

1 **Characteristics and source identification of polycyclic aromatic hydrocarbons and**
2 ***n*-alkanes in PM_{2.5} in Xiamen**

3 Ningning Zhang^{1,2*}, Junji Cao^{1,2}, Lijuan Li^{1,2,3}, Steven Sai Hang Ho^{1,2,4}, Qiyuan Wang^{1,2}, Chongshu
4 Zhu^{1,2}, Linlin Wang⁵

5 ¹ Key Lab of Aerosol Chemistry & Physics, Institute of Earth Environment, Chinese Academy of Sciences,
6 Xi'an, China

7 ² State Key Lab of Loess and Quaternary Geology (SKLLQG), Institute of Earth Environment, Chinese
8 Academy of Sciences, Xi'an, China

9 ³ University of Chinese Academy of Sciences, Beijing, China

10 ⁴ Division of Atmospheric Sciences, Desert Research Institute, Reno, NV, USA

11 ⁵ College of Architectural Engineering, Binzhou University, Binzhou, China

12 **Abstract:**

13 PM_{2.5} samples were collected to characterize organic compounds of polycyclic aromatic
14 hydrocarbons (PAHs) and *n*-alkanes in Xiamen, China in 2013. The concentrations of PM_{2.5},
15 PAHs and *n*-alkanes were 55.53 μg m⁻³ and 22.14 μg m⁻³, 15.73 ± 6.25 ng m⁻³ and 5.65 ± 3.73
16 ng m⁻³, 148.57±36.84 ng m⁻³ and 97.53±67.46 ng m⁻³ in winter and summer, respectively,
17 demonstrating that higher pollutant levels in wintertime. Benzo[e]pyrene (BeP) was the most
18 abundant PAH, accounting for 32% and 26%, respectively, of the total quantified PAHs in
19 winter and summer, followed by coronene (Cor) and phenanthrene (Phe). Five-rings PAHs
20 was the most dominant group, contributing 50.8% and 44.1% of the total quantified PAHs in
21 winter and summer, respectively. Two peaks were shown among the homologue distributions

* Corresponding Author, Fax:+86 02962336234 ; Tel: +86 02962336272 E-mail: zhangnn@ieecas.cn

22 of *n*-alkanes, and the dominant components enriched in high molecular weight fraction.
23 Meteorological parameters had stronger impacts on atmospheric PAHs levels in summer than
24 that in winter, while the most significant parameter was relative humidity (RH), followed by
25 temperature. Diagnostic ratios indicated that PAHs in Xiamen were mainly contributed by
26 petroleum combustion in the two seasons, and the concentrations of *n*-alkane were mainly
27 influenced by anthropogenic sources. The annual values of benzo[a]pyrene (BaP) equivalent
28 concentration and incremental lifetime cancer risk (ILCR) were $0.83 \pm 0.63 \text{ ng m}^{-3}$ and $7.17 \times$
29 10^{-5} and $1.11 \times 10^{-4} \text{ ng m}^{-3}$ and 3.29×10^{-5} in winter and summer, respectively. The findings
30 illustrated that the overall exposure risk to PM_{2.5}-bound PAHs did not exceed an alert level in
31 Xiamen, but the higher risks were still shown in winter.

32
33 Keywords: PM_{2.5}, PAHs and *n*-Alkanes, Source characterization, [Health effects](#), Xiamen

34

35 INTRODUCTION

36 Along with the industrial activities, urbanization, and rapid economic expansion, air
37 pollution has been become a serious environmental issue in China. Currently, fine particulate
38 matter (PM_{2.5}), referring to particle with an aerodynamic diameter less than 2.5 μm, has been
39 identified as the most important primary pollutant in China (www.zhb.gov.cn). In 2013, the
40 mean concentration of PM_{2.5} in 74 Chinese key cities was 72 μg m⁻³, twice exceed the newly
41 ambient air quality standards in China (GB 3095-2012) of 35 μg m⁻³. Besides the
42 environmental impacts such as visibility deterioration (Cao *et al.*, 2012a; Wang *et al.*, 2013),
43 PM_{2.5} also affects human health due to particle sizes, carbonaceous contents, and specific
44 chemicals that are bounded to the particles, especially of organic fractions (Duan *et al.*, 2010;
45 Cao *et al.*, 2012b; Huang *et al.*, 2014).

46 Polycyclic aromatic hydrocarbons (PAHs) are a group of organic substances that are
47 formed during the incomplete combustion processes of fossil fuel and industrial processes in
48 heavily urbanized or industrialized regions (Mostert *et al.*, 2010; Cheruiyot *et al.*, 2015; Lai
49 *et al.*, 2016; Redfern *et al.*, 2017). They can be also emitted from natural processes such as
50 biomass burning (Wang *et al.*, 2007; Saha *et al.*, 2017). PAHs are ubiquitous environmental
51 pollutants, which widely exist in the atmosphere, water, soil and sediments (Wang *et al.*, 2007;
52 Mostert *et al.*, 2010) and are toxic to all living organisms. The concentrations of PAHs were
53 varied on the size distribution of airborne particulate matters, and about 95% of total PAHs
54 present in the particulate with a diameter less than 3 μm (Baek *et al.*, 1991a, b; Venkataraman
55 *et al.*, 1994). *n*-Alkanes are also ubiquitous which is a group of non-polar and
56 photo-catalytically stable organic compounds. They are generated by biological source and

57 incomplete combustion of fossil fuel, wood and agricultural burning (Simoneit, 1977).

58 Xiamen is a coastal city located at southeastern China where is in the west coast of the
59 Taiwan Straits. Its climate is dominated by sub-tropic climate influenced by northeastern
60 monsoon from October to April of next year and southwestern monsoon from May to August
61 (Li *et al.*, 2013). The prevailing wind directions in winter are north and northeast, while south
62 and southeast winds are prevailing in summer. As similar as most cities in China, Xiamen has
63 been suffered from air pollution, especially during wintertime (Zhang *et al.*, 2012; Zhang *et*
64 *al.*, 2016). Most of the previous studies focused on the characterization, seasonal variations
65 and sources apportionment of PM (Zhao *et al.*, 2011a; Zhang *et al.*, 2012; Li *et al.*, 2013;
66 Zhang *et al.*, 2016). The characterization, seasonal variation and spatial and temporal
67 distribution of PAHs in PM_{2.5}, PM₁₀ and TSP in Xiamen have been studies during 2003~2004
68 and 2008~2009, respectively (Hong *et al.*, 2007; Wang *et al.*, 2007; Zhao *et al.*, 2011a, b).
69 Very limited studies were conducted on ambient *n*-alkanes, besides a report on their levels in
70 PM₁₀ (All undergraduate of 2003, 2007) and a measurement done inside and outside a temple
71 (Wu *et al.*, 2009). However, none of study focused on both PAHs and *n*-alkanes status. Thus,
72 the results of our work are expected to fully understand the current seasonal variations,
73 characterize their potential sources, evaluate the health risk of PAHs in Xiamen.

74

75 **MATERIAL AND METHODS**

76 *Ambient samples*

77 All the samples were collected at Huli district in Xiamen (Figure 1). Huli district was used
78 to be the industry district in Xiamen Island. Recently, in order to protect the environment,

79 most of industries have been moved out of the island. However, there still have port and
80 airport at Huli. Our sampling site is set at a building which is about 8m above ground with
81 about 300 m to main road, 2 km to port and 4 km to airport. Quartz microfiber filters (47 mm
82 diameter, Whatman) were used for PM_{2.5} sampling which were collected synchronously in
83 January and July representing winter and summer. For each month, three weeks were used for
84 sampling, and each sample was run for 24 h from 10:00 to 10:00 next day.

85 *Analysis*

86 All the samples were analyzed at the Institute of Earth Environment, Chinese Academy of
87 Sciences (Xi'an). The quartz filters were pre-combusted at 800 °C for 3 h prior to use. Before
88 sampling, they were maintained at a relative humidity (RH) of 50% and 25 °C for over 24 h
89 and then weighted using a Sartorius MC5 electronic microbalance (Sartorius, Göttingen,
90 Germany) with a sensitivity of ±1µg. After sampling, the filters were conditioned and
91 weighed again to determine the mass concentration of the filtered particles. One punch (0.526
92 cm²) of filter was analyzed using an Agilent 7890A GC/5975MSD system (City, Country,
93 USA) with in-injection port thermal-desorption-gas chromatography/mass spectrometry
94 (TD-GC-MS) method. The details of TD-GC-MS method have been described by Ho and Yu
95 (2004) and Ho *et al.* (2008 and 2011). The parameter settings and quality control on the
96 analysis were reported in Chen *et al.* (2014). A total of 17 PAHs (i.e., phenanthrene (Phe),
97 anthracene (Ant), fluoranthene (Flu), pyrene (Pyr), benzo[a]anthracene (BaA), chrysene
98 (Chr), benzo[b]fluoranthene (BbF), benzo[k]fluoranthene (BkF) benzo [a]fluoranthene (BaF),
99 benzo[e]pyrene (BeP), benzo[a]pyrene (BaP), perylene (Per), indeno[1,2,3-cd]pyrene (IcdP),
100 dibenzo[a,h]anthracene (DahA), benzo[ghi]perylene (BghiP), coronene (Cor), and

101 dibenzo[a,e] pyrene (DaeP)) and 19 *n*-alkanes (i.e., from C₁₇ to C₃₅) were quantified.

102 ***Meteorological data and gases data***

103 Meteorological parameters including temperature (T), RH, atmospheric pressure (P), wind
104 speed and direction were recorded continuously from Xiamen Meteorological Bureau (Figure
105 2). Higher temperature was seen in summer (29.5°C) than winter (14.7°C). However, due to it
106 is located in coastal area, less variations on RHs were seen between winter (57.6%) and
107 summer (66.0%). Affected by continental high pressure system, the pressure was 985.3 hPa
108 in winter, which is higher than 977.2 hPa in summer. A large difference was found for wind
109 direction. The main wind direction is northeast in winter but southwest in summer. The data
110 for other gases including SO₂, NO₂, O₃ and CO were obtained from Xiamen Environmental
111 Monitoring Central Station.

112 ***Health risk assessment***

113 The health risk of PAHs can be assessed by BaP equivalent concentration (BaP_{eq}). It is
114 calculated by equation (1):

$$115 \quad BaP_{eq} = \sum C_i \times TEF_i \quad (1)$$

116 where C_i refers to the mass concentration of specific PAHs components, TEF_i is their
117 corresponding toxic equivalent factor (TEF). According to U.S.EPA (2012), the TEF_i values
118 for Phe, Ant, Flu, Pyr, BaA, Chr, BbF, BkF, BeP, BaP, Per, IcdP, BghiP and DahA are 0.001,
119 0.01, 0.001, 0.001, 0.1, 0.01, 0.1, 0.1, 0.01, 1, 0.001, 0.1, 0.01 and 5 respectively. By defined
120 the UR_{BaP} (unit risk) as the number of people at risk of contracting cancer from inhalation a
121 BaP_{eq} of 1 ng m⁻³ within 70-year lifetime, the incremental lifetime cancer risk (ILCR) can be
122 calculated as equation (2), and the value of UR_{BaP} used in this study is 8.7 × 10⁻⁵ (WHO,

123 2000):

$$124 \quad ILCR = \sum BaP_{eq} \times UR_{eq} \quad (2)$$

125

126 **RESULTS AND DISCUSSION**

127 Obvious seasonal variations of PM concentrations have been reported in Xiamen by
128 previous works (Zhao *et al.*, 2011a; Li *et al.*, 2013). In this study, the concentrations of PM_{2.5}
129 were 55.53 $\mu\text{g m}^{-3}$ and 22.14 $\mu\text{g m}^{-3}$ in winter and summer, respectively, which were 25% and
130 12% lower than those measured in winter (74.2 $\mu\text{g m}^{-3}$) and summer (25.2 $\mu\text{g m}^{-3}$) in 10 years
131 ago (Cao *et al.*, 2012). Our results indicate that more decrement of PM_{2.5} appeared in winter
132 in the last decade. The annual (average values of winter and summer) PM_{2.5} mass
133 concentration is 38.84 $\mu\text{g m}^{-3}$ which is close to 36 $\mu\text{g m}^{-3}$ issued by Xiamen Environmental
134 Protection Bureau (<http://www.xmepb.gov.cn/>) and the new ambient quality standard of 35
135 $\mu\text{g m}^{-3}$ established by Ministry of Environmental Protection of China in 2012 (GB
136 3095-2012).

137 ***PAHs concentration***

138 The average concentration of total quantified PAHs was $10.10 \pm 7.07 \text{ ng m}^{-3}$ during the
139 whole sampling period (Table 1), which was lower than 14.78 ng m^{-3} in PM₁₀ from 2004 to
140 2005 and 21.89 ng m^{-3} in TSP from 2008 to 2009 (Hong *et al.*, 2007; Zhao *et al.*, 2011b). The
141 concentration levels in Xiamen are close to Lhasa (i.e., $10 \pm 6.6 \text{ ng m}^{-3}$ and $20 \pm 15 \text{ ng m}^{-3}$ at
142 two sites) (Liu *et al.*, 2013), higher than those reported at Hsinchu (1.0-8.4 ng m^{-3}) (Yang *et*
143 *al.*, 2017), a high mountain site (1.36 ng m^{-3}) (Liu *et al.*, 2017), and some rural areas such as
144 Lulang ($0.59 \pm 0.52 \text{ ng m}^{-3}$) (Chen *et al.*, 2014) and Mt. Halla ($0.40 \pm 0.58 \text{ ng m}^{-3}$) (Lee *et al.*,

145 2008), but much lower than the values of $104 \pm 130 \text{ ng m}^{-3}$ in Beijing (Ma *et al.*, 2011) and
146 $25.92\sim 90.80 \text{ ng m}^{-3}$ in Nanjing. Seasonal differences were seen with the values of $15.73 \pm$
147 6.25 ng m^{-3} in winter and $5.65 \pm 3.73 \text{ ng m}^{-3}$ in summer. The results are consistent to the trend
148 that the PAHs concentrations were higher in winter than summer in most Chinese cities (Zhao
149 *et al.*, 2011b, Bandowe *et al.*, 2014). The ratio of mean winter to mean summer (W/S) is 2.78,
150 which was slight lower than the W/S value (2.82) reported at Xiamen in PM_{10} from 2004 to
151 2005 (Hong *et al.*, 2007).

152 As shown in Figure 3, BeP was the most abundant PAHs, accounting for 32% and 26% of
153 the total quantified PAHs during the winter and summer, followed by Cor and Phe. This result
154 was different from that in a high mount site, where Phe, Flu and IcdP were the top three
155 abundant PAHs species (Liu *et al.*, 2017). Our targeted PAHs can be classified into 3- to
156 7-aromatic rings, and furtherly defined as low molecular weight (LMW, referred to 3-rings
157 PAHs), middle molecular weight (MMW, referred to 4-rings PAHs), and high molecular
158 weight (HMW, sum of 5-, 6- and 7-rings PAHs) (Table 1). The dominant PAHs in our study
159 were 5-rings in both winter and summer, accounting for 50.8% and 44.1% of the total PAHs
160 respectively (Figure 3). The contribution of 3-rings PAHs was only 7.8% in winter but as
161 high as 20.4% in summer, explaining by the gas/particulate partitioning effect due to the
162 temperature differences. During the entire sampling period, the ratio of HMW/LMW was
163 6.51, which is consistent with the ratio of 6.94 reported at Guangzhou, another southern
164 Chinese city (Liu *et al.*, 2015). Such ratio was 10.30 in winter but as low as 3.31 in summer.
165 It was related to the higher contributions of volatile PAHs in summer, and additional sources
166 for HMW PAHs in winter, such as long-rang transport from northern China (Hong *et al.*,

167 2007).

168 *n-Alkene concentrations*

169 Nineteen *n*-alkanes (C₁₇-C₃₅) were detected in this study with average values of
170 148.57±36.84 ng m⁻³ and 97.53±67.46 ng m⁻³ during winter and summer times, respectively
171 (Table 2). The mean concentration for the whole sampling period was 120.08±61.09 ng m⁻³,
172 which is close to 129.5 ng m⁻³, lower than 1.4 × 10⁴ ng m⁻³ and higher than 27.3 ng m⁻³
173 observed at the bus terminus, temple and campus at Xiamen in TSP, respectively (Wu *et al.*,
174 2009). In comparison, the average concentration level is equal to 163.0 ± 193.5 ng m⁻³ in
175 PM_{2.5} in Beijing (Huang *et al.*, 2006) and 94.2 ± 67.9 ng m⁻³ in TSP at Lahsa (Gong *et al.*,
176 2011), but lower than the value of 141~392 ng m⁻³ in PM₁₀ at Guangzhou (Bi *et al.*, 2002)
177 and 449~1733 ng m⁻³ in PM₁₀ at Baoji (Xie *et al.*, 2009). However, our value is higher than
178 an average of 23.5 ng m⁻³ in PM_{2.5} in Hong Kong and some rural areas, such as 1.25 ± 2.28
179 ng m⁻³ in Lulang (Chen *et al.*, 2014), 7-95 ng m⁻³ in Finland (Rissanen *et al.*, 2006) and 8.2
180 ng m⁻³ in Sweden (Wingfors *et al.*, 2011). The W/S for *n*-alkanes was 1.52, which was much
181 lower than the value of 6.99 reported in Taiyuan, where strong influence from heating was
182 found during wintertime, indicating the seasonal differences for the levels of *n*-alkanes were
183 relatively small in Xiamen.

184 The distribution of C₁₇-C₃₅ concentration as a function of carbon number had similar
185 pattern during the winter and summer with two peaks obviously divided by C₂₅ (Figure 4).
186 However, the peak values were found at C₂₂ and C₃₁ during winter while those were at C₃₁
187 and C₂₁ during summer. This distribution pattern was as similar as the results found in Tianjin,
188 a coastal city in northern China, where the distributions of *n*-alkane had two peaks at C₂₂-C₂₅

189 and C₂₉ (Wu *et al.*, 2007). In addition, our result was different with that only one peak at
190 C₂₉-C₃₃ was seen in Lulang. The LMW *n*-alkanes (carbon number <24) are mainly derived
191 from anthropogenic sources, such as incomplete combustion of fossil fuels, while the HMW
192 *n*-alkanes (carbon number ≥25) are regarded as originated from strong contributions of
193 plants waxes. The values of LMW/HMW was 1.03 and 0.56 during winter and summer,
194 respectively, indicating that there were more emissions from higher plants in summer.

195 ***Relations between meteorological parameters and gases with PAHs***

196 Sampling time and period are important factors influenced the relations between PAHs
197 concentration and meteorological parameters. For example, under high temperature and
198 humidity and strong radiation in summer, more PAHs partitioned from particulate phase to
199 vapor phase. Stronger photochemical reactions and thermal circulation, more rainfall and
200 deeper mixing layer can also impact on the PAHs concentration in summer than other seasons.
201 The correlations between PAHs and meteorological parameters have been widely reported in
202 previous studies. Chetwittayachan *et al.* (2002), Hien *et al.* (2007) and Liu *et al.* (2015) have
203 found significant correlations between PAHs and RH; however, no significant correlations
204 were seen by Li *et al.* (2016) and Fang *et al.* (2004), indicating that the relationship between
205 PAHs and meteorological parameters is complex.

206 In order to investigate the influences of meteorological parameters on the variation of
207 PAHs, Pearson's correlation analysis was applied in this study (Table 3). The relationship
208 between PAHs and meteorological parameters including wind speed, T, RH and P was more
209 significant in summer than in winter, and it may be also due to more influences from long
210 range transport in winter (Zhao *et al.*, 2011b). T has been regarded as the strongest parameter

211 that impact on most of the targeted PAHs. Pehnec *et al.* (2016) and Amarillo and Carreras
212 (2016) have reported that significantly negative correlation was found between T and PAHs.
213 In our study, the most significant meteorological parameters for PAHs in summer was RH,
214 followed by T (Table 2). RH correlated with 4- to 7-rings PAHs negatively in summer but
215 positive for 3-rings PAHs, and no significant correlation for both PAHs was seen between RH
216 and PAHs in winter. The negative correlations between RH and PAHs were also found at
217 Cordoba, Argentina (Amarillo and Carreras, 2016). Rainfall can wash out particles and lead
218 the decrease of concentrations of PM and its species, while RH increased at the same time.
219 However, positive correlations between PAHs and RH have been reported at Zagreb, Croatia
220 (Pehnec *et al.*, 2016) and Guangzhou, China (Liu *et al.*, 2015).

221 Due to the influences from traffic emissions, positive correlations between PAHs with CO
222 and NO but negative correlations between PAHs and NO₂ and O₃ were found by Valerio *et al.*
223 (2000) at a heavy traffic street in Genoa, Italy. However, no significant correlation was found
224 between PAHs and gases (SO₂, NO₂, O₃ and CO) in our study, except the 6-rings PAHs had a
225 significant correlation between SO₂ and NO₂ during summer and the 4-rings had good
226 correlation between CO during winter.

227 ***Source identification***

228 *Diagnostic ratios for PAHs*

229 Diagnostic ratios have been used to identify the possible sources of PAHs (Yunker *et al.*,
230 2002; Ravindra *et al.*, 2008). Tobiszewski and Namieśnik (2012) have entirely summarized
231 the PAH diagnostic ratios for the identification of pollution emission in different
232 environmental compartments. Because of their relative stability (Yunker *et al.*, 2002), Ant/

233 (Phe + Ant), Fla/(Fla + Pyr), IcdP/(IcdP + BghiP) and BaA/(BaA + Chr) have been often used
234 to distinguish the possible categories of PAH sources in the environment (Table 4). It has
235 been reported as the petroleum source if the ratio of Ant / (Ant + Phe) is lower than 0.1, and
236 as combustion source if the ratio is higher than 0.1 (Pies *et al.*, 2008; Han *et al.*, 2011). For
237 the ratio of Flu / (Flu + Pyr), its value is lower than 0.4, between 0.4-0.5 and higher than 0.5
238 considered as petroleum, fossil fuel combustion and grass, wood, coal combustion sources,
239 respectively (Han *et al.*, 2011, Li *et al.*, 2016). As for BaA/ (BaA + Chr), it indicates
240 petroleum source, coal combustion source and vehicular emission source when values of the
241 ratios are lower than 0.2, between 0.2-0.35 and higher than 0.35, respectively (Yunker *et al.*,
242 2002). The IcdP/(IcdP + BghiP) ratios below 0.2 is considered as petroleum, 0.2-0.5 for
243 petroleum combustion and more than 0.5 for biomass and coal combustion (Yunker *et al.*,
244 2002).

245 In this study, the mean values of Ant/ (Phe + Ant) ratios were 0.17 (ranged from 0.13 to
246 0.22) and 0.15 (ranged from 0.12 to 0.18) during winter and summer, respectively, suggesting
247 the stable combustion sources both in winter and summer (Figure 5). For BaA/(BaA + Chr),
248 its ratios ranged from 0.18 and 0.24 in winter and from 0.19 to 0.49 in summer, and most of
249 the ratios were in the range of 0.20-0.35, indicating that the main source was petroleum
250 combustion in both seasons. Meanwhile, the ratios exceeded 0.35 in summer should be
251 caused by biomass burning emissions. According to the result showed in Figure 5, it can be
252 seen that all the values of IcdP/(IcdP + BghiP) ratios were in the range of 0.20-0.50, and most
253 values of Flu / (Flu + Pyr) were higher than 0.5 during summer.

254 *Source characterization of n-alkane*

255 Carbon maximum number (C_{\max}) and carbon preference index (CPI) are two common
256 indices to identify the potential sources of *n*-alkane (Simoneit, 1984; Rogge *et al.*, 1993;
257 Chen *et al.*, 2014). Here, the C_{\max} values ranged from C_{19} - C_{22} in winter, suggesting the
258 *n*-alkane was mainly originated from anthropogenic sources. However, the C_{\max} was C_{31} in
259 summer with a dominant peak at C_{21} indicated the mixed sources of plant wax and human
260 activities. The definition and calculation of CPI for the whole range (CPI₁), petrogenic (CPI₂)
261 and biogenic (CPI₃) can be found anywhere (Simoneit, 1984; Rogge *et al.*, 1993; Wu *et al.*,
262 2007; Chen *et al.*, 2014). Because of the C_{\max} at small carbon number (i.e. $C < 25$), some
263 values of CPI₁ and CPI₂ is below 1, and the range for CPI₁ in winter and summer were
264 0.74-1.25 and 1.11-1.43 with the mean value of 0.98 and 1.27, respectively. It supported the
265 conclusion above that there were obvious biogenic sources in summer. CPI₂ ranged from
266 0.61-1.93 during the whole sampling with the average value of 0.94 and 1.59 during winter
267 and summer, respectively. The values were close to unity indicated that anthropogenic
268 sources affected the *n*-alkanes, especially during winter. For CPI₃, its mean values were 1.10
269 \pm 0.10 and 1.12 \pm 0.04, respectively, during winter and summer, suggesting the possible
270 contribution of biological sources to PM_{2.5}-bound *n*-alkanes.

271 ***Health risk assessment***

272 In our study, the concentration of BaP is 0.28 \pm 0.14 ng m⁻³ and 0.10 \pm 0.08 ng m⁻³ during
273 winter and summer, respectively, which are both below the annual mean values of 2.5 ng m⁻³
274 and 1 ng m⁻³ established with the Chinese standard (GB3095-2012) and EU standard
275 (2005/69/EC), respectively. Meanwhile, the mean total BaP_{eq} was calculated, and its values
276 were 1.27 \pm 0.60 ng m⁻³ and 0.38 \pm 0.29 ng m⁻³ during winter and summer, respectively. The

277 average annual value of BaP_{eq} was $0.77 \pm 0.63 \text{ ng m}^{-3}$, lower than the value of 0.92 ng m^{-3} in
278 PM_{2.5} in Xiamen during December in 2004 (Wang *et al.*, 2007) and 1.36 ng m^{-3} in TSP in
279 Xiamen during 2008-2009 (Zhao *et al.*, 2011b), showing a decrease trend along the years.
280 The value was also lower than those reported in other urban areas, such as $2\text{-}64 \text{ ng m}^{-3}$ in
281 Xi'an (China) (Bandowe *et al.*, 2014), 4.10 ng m^{-3} in Guangzhou (China) (Li *et al.*, 2004), 7.1
282 ng m^{-3} in Nanjing (China) (Wang *et al.*, 2006) and 1.91 ng m^{-3} in Fuji (Japan) (Takeshi *et al.*,
283 2004), but close to 0.84 ng m^{-3} in Nanjing during the period of the 2013 Asian Youth Games
284 (Li *et al.*, 2016) and 0.99 ng m^{-3} at Birmingham (Smith and Harrison, 1996).

285 It has been reported that ILCR values lower than 1×10^{-6} , between 1×10^{-6} and 1×10^{-4} and
286 higher than 1×10^{-4} can be treated as negligible, acceptable and serious, respectively (Liu *et*
287 *al.*, 2017, and reference therein). In our study, ILCR values were 1.11×10^{-4} and 3.29×10^{-5} in
288 winter and summer, respectively, and the annual value is 7.17×10^{-5} which was much lower
289 than those in Guangzhou (598) (Liu *et al.*, 2015) and Xi'an (1450) (Bandowe *et al.*, 2014).
290 Although the annual value of ILCR is between 1×10^{-6} and 1×10^{-4} indicated that potential
291 carcinogenic effect of PAHs was not serious in Xiamen, it just exceeds 1×10^{-4} in winter
292 suggesting that risk of carcinogenic exposure increased to a concern level in winter.

293

294 CONCLUSIONS

295 In this study, the PM_{2.5} samples were collected in winter and summer for characterization,
296 seasonal variation and sources of PAHs and *n*-alkanes in Xiamen, and the health risk of PAHs
297 was also assessed. It was found that the mass concentration of PM_{2.5}, PAHs and *n*-alkanes and
298 were all higher in winter than in summer, and their concentrations were lower than the values

299 detected by previous work, indicating the air quality in Xiamen is being improved from last
300 decade. BeP, Cor and Phe were the three most abundant components of the total PAHs, and 5-
301 rings PAHs were accounted for 44.1-50.8% of the total PAHs. For *n*-alkane, the dominant
302 components were rich in high molecular weight *n*-alkane, and bimodal distribution was found
303 for its species. Because of the long transport from other areas in winter and the high RH and
304 T in summer, RH and T were the two most important meteorological parameters influencing
305 the concentration of PAHs in summer rather than in winter. Anthropogenic sources impacted
306 the concentration of PAHs and *n*-alkane mostly, fuel combustion was the mainly contributor
307 for PAHs and *n*-alkane both in winter and summer in Xiamen. Totally, compared with
308 ambient air quality standards and the values reported in other Chinese cities, the values of
309 BaP, BaP_{eq} and ILCR were much lower suggested that the health effect of PM_{2.5}-bound PAHs
310 pollution was not serious in Xiamen. However, the serious risk of carcinogenic exposure
311 increased in winter should be noticed.

312 The information from our study will be useful for understanding the recent concentration
313 level and sources of PAHs and *n*-alkane, and the health risk of PAHs at Xiamen. Although the
314 concentrations of PAHs and *n*-alkane, and the health risk of PAHs was not serious recently,
315 the contribution of PAHs and *n*-alkane were mainly related to anthropogenic sources,
316 especially fuel combustion. Consequently, management of industry related to fuel combustion
317 and improvement of current air quality, especially in winter, should be considered carefully
318 by local government for future development.

319

320 **Acknowledgments:**

321 This work was supported by the Xiamen Environmental Protection Special Project (No.
322 19 (10), 2013), the National Natural Science Foundation of China (No. 41673125), the Major
323 Research and Development Plan of Shandong Province, China (2015GSF117032), and also
324 supported by a project from Ministry of Science and Technology (2013FY112700).

325

326 **References**

- 327 All undergraduate of 2003. (2007). Characteristic and Sources of *n*-alkanes Bound to PM₁₀ in the Air of
328 Xiamen. *Journal of Xiamen University (Natural Science)*. 46 (S1): 99-102.
- 329 Allen, J.Q., Dookeran, N.M., Smith, K.A., Sarofim, A., Taghizadeh, K., Lafleur, A.L. (1996). Measurement
330 of polycyclic aromatic hydrocarbons, showing a higher concentration level in wint associated with
331 size-segregated atmospheric aero-sols in Massachusetts. *Environ. Sci. Technol.* 30, 1023-1031.
- 332 Amarillo, A.C. and Carreras, H. (2016). Quantifying the influence of meteorological variables on
333 particle-bound PAHs in urban environments. *Atmospheric Pollution Research*, 7 (4):597-602.
- 334 Baek, S.O., Goldstone, M.E., Kirk, P.W.W., Lester, J.L.N., Perry, R. (1991a). Phase distribution and
335 particle size dependency of polycyclic aromatic hydrocarbons in the urban environment.
336 *Chemosphere* 22, 503-520.
- 337 Baek, S.O., Goldstone, M.E., Kirk, P.W.W., Lester, J.L.N., Perry, R. (1991b). Methodological aspects of
338 measuring PAHs in the urban environment. *Environ. Technol.* 12, 107-129
- 339 Bandowe, B.A., Meusel, H., Huang, R.J., Ho, K., Cao, J., Hoffmann, T., Wileke, W. (2014). PM_{2.5}-bound
340 oxygenated PAHs, nitro-PAHs and parent-PAHs from the atmosphere of a Chinese megacity: seasonal
341 variation, sources and cancer risk assessment. *Science of Total Environment* 473-474, 77-87.
- 342 Bi, X., Sheng, G., Peng, P., Zhang, Z., Fu, J. (2002). Extractable organic matter in PM₁₀ from LiWan
343 district of Guangzhou City, PR China. *Sci. Total. Environ.* 300:213-28
- 344 Cao, J.J., Wang, Q.Y., Chow, J.C., Watson, J.G., Tie, X.X, et al. (2012 a). Impacts of aerosol compositions
345 on visibility impairment in Xi'an, China. *Atmospheric Environment*, 59: 559-566
- 346 Cao, J.J., Xu, H.M., Xu, Q., Chen, B.H., Kan, H.D. (2012 b) Fine particulate matter constituents and
347 cardiopulmonary mortality in a heavily-polluted Chinese city. *Environmental Health Perspectives*,
348 120(3): 373-378
- 349 Cao, J. J., Shen, Z. X., Chow, J. C., Watson, J. G., Lee, S. C., et al. (2012 c). Winter and summer pm_{2.5}
350 chemical compositions in fourteen Chinese cities. *Journal of the Air & Waste Management*
351 *Association*, 62(10), 1214.
- 352 Cheruyiot, N.K., Lee, W.J., Mwangi, J.K., Wang, L.C., Lin, N.H., Lin, C.Y., Cao, J.J., Zhang, R.J.,
353 Chien-Chang, G.P. (2015). An overview: polycyclic aromatic hydrocarbon emissions from the
354 stationary and mobile sources and in the ambient air. *Aerosol & Air Quality Research*, 15(7),
355 2730-2762.
- 356 Chetwittayachan, T., Shimazaki, D., Yamamoto, K. (2002). A comparison of temporal variation of
357 particle-bound polycyclic aromatic hydrocarbons (pPAHs) concentration in different urban
358 environments: Tokyo, Japan, and Bangkok Thailand. *Atmos. Environ.* 36, 2027-2037.
- 359 Chen, Y., Cao, J.J., Zhao, J., Xu, H.M., Arimoto, R., et al. (2014). *n*-alkanes and polycyclic aromatic
360 hydrocarbons in total suspended particulates from the southeastern Tibetan Plateau: Concentrations,

361 seasonal variations, and sources. *Science of Total Environment* 470, 9-18.

362 Duan, F.K., He, K.B., Liu, X.D. (2010). Characteristics and source identification of fine particulate
363 *n*-alkane in Beijing, China. *Journal of Environmental Sciences*, 22(7): 998-1005.

364 Fang, G., Chang, C., Wu, Y., Fu, P., Yang, I., Chen, M. (2004). Characterization, identification of ambient
365 air and road dust polycyclic aromatic hydrocarbons in central Taiwan Taichung. *Sci. Total Environ.*
366 327 (1), 135-146.

367 Gong, P., Wang, X., Yao, T. (2011). Ambient distribution of particulate- and gas-phase *n*-alkanes and
368 polycyclic aromatic hydrocarbons in the Tibetan Plateau. *Environ. Earth Sci.* 64, 1703-1711.

369 Han, B., Ding, X., Bai, Z., Kong, S., Guo, G. (2011). Source analysis of particulate matter associated
370 polycyclic aromatic hydrocarbon (PAHs) in an industrial city in northeastern China. *J. Environ. Monit.*
371 13, 2597-2604.

372 Hien, T.T., Nam, P.P., Yasuhiro, S., Takayuki, K., Norimichi, T., Hiroshi, B. (2007). Comparison of
373 particle-phase polycyclic aromatic hydrocarbons and their variability causes in the ambient air in Ho
374 Chi Minh City, Vietnam and in Osaka, Japan, during 2005-2006. *Sci. Total Environ.* 382 (1), 70-81.

375 Ho, S.S., Yu, J.Z., Chow, J.C., Zielinska, B., Watson, J.G., Sit, E.H., Schauer, J.J. (2008). Evaluation of an
376 in-injection port thermal desorption-gas chromatography/mass spectrometry method for analysis of
377 non-polar organic compounds in ambient aerosol samples. *J. Chromatogr. A* 1200, 217-227.

378 Ho, S.S.H., Chow, J.C., Watson, J.G., Ng, L.P.T., Kwok, Y., Ho, K.F., Cao, J. (2011). Precautions for
379 in-injection port thermal desorption-gas chromatography/mass spectrometry (TD-GC/MS) as applied
380 to aerosol filter samples. *Atmos. Environ.* 45, 1491-1496.

381 Ho, S.S.H., Yu, J.Z. (2004). In-injection port thermal desorption and subsequent gas chromatography/mass
382 spectrometric analysis of polycyclic aromatic hydrocarbons and *n*-alkanes in atmospheric aerosol
383 samples. *J. Chromatogr. A* 1059, 121-129.

384 Hong, H.S., Yin, H.L., Wang, X.H., Ye, C.X. (2007). Seasonal variation of PM₁₀-bound PAHs in the
385 atmosphere of Xiamen, China. *Atmospheric Research*, 85, 429-441.

386 Huang, X.F., He, L.Y., Hu, M., Zhang, Y.H. (2006). Annual variation of particulate organic compounds in
387 PM_{2.5} in the urban atmosphere of Beijing. *Atmos. Environ.* 40:2449-2458.

388 Huang, R. J., Zhang, Y., Bozzetti, C., Ho, K. F., Cao, J. J. et al. (2014). High secondary aerosol
389 contribution to particulate pollution during haze events in china. *Nature*, 514(7521), 218.

390 Lai, Y.C., Tsai, C.H., Chen, Y.L., Chang-Chien, G.P. (2017). Distribution and sources of atmospheric
391 polycyclic aromatic hydrocarbons at an industrial region in Kaohsiung, Taiwan. *Aerosol & Air*
392 *Quality Research*, 17(3), 776-787.

393 Lee, J.Y., Kim, Y.P., Kaneyasu, N., Kumata, H., Kang, C.H. (2008). Particulate PAHs levels at Mt. Halla
394 site in Jeju Island, Korea: Regional background levels in northeast Asia. *Atmospheric Research*, 90:
395 91-98.

396 Li, J., Zhang, G., Qi, S.H., Liu, G.Q. (2004). Source analysis and apportionment of particulate phase
397 polycyclic aromatic hydrocarbons (PAHs) in Guangzhou City. *Acta Sci Circumst* 24(4):661-666.

398 Li, T.C., Chen, W.H., Yuan, C.S., Wu, S.P., Wang, X.H. (2013). Physicochemical characteristics and source
399 apportionment of atmospheric aerosol particles in Kinmen-Xiamen Airshed. *Aerosol & Air Quality*
400 *Research*, 13, 308-323

401 Li, X.X., Kong, S.F., Yin, Y., Li, L., Yuan, L., et al. (2016). Polycyclic aromatic hydrocarbons (PAHs) in
402 atmospheric PM_{2.5} around 2013 Asian Youth Games period in Nanjing. *Atmospheric Research*,
403 174-175, 85-96.

404 Liu, J., Li, J., Lin, T., Liu, D., Xu, Y., Chaemfa, C. (2013). Diurnal and nocturnal variations of PAHs in the

405 Lhasa atmosphere, Tibetan Plateau: implication for local sources and the impact of atmospheric
406 degradation processing. *Atmospheric Research* 124:34-43.

407 Liu, J., Wang, Y., Li, P. H., Shou, Y. P., Li, T., & Yang, M. M., et al. (2017). Polycyclic Aromatic Hydrocarbons
408 (PAHs) at high mountain site in north china: concentration, source and health risk assessment. *Aerosol &
409 Air Quality Research*, 17: 2867–2877.

410 Liu, J.J., Man, R.L., Ma, S.X., Li, J.S., Wu, Q., Peng, J.Y. (2015). Atmospheric levels and health risk of
411 polycyclic aromatic hydrocarbons (PAHs) bound to PM_{2.5} in Guangzhou, China. *Marine Pollution
412 Bulletin*, 100, 134-143.

413 Mostert, M.M.R., Ayoko, G.A., Kokot, S. (2010). Application of chemometrics to analysis of soil
414 pollutants. *Trends in Analytical Chemistry*, 29, 430-435.

415 Pehnek, G., Jakovljević, I, Šišović, A., Bešlić, I., Vadić, V. (2016). Influence of ozone and meteorological
416 parameters on levels of polycyclic aromatic hydrocarbons in the air. *Atmospheric Environment*, 131,
417 263-268.

418 Pies, C., Hoffmann, B., Petrowsky, J., Yang, Y., Ternes, T.A., Hofmann, T. (2008). Characterization and
419 source identification of polycyclic aromatic hydrocarbons (PAHs) in river bank soils. *Chemosphere*,
420 72, 1594-1601.

421 Redfern, F.M., Lin, S.L., Wang, L.C., Shih, S.I. (2017). Influences of waste cooking oil-based biodiesel
422 blends on PAH and PCDD/F emissions from diesel engines in durability testing cycle. *Aerosol & Air
423 Quality Research*, 17(5), 1224-1233.

424 Ravindra, K., Sokhi, R., Van Grieken, R. (2008). Atmospheric polycyclic aromatic hydrocarbons: source
425 attribution, emission factors and regulation. *Atmospheric Environment* 42, 2895-2921.

426 Rissanen, T., Hyötyläinen, T., Kallio, M., Kronholm, J., Kulmala, M., et al. (2006). Characterization of
427 organic compounds in aerosol particles from a coniferous forest by GC–MS. *Chemosphere*,
428 64:1185-1195

429 Rogge, W.F., Hildemann, L.M., Mazurek, M.A., Cass, G.R., Simoneit, B.R.T. (1993). Sources of fine
430 organic aerosol. 2. noncatalyst and catalyst-equipped automobiles and heavy-duty diesel trucks.
431 *Environ Sci Technol* 27:636-651.

432 Saha, M., Maharana, D., Kurumisawa, R., Takada, H., Yeo, G.B., Rodrigues, A.C., et al. (2017). Seasonal
433 trends of atmospheric PAHs in five Asian megacities and source detection using suitable
434 biomarkers. *Aerosol & Air Quality Research*, 17: 2247-2262.

435 Simoneit, B.R.T. (1977). Organic matter in eolian dusts over the Atlantic Ocean. *Marine Chemistry*, 5:
436 443-464.

437 Simoneit, B.R.T. (1984). Organic matter of the troposphere—III. Characterization and sources of
438 petroleum and pyrogenic residues in aerosols over the western United States. *Atmos Environ (1967)*,
439 18:51-67.

440 Smith, D.J.T., Harrison, R.M. (1996). Concentrations, trends and vehicle source profile of polycyclic
441 aromatic hydrocarbons in the U.K. atmosphere. *Atmos. Environ.* 30(14): 2513-2525.

442 Takeshi, O., Takashi, A., Masahiro, F., Hidetsuru, M. (2004). Spatial distributions and profiles of
443 atmospheric aromatic hydrocarbons in two industrial cities in Japan. *Environ. Sci. Technol.*
444 38(1):49-55.

445 Tobiszewski, M., Namieśnik, J. (2012). PAH diagnostic ratios for the identification of pollution emission
446 sources. *Environmental Pollution*, 162(1):110-119.

447 US EPA (2012). Regional screening levels for chemical contaminants at superfund sites. Regional
448 Screening Table User's Guide (Access date: November 2012).

449 http://www.epa.gov/reg3hwmd/risk/human/rb-concentration_table/usersguide.htm(N).

450 Valerio, F., Stella, A., Munizzi, A. (2000). Correlations between PAHs and CO, NO, NO₂, O₃ along an
451 urban street. *Polycyclic Aromatic Compounds*, 20(1-4), 235-244.

452 Venkataraman, C., Lyons, J.M., Friedlander, S.K. (1994). Size distributions of polycyclic aromatic
453 hydrocarbons and elemental carbon. 1. Sampling, measurement methods, and source characterization.
454 *Environ. Sci. Technol.* 28, 555-562.

455 Wang, Q.Y., Cao, J.J., Tao, J., Li, N., Su, X.L. et al. (2013). Long-Term Trends in Visibility and at
456 Chengdu China. *Plos One*, 8(7), DOI: 10.1371/journal.pone.0068894.

457 Wang, G., Huang, L., Zhao, X., Niu, H., Dai, Z. (2006). Aliphatic and polycyclic aromatic hydrocarbons of
458 atmospheric aerosols in five locations of Nanjing urban area, China. *Atmos. Res.* 81, 54-66.

459 Wang, X.H., Ye, C.X., Yin, H.L., Zhuang, M.Z., Wu, S.P., Mu, J.L., Hong, H.S. (2007). Contamination of
460 Polycyclic Aromatic Hydrocarbons Bound to PM₁₀/PM_{2.5} in Xiamen, China. *Aerosol & Air Quality
461 Research*, 7(2): 260-276.

462 Wang, Z., Chen, J., Yang, P., Qiao, X., Tian, F. (2007). Polycyclic aromatic hydrocarbon in Dalian soils:
463 distribution and toxicity assessment. *Journal of Environmental Monitoring* 9, 199-204.

464 WHO. Air quality guidelines for Europe. Copenhagen, Denmark: WHO Regional Office for Europe; 2000
465 [http://www.euro.who.int/_data/assets/pdf_file/0005/74732/E71922.pdf accessed March 19, 2013].

466 Wingfors, H., Häggglund, L., Magnusson, R. (2011). Characterization of the size-distribution of aerosols
467 and particle-bound content of oxygenated PAHs, PAHs, and n-alkanes in urban environments in
468 Afghanistan. *Atmos. Environ.* 45:4360-4369

469 Wu, S.P., Tao, S., Zhang, Z.H., Lan, T., Zuo, Q. (2007). Characterization of TSP-bound n-alkanes and
470 polycyclic aromatic hydrocarbons at rural and urban sites of Tianjin, China. *Environ. Pollut.*
471 147:203-210.

472 Wu, S.P., Wang, X.P., Hong, H.S., Yan, J.M. (2009). Measurement of Particulate n-alkanes and PAHs
473 Inside and Outside a Temple in Xiamen, China. *Aerosol & Air Quality Research*, 9(1): 120-138.

474 Xie, M., Wang, G., Hu, S., Han, Q., Xu, Y., Gao, Z. (2009). Aliphatic alkanes and polycyclic aromatic
475 hydrocarbons in atmospheric PM₁₀ aerosols from Baoji, China: implications for coal burning. *Atmos.
476 Res.* 93:840-848.

477 Yang, T. T., Hsu, C. Y., Chen, Y. C., Young, L. H., Huang, C. H., & Ku, C. H. (2017). Characteristics, sources,
478 and health risks of atmospheric PM_{2.5}-bound Polycyclic Aromatic Hydrocarbons in Hsinchu,
479 Taiwan. *Aerosol & Air Quality Research*, 17: 563-573.

480 Yunker, M.B., Macdonald, R.W., Vingarzan, R., Mitchell, R.H., Goyette, D., Sylvestre, S. (2002). PAHs in
481 the Fraser River basin: a critical appraisal of PAH ratios as indicators of PAH source and composition.
482 *Organic Geochemistry* 33, 489-515.

483 Zhang, F.W., Xu, L.L., Chen, J.S., Yu, Y.K., Niu, Z.C., Yin, L.Q. (2012). Chemical compositions and
484 extinction coefficients of PM_{2.5} in peri-urban of Xiamen, China, during June 2009-May 2010.
485 *Atmospheric Research*, 106, 150-158

486 Zhang, N.N., Zhuang, M.Z., Tian, J., Zhang, J.R., Wang, Q.Y. et al. (2016). Development of source profiles
487 and their application in source apportionment of PM_{2.5} in Xiamen, China. *Frontiers of Environmental
488 Science & Engineering*, 10(5), 17.

489 Zhao, J.P., Zhang, F.W., Xu, Y., Chen, J.S., Yin, L.Q., et al. (2011 a). Chemical Characteristics of
490 Particulate Matter during a Heavy Dust Episode in a Coastal City, Xiamen. *Aerosol & Air Quality
491 Research*, 11: 299-308

492 Zhao, J.P., Zhang, F.W., Xu, L.L., Chen, J.S., Xu, Y. (2011 b). Spatial and temporal distribution of

493 polycyclic aromatic hydrocarbons (PAHs) in the atmosphere of Xiamen, China. *Science of the Total*
494 *Environment*, 409, 5318-5327.

ACCEPTED MANUSCRIPT

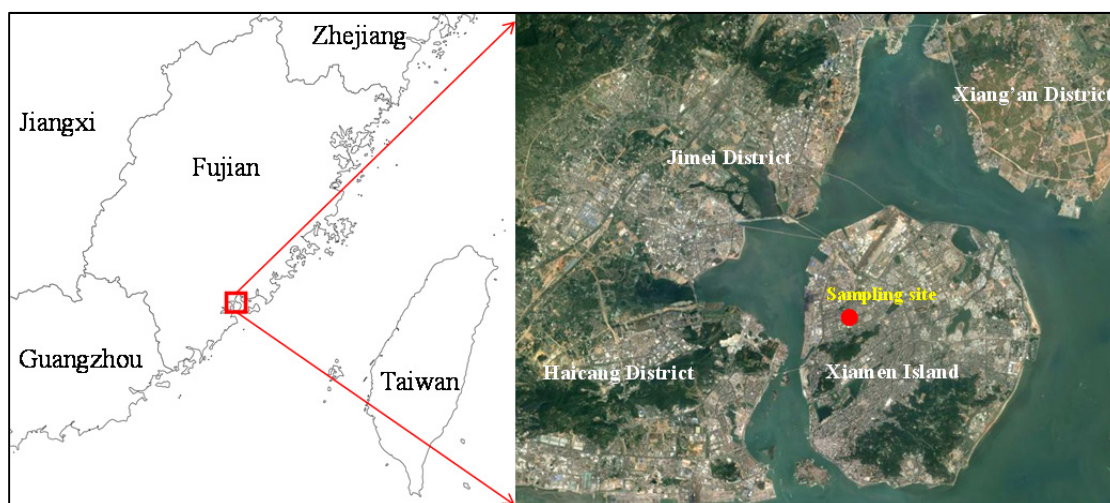


Figure 1 Location of sampling site

ACCEPTED MANUSCRIPT

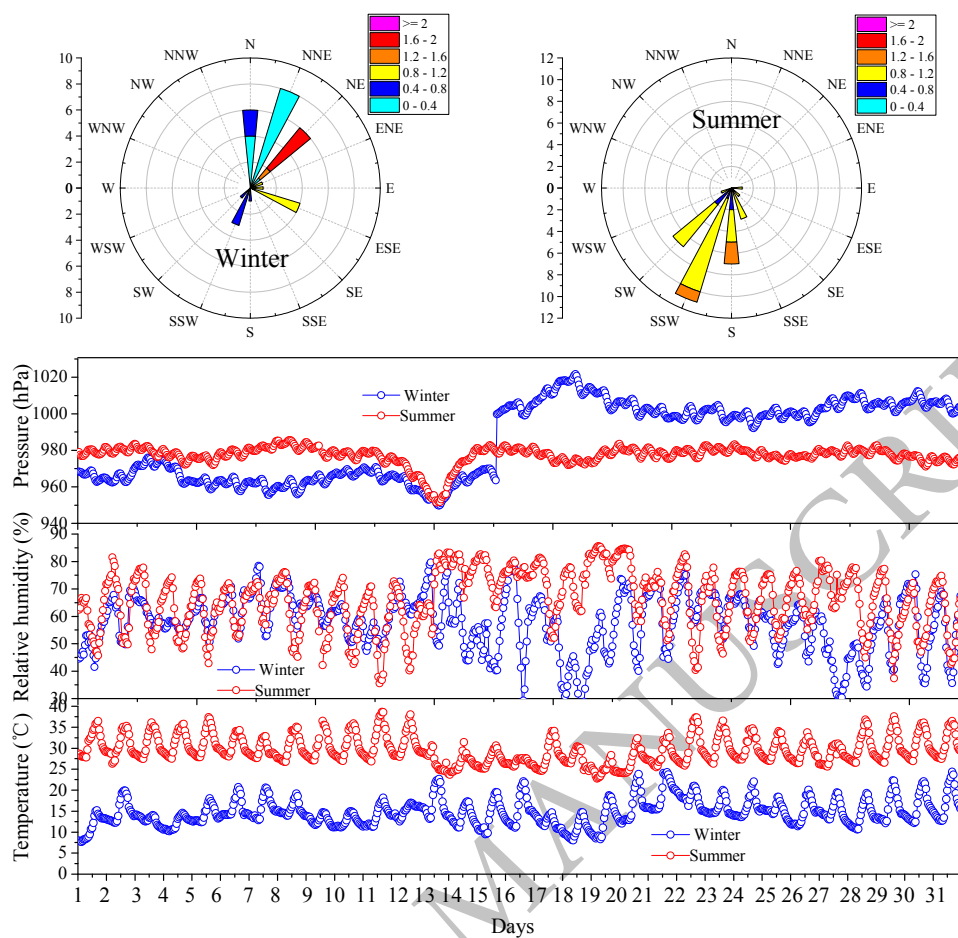


Figure 2 Variation of meteorological parameters in January (winter) and July (summer)

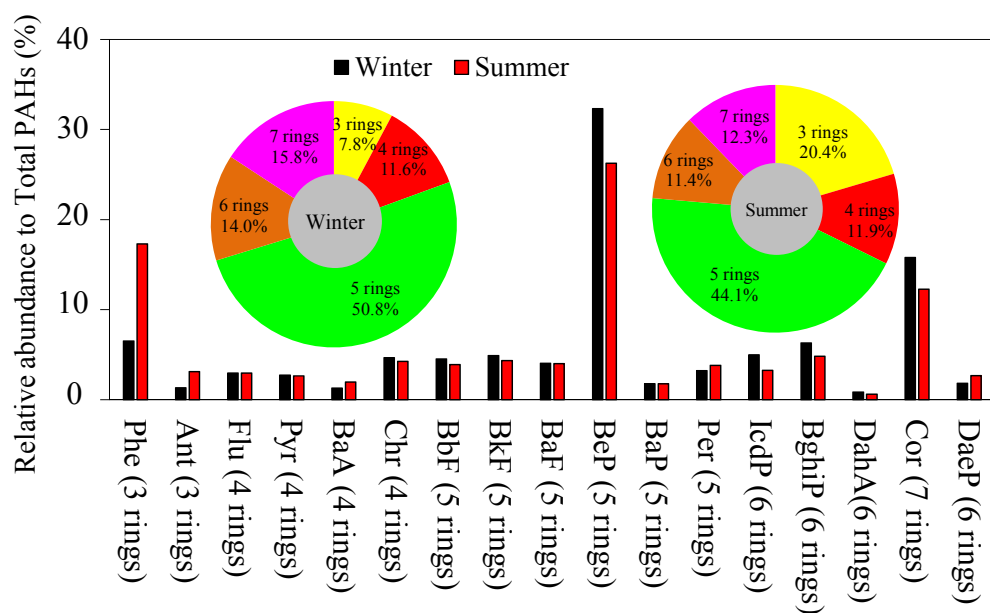


Figure 3 Relative abundance of detected PAHs components to total PAHs

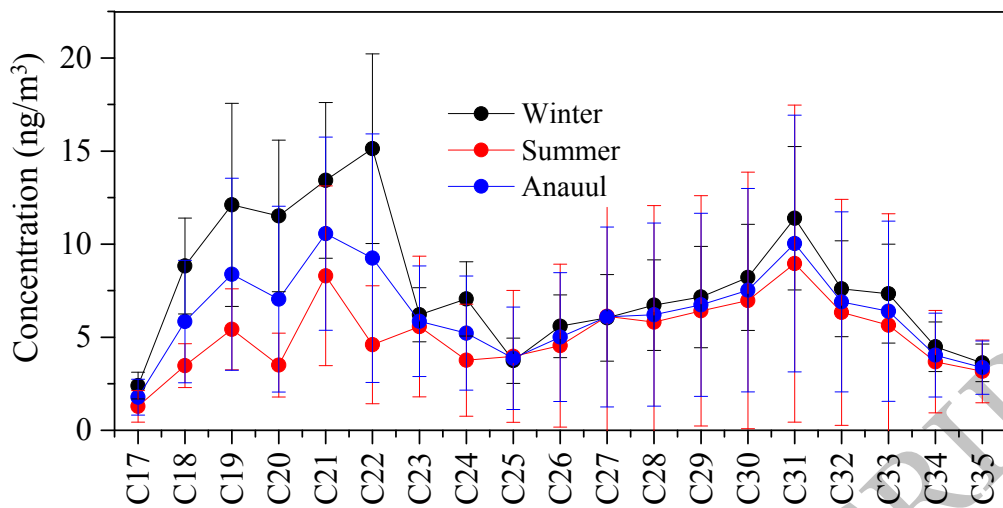


Figure 4 Variation of detected *n*-alkane concentration

ACCEPTED MANUSCRIPT

Winter Summer Annual

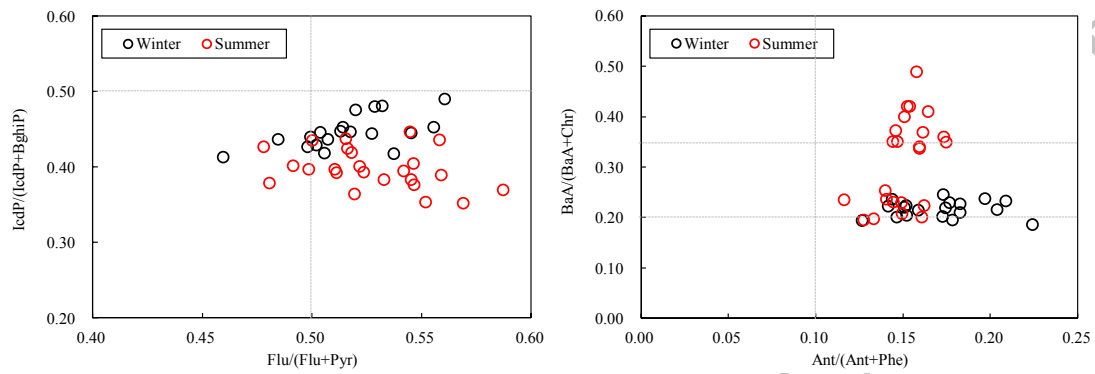


Figure 5 Diagnostic ratios for determining sources of PAHs

ACCEPTED MANUSCRIPT

	Average	SD	Average	SD	Average	SD	W/S
Phe (3 rings)	1.02	0.23	0.98	0.59	1.00	0.46	1.05
Ant (3 rings)	0.21	0.03	0.18	0.11	0.19	0.09	1.16
Flu (4 rings)	0.47	0.20	0.17	0.07	0.30	0.20	2.77
Pyr (4 rings)	0.43	0.15	0.15	0.06	0.27	0.18	2.85
BaA (4 rings)	0.20	0.09	0.11	0.05	0.15	0.08	1.82
Chr (4 rings)	0.73	0.30	0.24	0.12	0.46	0.33	3.05
BbF (5 rings)	0.71	0.28	0.22	0.20	0.44	0.34	3.24
BkF (5 rings)	0.77	0.35	0.25	0.19	0.48	0.38	3.13
BaF (5 rings)	0.64	0.29	0.23	0.16	0.41	0.30	2.80
BeP (5 rings)	5.08	2.40	1.48	1.24	3.07	2.57	3.43
BaP (5 rings)	0.28	0.14	0.10	0.08	0.18	0.14	2.79
Per (5 rings)	0.51	0.29	0.22	0.17	0.35	0.27	2.37
IcdP (6 rings)	0.78	0.37	0.18	0.15	0.45	0.40	4.25
BghiP (6 rings)	0.99	0.50	0.27	0.23	0.59	0.52	3.63
DahA(6 rings)	0.13	0.06	0.03	0.03	0.08	0.07	3.90
Cor (7 rings)	2.48	1.09	0.69	0.60	1.48	1.23	3.58
DaeP (6 rings)	0.29	0.11	0.15	0.20	0.21	0.18	1.90
PAHs	15.73	6.25	5.65	3.73	10.10	7.07	2.78
LMW-PAHs	1.23	0.24	1.15	0.70	1.19	0.55	1.07
MMW-PAHs	1.83	0.67	0.67	0.27	1.18	0.75	2.73
HMW-PAHs	12.67	5.70	3.83	3.17	7.73	6.26	3.31

Table 1 Mass concentration of PAHs components in PM_{2.5} at Xiamen

Table 2 Mass concentration of *n*-alkanes components in PM_{2.5} at Xiamen

	Winter		Summer		Annual		W/S
	Average	SD	Average	SD	Average	SD	
<i>n</i> -Heptadecane (C ₁₇)	2.40	0.71	1.29	0.85	1.78	0.96	1.86
<i>n</i> -Octadecane (C ₁₈)	8.83	2.58	3.47	1.18	5.84	3.29	2.54
<i>n</i> -Nonadecane (C ₁₉)	12.12	5.45	5.42	2.17	8.38	5.16	2.24
<i>n</i> -Icosane (C ₂₀)	11.52	4.08	3.50	1.72	7.04	5.00	3.29
<i>n</i> -Heneicosane (C ₂₁)	13.43	4.19	8.30	4.83	10.56	5.19	1.62
<i>n</i> -Docosane(C ₂₂)	15.13	5.09	4.60	3.17	9.25	6.68	3.29
<i>n</i> -Tricosane (C ₂₃)	6.21	1.45	5.58	3.78	5.85	2.97	1.11
<i>n</i> -Tetracosane (C ₂₄)	7.05	1.99	3.76	3.01	5.22	3.07	1.88
<i>n</i> -Pentacosane(C ₂₅)	3.74	1.22	3.97	3.55	3.87	2.74	0.94
<i>n</i> -Hexacosane (C ₂₆)	5.59	1.69	4.55	4.38	5.01	3.46	1.23
<i>n</i> -Heptacosane (C ₂₇)	6.04	2.33	6.13	6.20	6.09	4.84	0.99
<i>n</i> -Octacosane(C ₂₈)	6.72	2.43	5.81	6.26	6.21	4.92	1.16
<i>n</i> -Nonacosane (C ₂₉)	7.15	2.72	6.41	6.19	6.74	4.93	1.12
<i>n</i> -Triacosane (C ₃₀)	8.21	2.85	6.97	6.89	7.52	5.47	1.18
<i>n</i> -Hentriacotane (C ₃₁)	11.39	3.86	8.96	8.51	10.03	6.90	1.27
<i>n</i> -Dotriacotane(C ₃₂)	7.60	2.57	6.34	6.07	6.89	4.84	1.20
<i>n</i> -Tetriactotane (C ₃₃)	7.34	2.66	5.64	5.99	6.39	4.84	1.30
<i>n</i> -Tetratriactone (C ₃₄)	4.49	1.33	3.68	2.75	4.04	2.25	1.22
<i>n</i> -Pentatriacotane (C ₃₅)	3.62	1.01	3.16	1.69	3.36	1.43	1.14
<i>n</i> -alkane	148.57	36.84	97.53	67.46	120.08	61.09	1.52
CPI 1	0.96	0.17	1.27	0.07	1.13	0.20	0.76
CPI 2	0.86	0.19	1.51	0.15	1.22	0.36	0.57
CPI 3	0.99	0.15	1.11	0.04	1.05	0.12	0.89
LMW	76.68	20.62	35.91	18.19	53.92	27.98	2.14
HMW	71.89	23.78	61.62	57.98	66.16	45.93	1.17

Table 3 Correlation analysis between differeny rings of PAHs and meteorological parameters and Gases

	Rings	SO ₂	NO ₂	O ₃	CO	PM ₁₀	PM _{2.5}	WS	T	RH	P
Winter	3	-.070	-.238	.410	.269	.364	.196	.443	-.535*	.038	.552*
	4	.228	.040	.297	.461*	.585**	.270	.508*	-.378	-.082	.428
	5	.211	.187	-.154	.095	.225	-.225	.081	-.336	.019	.361
	6	.124	.139	-.219	-.012	.094	-.376	.002	-.365	.072	.377
	7	-.045	.015	-.245	-.159	-.095	-.511*	-.091	-.419	.140	.410
	3	-.169	-.166	.389	-.212	-.019	-.022	.526**	.187	.078	-.130
	4	.228	.326	.106	-.002	.277	.363	-.054	.335	-.441*	.338
Summer	5	.378	.352	-.199	-.019	.234	.336	-.402	.472*	-.548**	.434*
	6	.440*	.407*	-.125	.018	.331	.433*	-.364	.461*	-.564**	.440*
	7	.417*	.364	-.041	-.045	.334	.407*	-.285	.447*	-.459*	.325

** Correlation is significant at the 0.01 level (2-tailed).

* Correlation is significant at the 0.05 level (2-tailed).

Table 4 Diagnostic ratios of PAHs in previous works and our study

Diagnostic ratios	Range	Winter	Summer	Annual
Ant/(Ant + Phe)	<0.1: Petroleum; >0.1: combustion	0.17	0.15	0.16
FLA/(FLA+PYR)	<0.4: Petroleum; 0.4-0.5: fossil fuel combustion; >0.5: biomass, coal combustion	0.52	0.53	0.52
BaA /(BaA +CHR)	0.2-0.35: petroleum combustion; >0.35: biomass, coal combustion	0.22	0.31	0.28
IcdP/(IcdP+BghiP)	<0.2: Petroleum; 0.2-0.5: petroleum combustion; >0.5: biomass, coal combustion	0.45	0.40	0.42