in Beijing. *Atmospheric Research* 167, 24-
571 33.

572 Tang, Z.J., Hu, X., Qiao, J.Q., Lian, H.Z., 2018, Size distribution, bioaccessibility and health risks
573 of indoor/outdoor airborne toxic elements collected from school office room. *Atmosphere* 9, 340-
574 353.

575 Tao, J., Zhang, L.M., Zhang, R.J., Wu, Y.F., Zhang, Z.S., Zhang, X.L., Tang, Y.X., Cao, J.J., Zhang,
576 Y.H., 2016, Uncertainty assessment of source attribution of PM_{2.5} and its water-soluble organic
577 carbon content using different biomass burning tracers in positive matrix factorization analysis-a
578 case study in Beijing, China. *Science of the Total Environment* 543, 326-335.

579 Tian, L.W., Lucas, D., Fischer, S.L., Lee, S.C., Hammond, S.K., Koshland, C.P., 2008, Particle and
580 gas emissions from a simulated coal-burning household fire pit. *Environmental Science and*
581 *Technology* 42, 2503-2508.

582 Thurston, G.D., Burnett, R.T., Turner, M.C., Shi, Y., Krewski, D., Lall, R., Ito, K., Jerrett, M., Gapstur,
583 S.M., Diver, W.R., Pope, C.A., 2016, Ischemic heart disease mortality and long-term exposure to
584 source-related components of U.S. fine particle air pollution. *Environmental Health Perspectives*

585 124, 785-794.

586 Wang, J., Shao, L.Y., Wang, H., Spiro, B., Large, D.J., 2018, SHRIMP zircon U-Pb ages from coal
587 beds across the Permian-Triassic boundary, eastern Yunnan, southwestern China. *Journal of*
588 *Palaeogeography-ENGLISH* 7(2): 117-129.

589 Wang, Q.X., Tan, Z.Y., Zhao, H., Li, J. H., Tian, L. W., Wang, Q.Y., 2017, Species of iron in size-
590 resolved particle emitted from Xuanwei coal combustion and their oxidative potential.
591 *Environmental Science* 38, 2273-2279. (in Chinese with English abstract)

592 Wang, Y., Peng, L., Li, L.J., Wang, Y.X., Zhang, T., Liu, H.L., Mu, L., 2016, Chemical Compositions
593 and Sources Apportionment of Re-suspended Dust in Jincheng. *Environmental science* 37, 82-87.
594 (in Chinese with English abstract)

595 Wang, W.H., Shao, L.Y., L, J., Chang, L.L., Zhang, D.Z., Zhang, C.C., Jiang, J.K., 2019,
596 Characteristics of individual particles emitted from an experimental burning chamber with coal
597 from the lung cancer area of Xuanwei, China. *Aerosol and Air Quality Research* 19, 355–363.

598 Wang, T., Rovira, J., Sierra, J., Chen, S.J., Mai, B.X., Schuhmacher, M., Domingo, J.L., 2019,
599 Characterization and risk assessment of total suspended particles (TSP) and fine particles (PM_{2.5})
600 in a rural transformational e-waste recycling region of Southern China. *Science of the Total*
601 *Environment* 692, 432-440.

602 Xing, J.P., Shao L.Y., Zhang, W. B., Peng, J.F., Wang, W.H., Shuai, S.J., Hu, M., Zhang, D.Z., 2020,
603 Morphology and size of the particles emitted from a gasoline-direct-injection-engine vehicle and
604 their ageing in an environmental chamber. *Atmos. Chem. Phys.* 20, 2781–2794.

605 Xu, H. H., Wang, Y. S., Wen, T. X., He, X. X., 2007. Size distributions and vertical distributions of
606 metal elements of atmospheric aerosol in Beijing. *Environmental Chemistry*, 26(5), 675 - 679.

607 Xue, X., Chen, J., Sun, B., Zhou, B., Li, X., 2018, Temporal trends in respiratory mortality and short-
608 term effects of air pollutants in Shenyang, China. *Environmental Science and Pollution Research*
609 25, 11468-11479.

610 Yang, K., Huang, Y., Zhao, G., Lei, Y., Wang, K., 2010, The expression of PAH-DNA adducts in lung

611 tissues of Xuanwei female lung cancer patients. *Sino-German Journal of Clinical Oncology:*
612 *English Edition 09*, 497 – 501.

613 Zelikoff, J.T., Schermerhorn, K.R., Fang, K., Cohen, M.D., Schlesinger, R.B., 2002, A role for
614 associated transition metals in the immunotoxicity of inhaled ambient particulate matter.
615 *Environmental Health Perspectives 110*, 871-875.

616 Zhang, R.C., Hao, X.J., Zhang, W.C., Liu, P.W., Ma, J., Shang, Y., Wu, M.H., Lv, S.L., 2016,
617 Distribution of PAHs in size-resolved particles emitted from Xuanwei C1 coal combustion and
618 their health risk assessment. *Asian Journal of Ecotoxicology 11*, 580-585.

619 Zhou, X.T., He, X.Z., 2006, The influence of indoor air pollution on chronic obstructive pulmonary
620 disease (COPD). *China Environmental Science 26*, 591-594. (in Chinese with English abstract)

621

622

623 **Figures**

624 Figure 1 A sketch showing principle of the plasmid scission assay

625 Figure 2 Mass concentrations of particulate matter with different particle sizes in Hutou and Xize
626 villages in winter and spring

627 Figure 3 Particle mass concentration with small and large size ranges in Hutou and Xize villages

628 Figure 4 Mass-size distributions of water-soluble trace element Zn with different particle sizes in
629 Hutou and Xize villages in winter and spring

630 Figure 5 Mass-size distributions of water-soluble trace element Cu with different particle sizes in
631 Hutou and Xize villages in winter and spring

632 Figure 6 Mass-size distributions of water-soluble trace element Cd with different particle sizes in
633 Hutou and Xize villages in winter and spring

634 Figure 7 Mass-size distributions of water-soluble trace elements Rb(a) and Tl(b) with different
635 particle sizes in Hutou and Xize villages in winter and spring

636 Figure 8 Mass-size distributions of water-soluble trace elements Cs(a) and Sb(b) with different
637 particle sizes in Hutou and Xize villages in winter and spring

638 Figure 9 Mass-size distributions of water-soluble trace element Sr with different particle sizes in
639 Hutou and Xize villages in winter and spring

640

641

642 **Tables**

643 Table 1 Meteorological conditions during the sampling periods in Hutou and Xize villages in winter
644 and spring.

645 Table 2 The PM mass concentration ($\mu\text{g}/\text{m}^3$) of size-segregated particles in Hutou and Xize villages
646 in winter and spring.

647 Table 3 Plasmid DNA damage rate induced by size-segregated airborne particles at $250\mu\text{g}/\text{mL}$ for
648 samples in Hutou and Xize villages in winter and spring.

649 Table 4 Correlation between water-soluble trace elements ($\mu\text{g}/\text{g}$) and DNA damage rate at $250\mu\text{g}/\text{mL}$
650 for the size-segregated particles in Hutou and Xize villages in winter and spring.

651 Table 5 The total water-soluble element mass concentration (ng/m^3) of size-segregated particles in
652 Hutou and Xize villages in winter and spring.

653 Table 6 The comparison of the water-soluble trace elements in different area (Unit: ng/m^3)

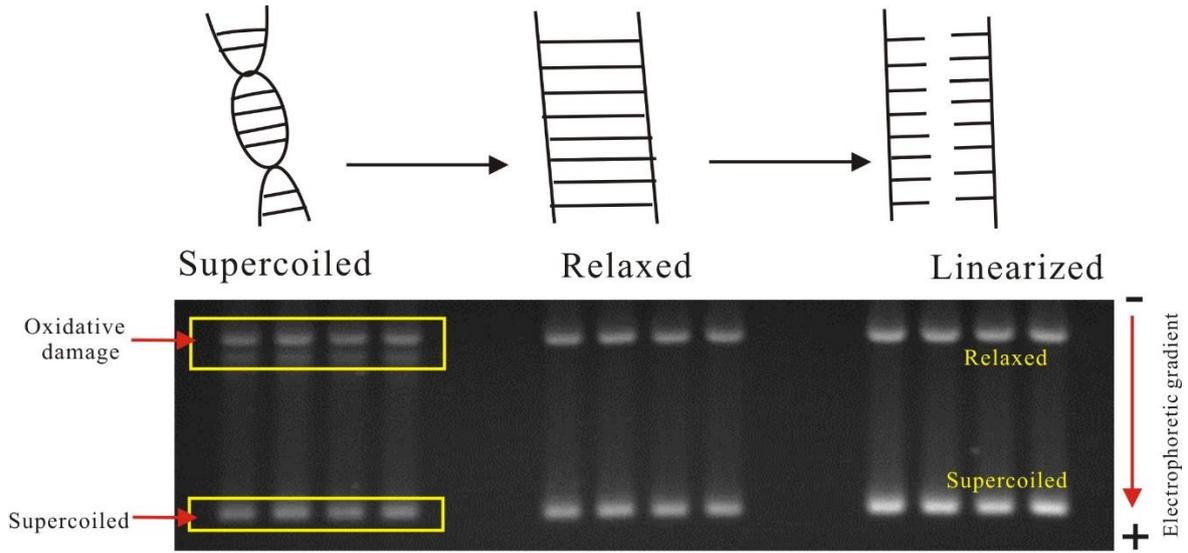
654

655

656

657

658



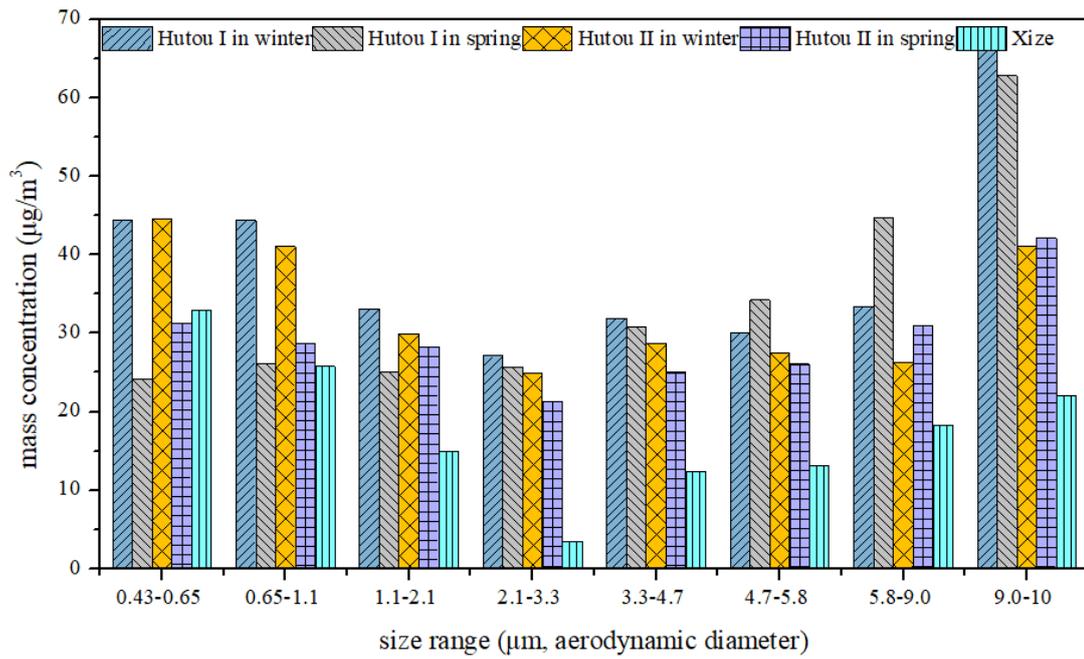
660

661

662

663

Fig. 1. DNA Plasmid Scission Assay



664

665

666

667

668

Fig. 2 Mass concentrations of particulate matter with different particle sizes in Hutou and Xize villages in winter and spring.

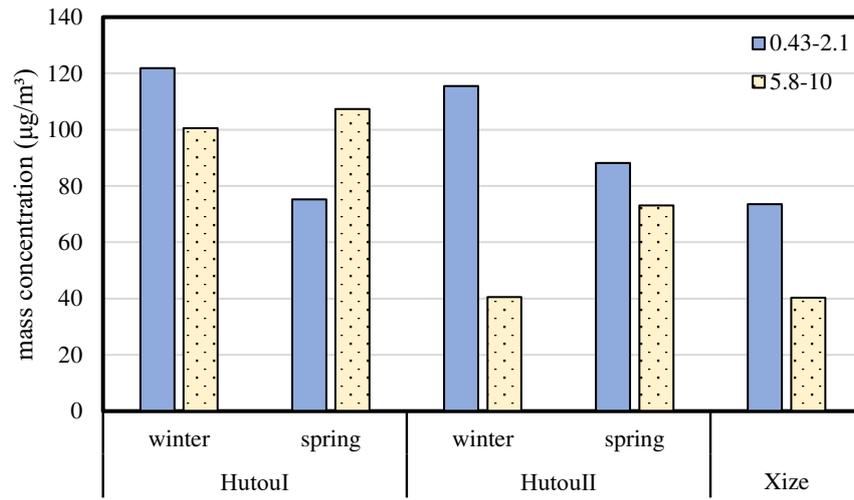
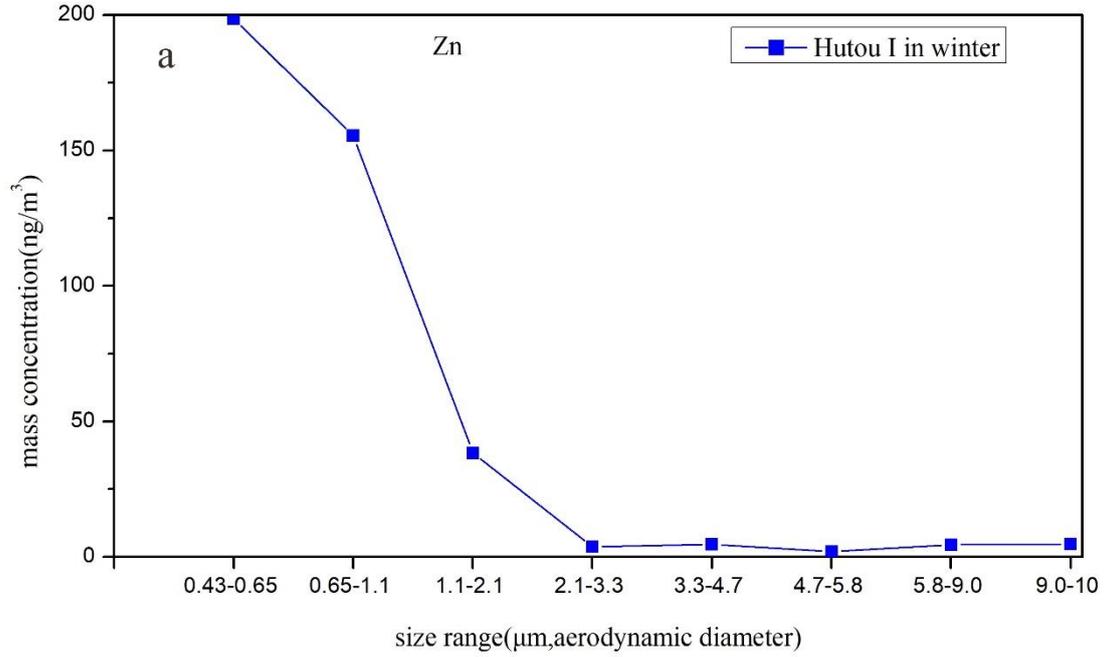


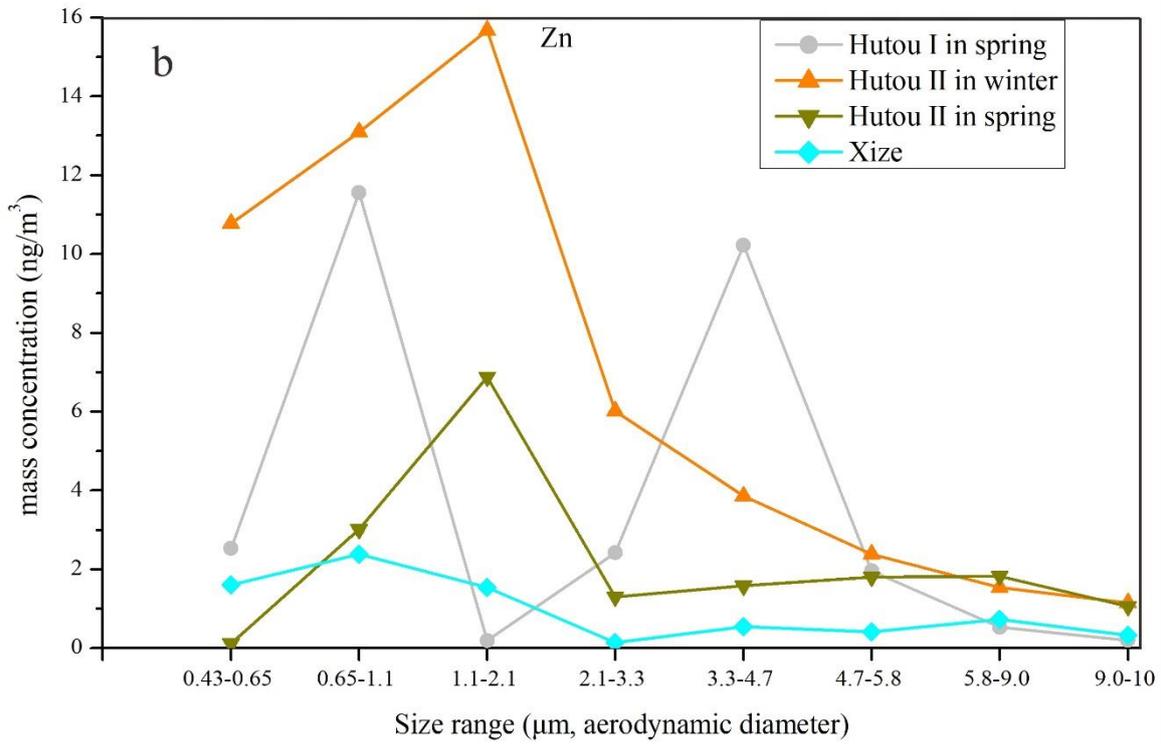
Fig. 3 Particle mass concentration with small and large size ranges in Hutou and Xize villages

669
 670
 671
 672
 673
 674
 675
 676
 677
 678
 679
 680
 681
 682
 683
 684
 685
 686
 687
 688
 689
 690
 691
 692
 693
 694

695
696



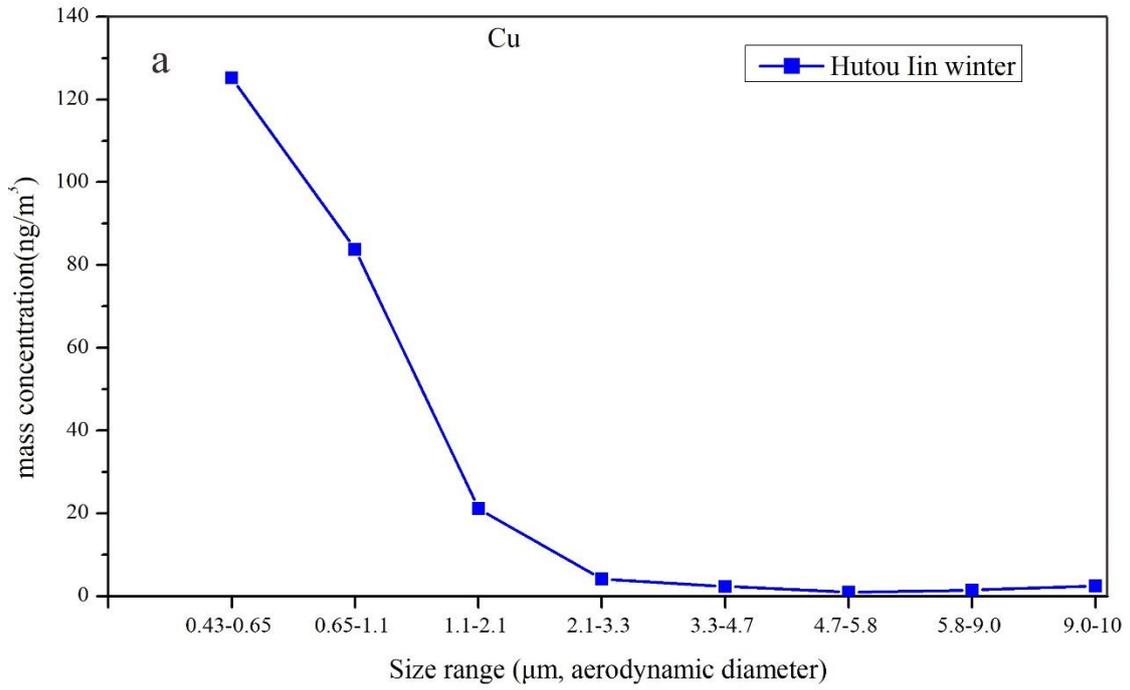
697
698
699
700



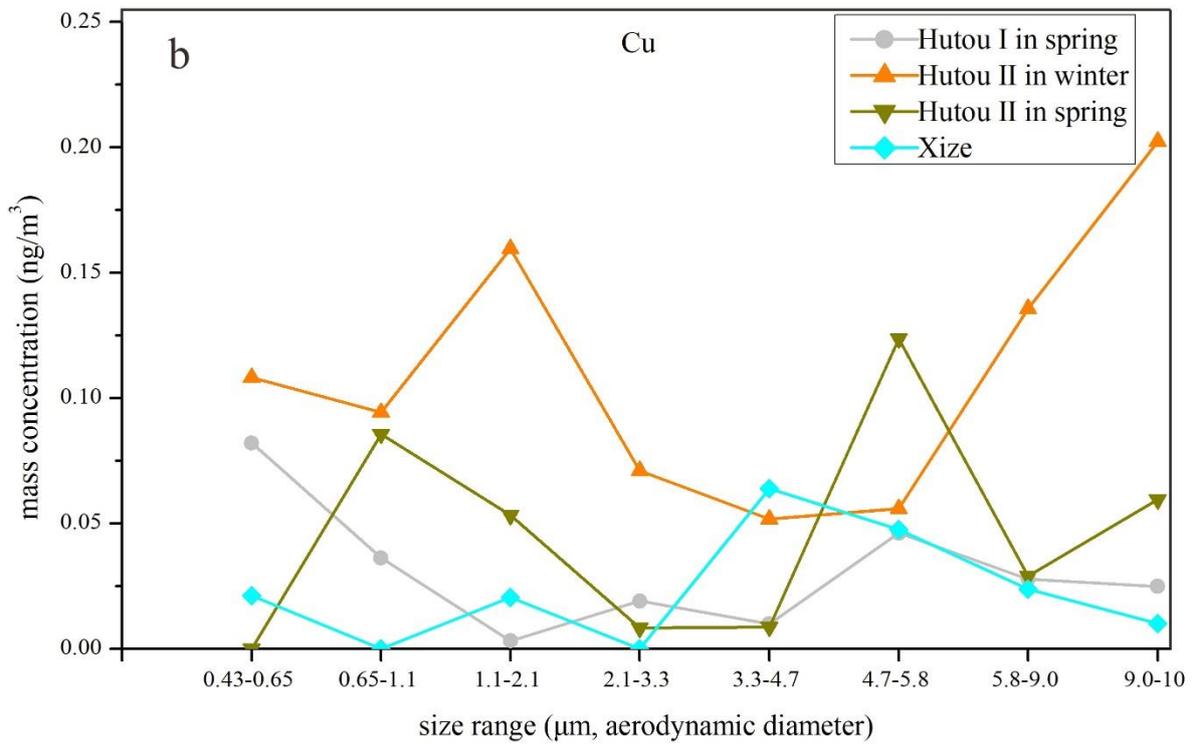
701
702
703
704
705
706
707
708
709

Fig. 4 Mass-size distributions of water-soluble trace element Zn with different particle sizes in Hutou and Xize villages in winter and spring

710
711
712



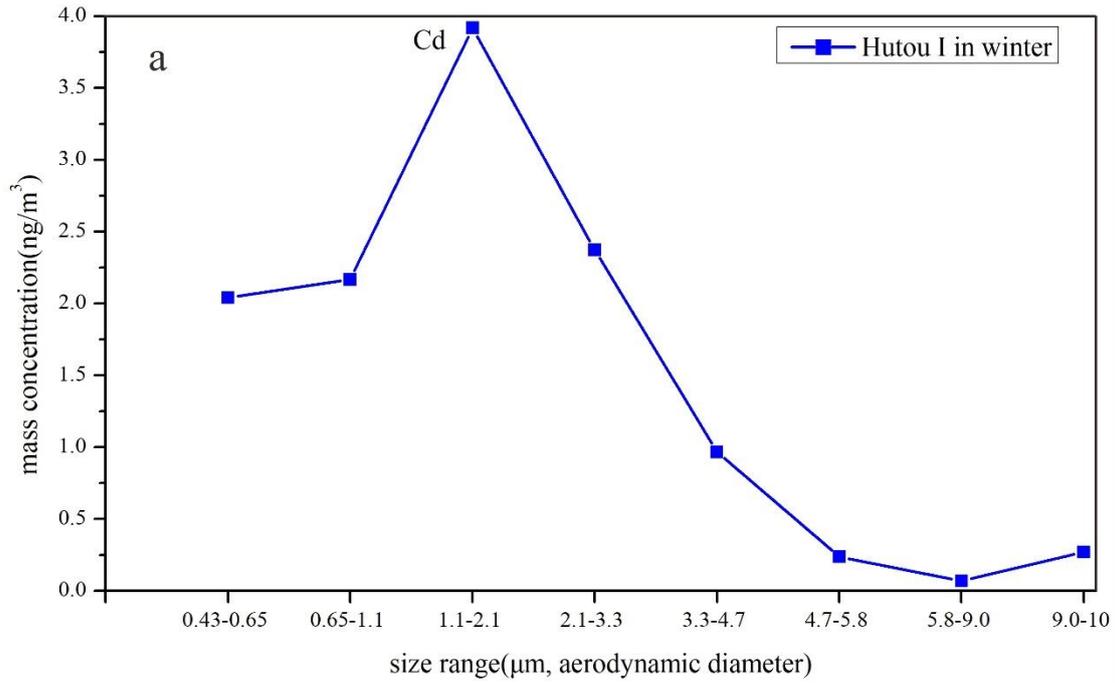
713
714
715
716



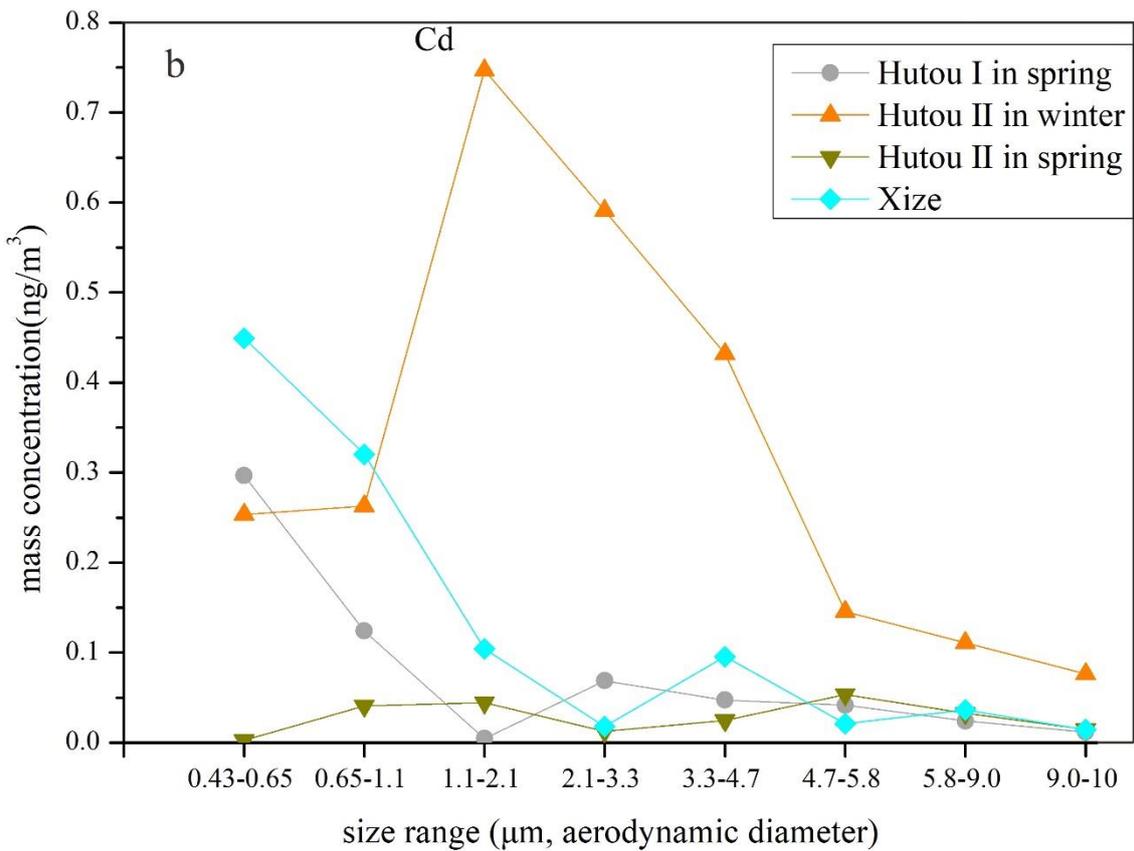
717
718
719
720
721
722

Fig. 5 Mass-size distributions of water-soluble trace element Cu with different particle sizes in Hutou and Xize villages in winter and spring

723



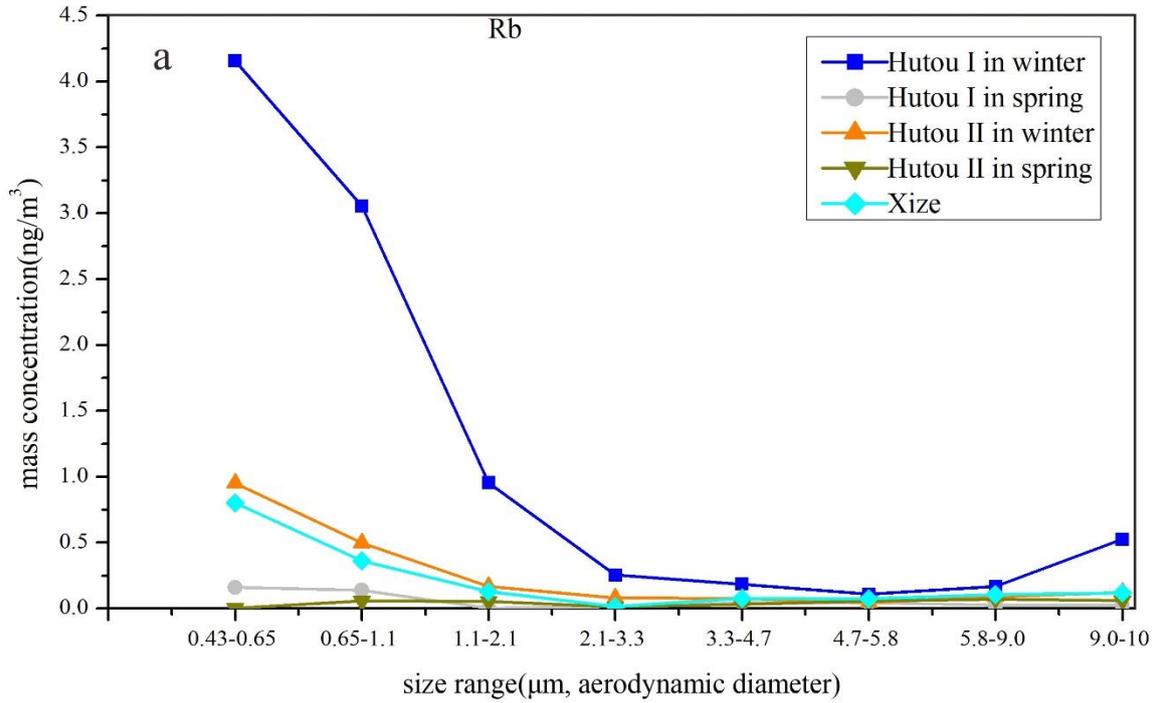
724
725
726



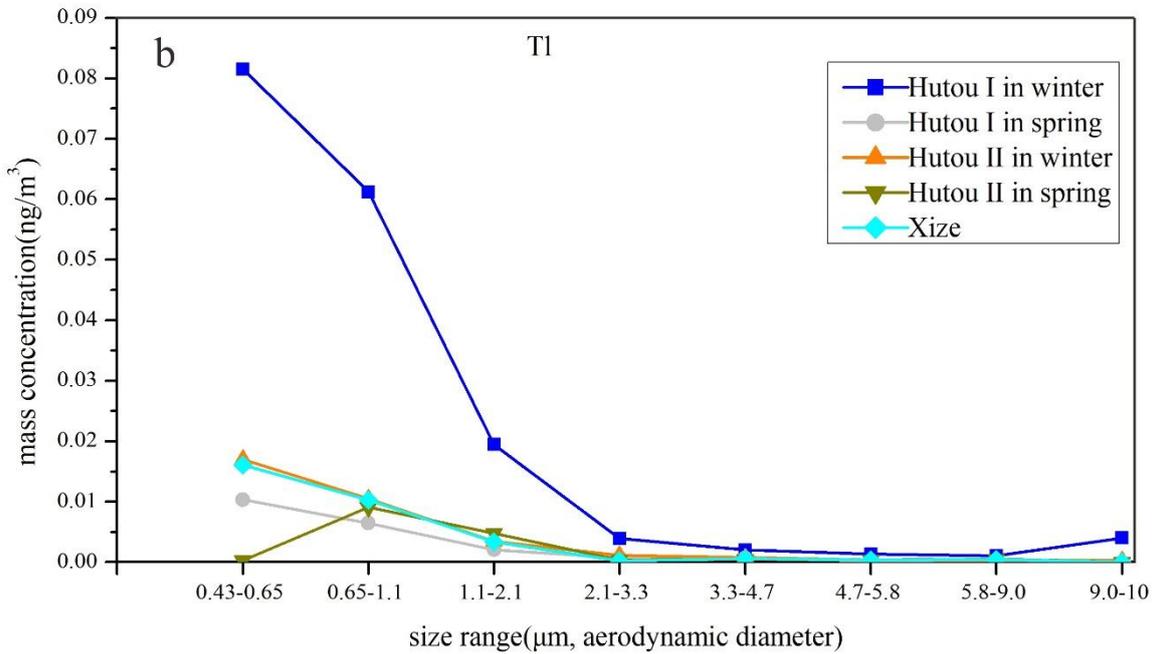
727
728
729
730
731
732

Fig. 6 Mass-size distributions of water-soluble trace element Cd with different particle sizes in Hutou and Xize villages in winter and spring

733
734
735

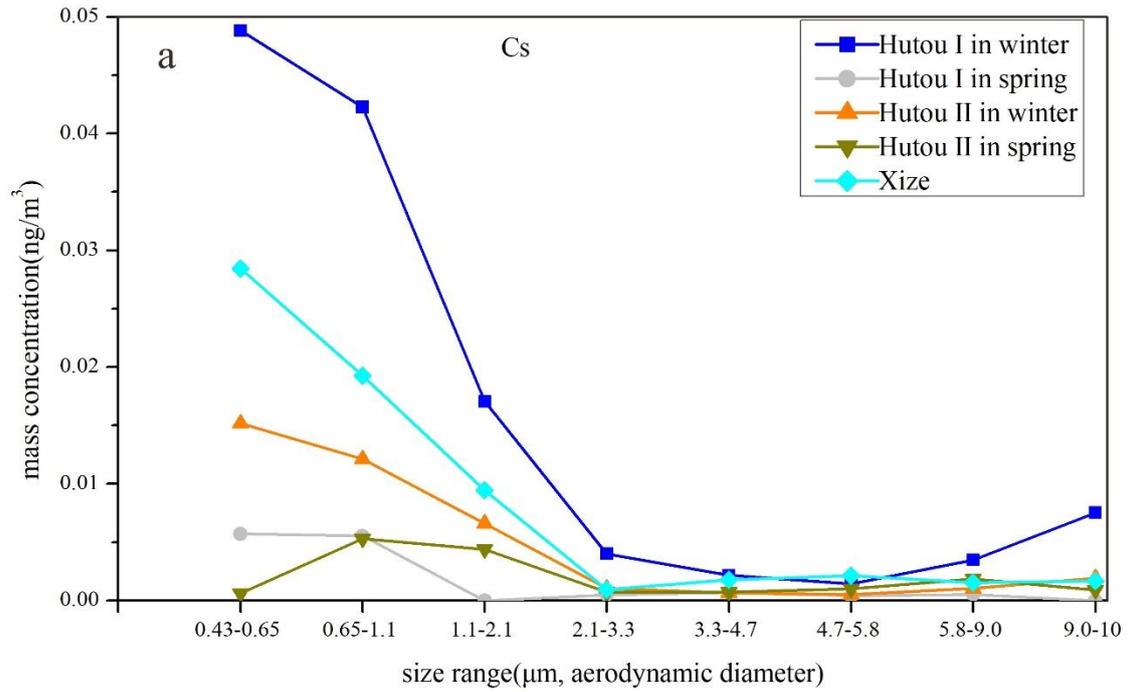


736
737
738
739

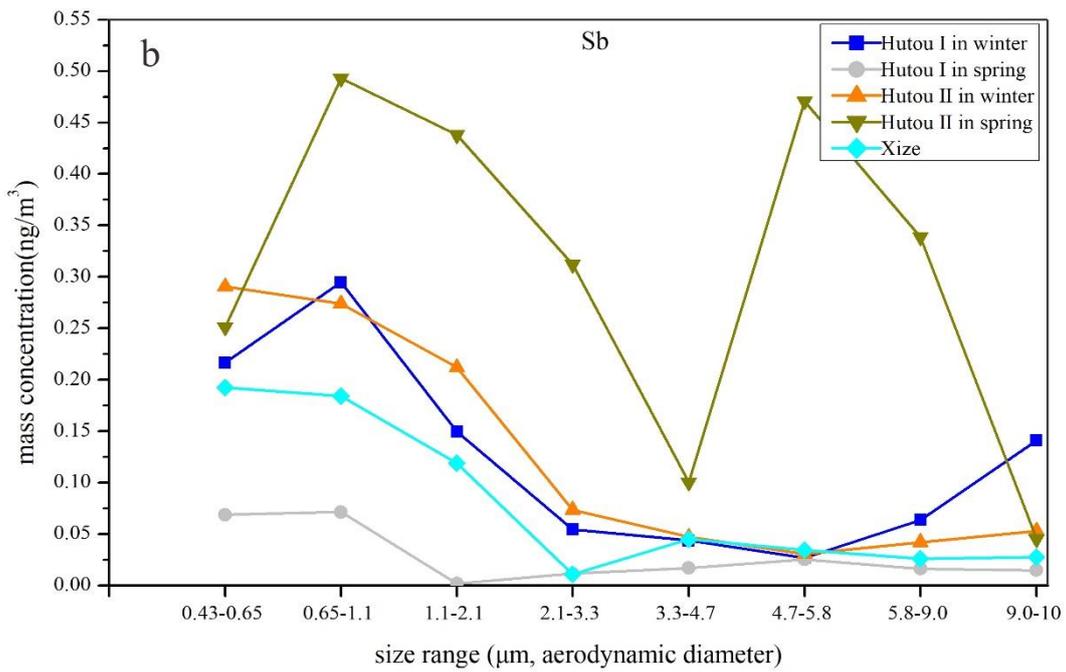


740
741
742
743
744
745
746
747
748

Fig. 7 Mass-size distributions of water-soluble trace elements Rb and Tl with different particle sizes in Hutou and Xize villages in winter and spring

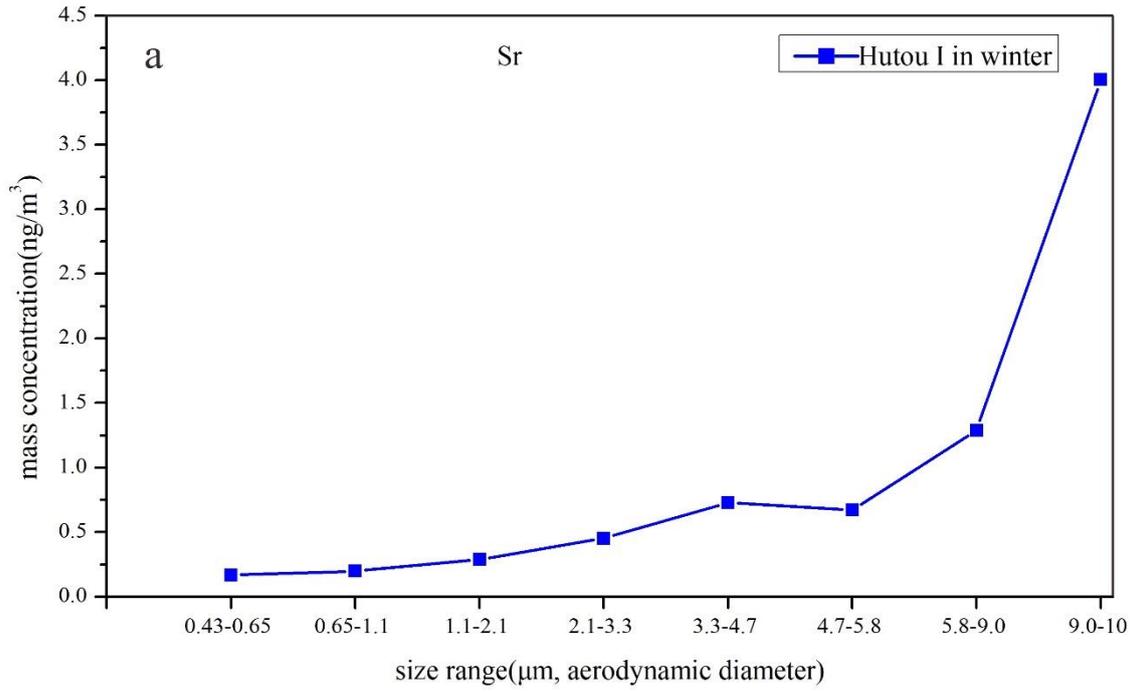


750
751
752
753

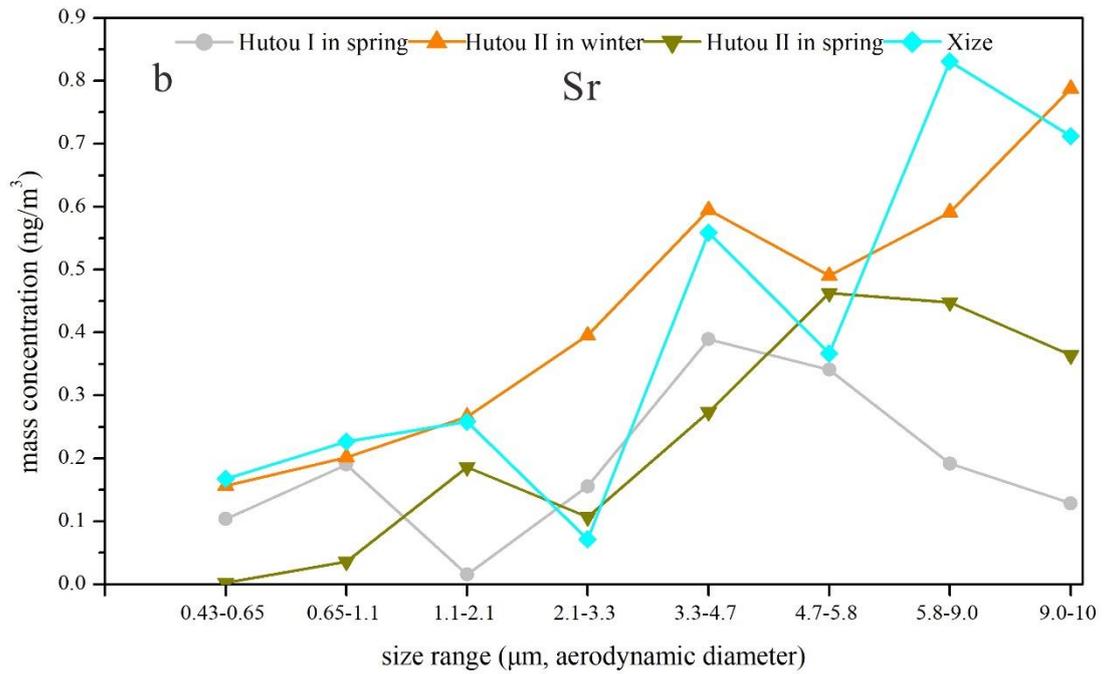


754
755
756
757
758
759
760
761
762
763

Fig. 8 Mass-size distributions of water-soluble trace elements Cs and Sb with different particle sizes in Hutou and Xize villages in winter and spring



765
766
767
768
769



770
771
772
773
774

Fig. 9 Mass-size distributions of water-soluble trace element Sr with different particle sizes in Hutou and Xize villages in winter and spring

775
776
777
778
779
780
781

Table 1 Meteorological condition during the sampling periods in Hutou and Xize villages in winter and spring

Number	Sampling period	Sampling times	Average temperature(°C)	Relative humidity (%)	Average pressure(hPa)	Sample sites
1	2016.12.20- 2016.12.22	9:00am- 8:00am	14.9	59.4	805.3	Hutou I
2	2016.12.24- 2016.12.26	9:00am- 8:00am	12.1	65.2	801.8	Hutou II
3	2016.12.26.- 2016.12.28	10:00am- 8:00am	13.4	56.3	814.2	Xize
A	2019.3.1- 2019.3.3	8:00am- 8:00am	14.6	54.5	818.5	Hutou I
B	2019.3.3- 2019.3.5	8:00am- 8:00am	14.0	45.3	816.5	Hutou II

782

783
784
785
786
787

Table 2 The PM mass concentration ($\mu\text{g}/\text{m}^3$) of size-segregated particles in Hutou and Xize villages in winter and spring

Particulate sizes (μm)	Hutou I		Hutou II		Xize
	winter	spring	winter	spring	
0.43-0.65	44.39	24.17	44.56	31.31	32.93
0.65-1.1	44.39	26.12	41.05	28.69	25.74
1.1-2.1	33.09	24.98	29.87	28.23	14.88
2.1-3.3	27.1	25.63	24.86	21.29	3.51
3.3-4.7	31.92	30.82	28.7	24.99	12.37
4.7-5.8	30.09	34.23	27.53	26.07	13.04
5.8-9.0	33.42	44.61	26.2	31.01	18.22
9.0-10	67.17	62.78	41.05	42.11	22.06
total	311.57	273.32	263.83	233.7	142.75

788
789
790
791
792
793

Table 3 Plasmid DNA damage rate induced by size-segregated airborne particles at $250\mu\text{g}/\text{mL}$ for samples in Hutou and Xize villages in winter and spring

Particulate size (μm)	Hutou I (%)		Hutou II (%)		Xize (%)	blank
	winter	spring	winter	spring		
0.43-0.65	47.97	31.68	40.39	38.21	30.15	
0.65-1.1	49.43	35.72	42.47	33.04	33.23	
1.1-2.1	42.31	34.7	37.68	27.03	32.63	
2.1-3.3	38.12	31.34	36.28	28.91	32.01	
3.3-4.7	33.76	25.18	33.26	31.73	30.28	< 10
4.7-5.8	29.97	26.38	35.89	31.32	30.07	
5.8-9.0	33.67	27.43	33.07	37.42	22.06	
9.0-10	31.92	31.66	31.83	33.15	21.63	

794
795
796
797
798
799
800

801
802
803
804
805
806
807
808
809
810
811
812
813
814
815
816
817
818
819
820
821
822
823
824

Table 4 Correlation between water-soluble trace elements ($\mu\text{g/g}$) and DNA damage rate at $250\mu\text{g/mL}$ for the size-segregated particles in Hutou and Xize villages in winter and spring

The type of elements	total	Rb	Tl	Zn	Cu	Cs	Cd	Sb
Correlation coefficient	0.649**	0.690**	0.681**	0.679**	0.639**	0.574**	0.560**	0.510**
The type of elements	Pb	Sr	Sc	La	Ga	Y	Mn	Ti
Correlation coefficient	0.108	- 0.418**	-0.272	-0.262	-0.252	-0.240	-0.200	-0.198
The type of elements	V	Mo	Cr	Be	Nd	Li	Ba	Co
Correlation coefficient	-0.146	-0.161	-0.160	-0.118	-0.058	-0.044	-0.037	-0.031

** Significantly correlated at the 0.01 level (both sides).

Table 5 The total water-soluble element mass concentration (ng/m^3) of size-segregated particles in Hutou and Xize villages in winter and spring

Particulate sizes (μm)	Hutou I		Hutou II		Xize
	winter	spring	winter	spring	
0.43-0.65	333.83	6.45	16.53	0.45	19.47
0.65-1.1	249.12	15.44	18.51	6.15	21.69
1.1-2.1	68.57	1.71	21.47	10.56	17.9
2.1-3.3	15.4	4.06	10.14	2.62	2.93
3.3-4.7	13.44	14.61	8.95	3.55	12.57
4.7-5.8	9.02	4.77	6.26	5.8	10.46
5.8-9.0	21.31	1.71	7.06	6.06	6.43
9.0-10	44.15	0.53	8.74	2.88	4.63
total	754.84	49.29	97.66	38.07	96.08

Table 6 The comparison of the water-soluble trace elements in different area (Unit: ng/m³)

Water-soluble elements	Harbin	Guangzhou-electronic-waste recycling region*	Guangzhou-Background site*	Beijing*	Beijing	Hutou	Hutou
	Liu (2010)	Wang et al., (2019)	Wang et al., (2019)	Tao et al., (2016)	Chang (2019)	Hu et al., (2016)	this study
Zn	318	----	----	270	140	98	412
Cd	1	7	3	3	1	9	12
Mn	33	22	30	78	24	21	12
Cu	7	162	32	34	10	85	241
Ni	2	31	79	4	1	82	----
Tl	----	1	1	2	1	0	0
Co	1	0	0	----	0	2	0
Cr	2	35	39	11	8	21	21
Cs	----	0	1	1	0	0	0
Sb	----	10	3	10	4	1	1
Sr	----	7	8	16	4	9	8
Rb	----	3	4	9	3	3	9
Ti	7	34	48	130	5	187	7
V	2	7	4	4	2	19	6
Ga	----	----	---	5	0	1	0
Pb	51	94	30	143	16	30	2
total	424	413	282	577	219	568	731

*The levels of elements are taken from the whole sample.