Atmospheric Environment 216 (2019) 116929

Contents lists available at ScienceDirect

Atmospheric Environment

journal homepage: www.elsevier.com/locate/atmosenv

# Comparison of cytotoxicity induced by $PM_{2.5}$-bound polycyclic aromatic compounds from different environments in Xi'an, China

Xinyi Niu[a,b], Kin Fai Ho[a,*], Hsiao-Chi Chuang[c], Jian Sun[d], Yu Huang[b,e], Tafeng Hu[b,e], Hongmei Xu[f], Jing Duan[b,e], Ka Hei Lui[a], Junji Cao[b,e,**]

[a] The Jockey Club School of Public Health and Primary Care, The Chinese University of Hong Kong, Hong Kong, China
[b] Key Lab of Aerosol Chemistry & Physics, Institute of Earth Environment, Chinese Academy of Sciences, Xi'an, China
[c] School of Respiratory Therapy, College of Medicine, Taipei Medical University, Taipei, Taiwan
[d] School of Human Settlements and Civil Engineering, Xi'an Jiaotong University, Xi'an, China
[e] State Key Lab of Loess and Quaternary Geology (SKLLQG), Institute of Earth Environment, Chinese Academy of Sciences, Xi'an, China
[f] Department of Environmental Science and Engineering, Xi'an Jiaotong University, Xi'an, China

ARTICLE INFO

*Keywords:*
$PM_{2.5}$
PAHs
Bioreactivity
Lung function

ABSTRACT

The chemical and bioreactivity properties of fine particulate matter ($PM_{2.5}$) in indoor and outdoor environments in Xi'an were characterized, and the lung function of various participants was investigated. The concentrations of polycyclic aromatic hydrocarbons (PAHs), oxygenated polycyclic aromatic hydrocarbons, and nitrated polycyclic aromatic hydrocarbons were higher in outdoor environments than in indoor environments; in addition, urban areas had higher concentrations of these compounds than did suburban areas, with fossil fuel combustion likely being the primary source. Moreover, $PM_{2.5}$-induced inflammation was higher in urban areas than in suburban areas. Indoor environments with coal combustion emissions showed relatively higher oxidative potential and inflammation. Moderate (phenanthrene) to strong (acenaphthylene and benzo(a)pyrene) correlations were observed between selected PAHs against interleukin 6 (IL-6), 8-hydroxy-desoxyguanosine (8-OHdG), and necrosis factor-α (TNF-α). Moreover, 9-fluorenone, 9,10-anthraquinone, and 5,12-naphthacenequinone exhibited higher oxidative stress and inflammation than did their parent PAHs. Forced vital capacity (FVC) and forced expiratory volume in 1 s ($FEV_1$) were negatively correlated with 8-OHdG, and $FEV_1$/FVC was negatively correlated with TNF-α and IL-6. These findings—which integrates $PM_{2.5}$ with lung function and bioreactivity analyses—suggest that coal burning, especially indoors, could elevate the cytotoxicity of $PM_{2.5}$ to the occupants and that chronic exposure may lead to a decline in lung function.

## 1. Introduction

Polycyclic aromatic hydrocarbons (PAHs) and their polar derivatives, such as oxygenated polycyclic aromatic hydrocarbons (OPAHs) and nitrated polycyclic aromatic hydrocarbons (NPAHs)—collectively called polycyclic aromatic compounds (PACs)—are ubiquitous in indoor and outdoor environments and possess mutagenic and carcinogenic properties (Wang et al., 2011; Ding et al., 2012). PAH emissions can be of natural or anthropogenic origin. Common anthropogenic outdoor sources of PAHs are fossil fuel combustion and diesel vehicles (Okuda et al., 2010); indoor sources include various household activities such as cooking, heating, smoking, burning incense, and outdoor air pollution (Naumova et al., 2002). Other than as primary emissions, OPAHs and NPAHs can also be formed through secondary reactions (Jariyasopit et al., 2014; Ringuet et al., 2012), but the contributions from primary and secondary sources could vary by sampling location and season (Kojima et al., 2010; Li et al., 2015). For example, OPAHs and NPAHs can be generated through the photolysis of parental PAHs as well as through homogeneous and heterogeneous reactions between parent PAHs and free radicals (e.g., •OH; Wang et al., 2016; Zimmermann et al., 2013). Epidemiological studies have shown that exposure to high levels of PAHs can cause genetic damage, immunodeficiency, and cardiovascular disease (Neophytou et al., 2016; Salo et al., 2004; Zhao et al., 2008). Furthermore, several studies have suggested that OPAHs and NPAHs are more carcinogenic than are their parental PAHs (Albinet et al., 2007; Durant et al., 1996).

Oxidative stress is a crucial mechanism through which exposure to particulate matter may induce adverse health effects as overproduction

* Corresponding author.
** Corresponding author. Key Lab of Aerosol Chemistry & Physics, Institute of Earth Environment, Chinese Academy of Sciences, Xi'an, China.
*E-mail addresses:* kfho@cuhk.edu.hk (K.F. Ho), cao@loess.llqg.ac.cn (J. Cao).

https://doi.org/10.1016/j.atmosenv.2019.116929
Received 7 March 2019; Received in revised form 20 August 2019; Accepted 25 August 2019
Available online 27 August 2019

of oxidants (e.g., reactive oxygen species (ROS) and free radicals) counteracts antioxidative defenses (Charrier et al., 2014). ROS comprise chemically reactive oxygen radicals or oxygen-derived species, such as hydroxyl radicals (•OH) and hydrogen peroxide (HOOH). PAHs and their derivatives have been correlated with the formation of ROS (Knecht et al., 2013). High levels of ROS in the cells could change the redox status and induce inflammation and apoptosis (Dagher et al., 2007; Risom et al., 2005). Studies have demonstrated that PAHs in fine particulate matter ($PM_{2.5}$) can damage DNA expression in winter (Perrone et al., 2010; Shi et al., 2006). OPAHs and NPAHs contain functional groups (ketones and quinones) that can cause direct oxidative damage in biological molecules (DNA and proteins), meaning that they have more potential to cause adverse human health effects than do PAHs (Chuang et al., 2012; Lin et al., 2015).

Xi'an is the capital of Shaanxi Province, a region in China experiencing robust economic growth. Consequently, the city has been experiencing severe air pollution problems in recent years (Ren et al., 2017; Wang et al., 2017). Coal, biomass burning, and vehicular emissions are the main sources of PAHs and their derivatives in winter (Hong et al., 2017). People spend most of their time in indoor environments (Chen et al., 2017). However, data on the characteristics of PAHs, OPAHs, and NPAHs in microenvironments are lacking, and information about their potential health effects remain scarce.

The objectives of this study are to (1) characterize PAHs, OPAHs, and NPAHs in indoor and outdoor environments in Xi'an, (2) identify the relationships between PAHs and their derivatives at the sampling locations, and (3) deduce the potential health implications of PAH, OPAH, and NPAH exposure.

## 2. Methodology

### 2.1. Sampling

Sampling was conducted in Xi'an, China, a city located in the center of the Guanzhong Plain (Fig. 1). The local geography results in cold and dry winters with poor diffusion conditions. The sampling locations were as follows: a central downtown area (Hui Street, HS), a residential area (Qujiang, QJ), and a suburban area (Xiangyang, XY) (Fig. 1). At these locations, 4 (HS), 5 (QJ), and 5 (XY) households participated in indoor sampling, and one household from each of the three communities participated in outdoor sampling. The details of each sampling sites were shown in Table S1. Daily samples (24-h) were collected from each household during the sampling period—January 26 to February 1 at QJ, February 27 to March 4 at XY, and March 9 to March 15 at HS (all in 2016). All windows in the indoor sampling environments were fastened because of the low temperature. The air exchange rate (ACH) was monitored by $CO_2$ analyzer (LI-820,LI-COR,USA), and ACH at HS, QJ, and XY were 0.04–0.22, 0.07–0.09, and 0.04–0.07 air changes per hour, respectively. The $NO_2$ concentration were monitored continuously by online gaseous pollutants instruments (Wei et al., 2018) with electrochemical sensors (Alphasense B4 series).

Filter-based $PM_{2.5}$ samples were collected at a flowrate of 5 L $min^{-1}$ by using two paralleled mini-volume samplers (Airmetrics, Eugene, OR, USA) equipped with 47-mm quartz fiber filters (QM-A, Whatman Inc., Clifton, NJ, USA) and 47-mm Teflon filters (Pall Life Sciences, Ann Arbor, MI, USA). A blank sample was collected at each household without switching on the mini-volume samplers. The filters were weighed before and after sampling by using an electronic microbalance (MC5, Sartorius, Göttingen, Germany) with 1 μg sensitivity after 24 h of equilibration. All filters were stored in a freezer at −20 °C until analysis.

### 2.2. PAHs, OPAHs, and NPAHs

A quarter of each sample quartz filter was extracted using a dichloromethane and methanol solution (2:1, v/v) in triplicate. The filters were spiked with deuterated PAHs (naphthalene-D8, acenaphthene-D8, phenanthrene-D10, pyreneD10, chrysene-D12, perylene-D12, and

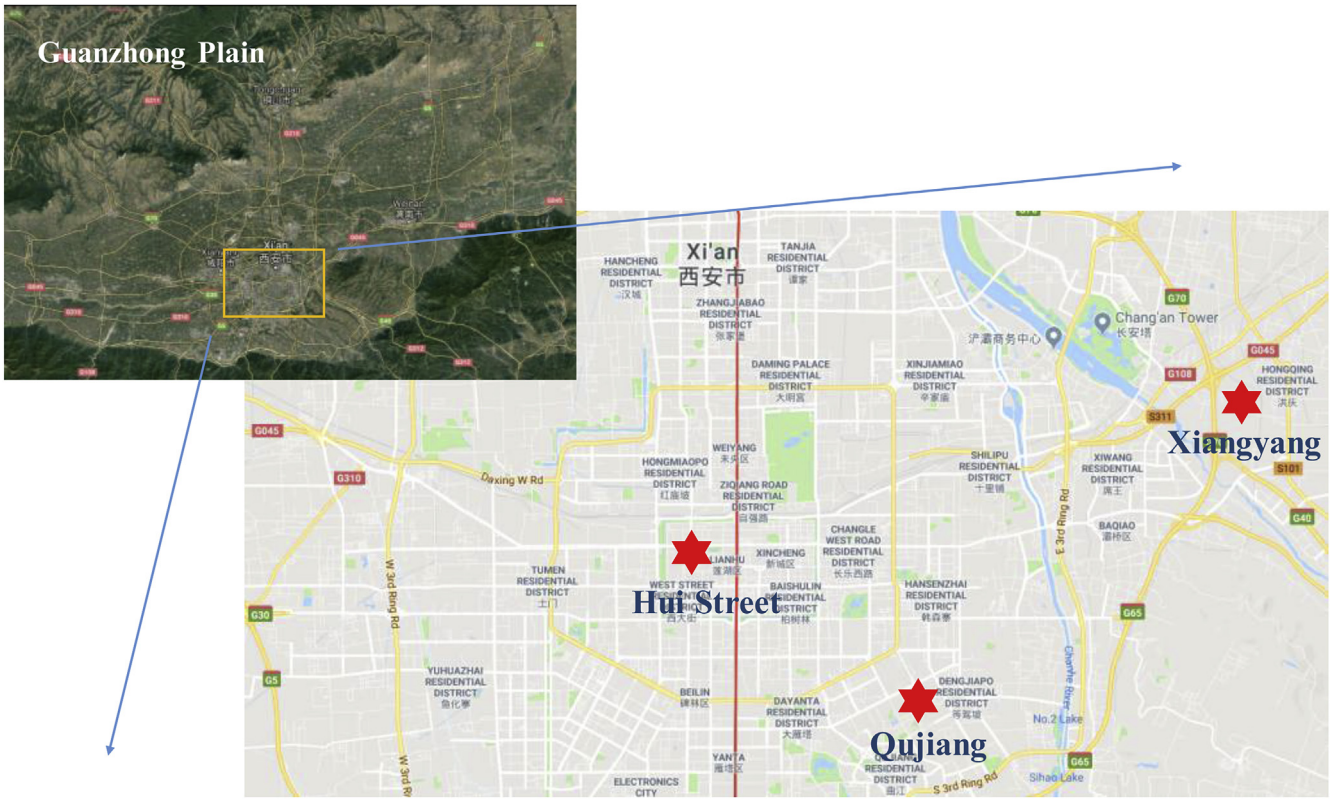


**Fig. 1.** Locations of Xi'an in Guanzhong Plain and three communities in Xi'an city.

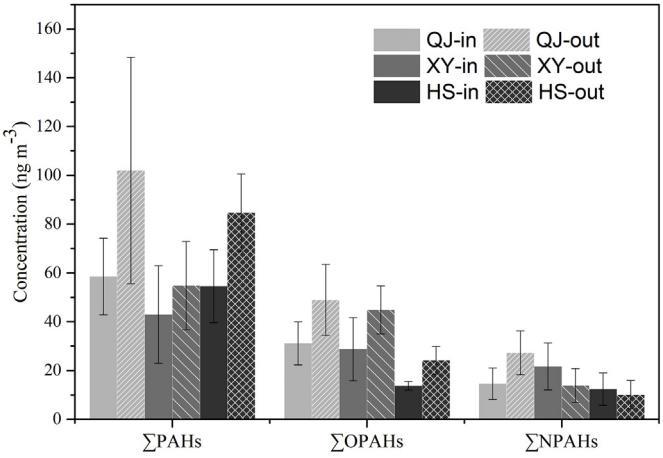


**Fig. 2.** Concentrations of total PAHs, OPAHs and NPAHs at different sampling locations.

benzo[ghi]perylene-D12), deuterated OPAHs (benzophenone-D10 and anthraquinone-D8), and deuterated NPAHs (1-Nitronaphthalene-D7, 5-Nitrofluorene-D9, 3-Nitrofluoranthene-D9, and 6-Nitrochrysene-D11) as an internal standard (IS) before extraction. To remove any interfering compounds, the extract was transferred to a column filled with anhydrous sodium sulfate. The eluent was concentrated using a rotary evaporator to approximately 0.5 mL, following which 25 μL of fluoranthene-D10 was spiked as an IS before transfer to a 2 mL vial for analysis. The PAHs and their derivatives were analyzed through a gas chromatography–mass spectrometry device (GC/MS; Agilent 7890 N/Agilent 5797C) equipped with a HP-5MS column (5% phenyl–95% methylpolysiloxane, 30 m × 0.25 mm i. d. × 0.25 μm film thickness, Agilent Technologies). PAH and OPAH concentrations were measured in the electron ionization mode of GC/MS, whereas NPAH concentrations were measured using the negative chemical ionization mode with $CH_4$ as the ionizing gas. The analysis covered 26 PAHs, 12 OPAHs, and 9 NPAHs. PAHs were classified as having a low molecular weight (LMW) PAHs (2–3 rings) and high molecular weight (HMW) PAHs (4–6 rings). The quality assurance and quality control were introduced in previous study (Bandowe et al., 2014).

### 2.3. Cell culture

The $PM_{2.5}$ samples captured on the Teflon filters were used in *vitro* experiments. The filters were immersed in 5 mL of high-purity methanol, ultrasonicated in a water bath for 30 min, and purged with nitrogen (> 99.995% purity). The filters were weighted before and after extraction, and the extraction efficiencies ranged from 78.1% to 92.6%. The $PM_{2.5}$ samples were resuspended in phosphate-buffered saline in various concentrations (0, 100, and 200 μg $mL^{-1}$) depending on the extraction fraction of each sample.

Human alveolar epithelial cells (A549) obtained from the American Type Culture Collection (ATCC, Maryland, VA, USA) were cultured in F-12 cell culture medium (Thermo Fisher Scientific Inc., MA, USA) containing 10% fetal bovine serum and 1% antibiotics (50 U·$mL^{-1}$ penicillin and 100 U·$mL^{-1}$ streptomycin). The A549 cells were seeded on 24-well transwells ($1 \times 10^5$ cells·$mL^{-1}$; ThermoFisher, Waltham, MA, USA) and cultured with 5% $CO_2$ at 37 °C. When the cells grow adhering to the wall to a certain amount, 450 μl culture medium and 50 μl $PM_{2.5}$ extracts were added to each well. The cells were exposed to the $PM_{2.5}$ samples for 24 h. The supernatants were collected for further analysis. The concentrations set in the *in vitro* exposure experiments are as described in Lai et al. (2016). In all experiments, cell viability was > 80% after sample exposure.

### 2.4. Oxidative stress and inflammation

Oxidative stress marker 8-hydroxy-desoxyguanosine (8-OHdG) and inflammation makers, such as tumor necrosis factor-α (TNF-α) and interlukin-6 (IL-6), were detected through enzyme-linked immunosorbent assay kits (R&D systems, Inc., MN, USA) as per manufacturer guidelines.

### 2.5. Participants and lung function test

Lung function tests were carried out for 47 participants (10–80 years of age) from 14 households (once for each participants during sampling periods) by using a portable spirometer (Spirobank G; MIR, Italy) according to the manufacturer's instructions. The personal information of the participants was collected through a questionnaire survey. The following lung function parameters were referenced to predicted values (%): (1) forced vital capacity (FVC), (2) forced expiratory volume in 1 s ($FEV_1$), (3) the $FEV_1$/FVC ratio, and (4) forced expiratory flow 25–75 ($FEF_{25-75}$). This study was approved by the Ethical Committee of Xi'an Jiaotong University, and all experiments were performed in accordance with the approved study protocol.

### 2.6. Statistical analysis

Pearson's correlation coefficient analysis was used to test for correlations between (1) oxidative stress and inflammation, (2) all analyzed chemical compounds and (3) lung function results (Ho et al., 2016a; Zeng et al., 2016; Chuang et al., 2018). All data were analyzed using SPSS (version 21.0, IBM, New York, NY) or GraphPad Prism (Version 5 for Windows). The significance level was set at $p < 0.05$.

## 3. Results and discussion

### 3.1. Concentration of PAHs, OPAHs and NPAHs

The total PAHs (ΣPAHs), OPAHs (ΣOPAHs) and NPSHs (ΣNPAHs) in indoor and outdoor environments were shown in Fig. 2, and Table S2 lists the concentrations of individual PAHs, OPAHs, and NPAHs. The outdoorΣPAHs concentration in Xi'an was 80.5 ± 26.8 ng $m^{-3}$, approximately 1.6 times higher than the average indoor concentration (52.0 ± 16.9 ng $m^{-3}$). Outdoor PAHs concentration exhibited the following trend: QJ > HS > XY. The outdoor ΣOPAHs and ΣNPAHs concentrations were 39.4 and 17.1 ng $m^{-3}$, respectively, and the corresponding indoor concentrations were 24.6 and 16.2 ng $m^{-3}$. Outdoor ΣOPAHs and ΣNPAHs concentrations exhibited the following trend: QJ > XY > HS. The higher concentrations at QJ could be possibly due to fossil fuel combustion from central heating and individual heating activities at the residential area. XY had the highest indoor ΣNPAHs concentration and the second highest ΣOPAHs concentration, suggesting potential secondary PAH formations due to chemical reactions in households without adequate ventilation. The aforementioned concentrations are comparable with those reported in a previous study (Lin et al., 2015).

Fig. S1 (Supplementary Materials) shows the contributions of individual components to total PACs. BbF was the most abundant PAHs in the outdoor environments at QJ and HS, accounting for 8.9%–12.4% of the ΣPAHs; IcdP and BeP were the next most abundant, accounting for 8.5%–11.6% and 8.3%–10.5%, respectively. Indoor PAHs concentrations at HS and QJ were comparable whereas the outdoor PAHs at HS were lower than QJ, indicating the major contributions of indoor sources at HS. The incomplete combustion of coal for heating and cooking purpose could emit more PAHs and possibly more harmful to human. BghiP (8.6%), BkF (7.2%), and CHR (7.0%), which are markers of combustion sources (Albinet et al., 2007), were also present in high concentrations. ΣOPAHs concentration was dominated by 1,8-NAP (28.1%–43.1%), followed by BEZ (11.6%–23.8%), BPYRone

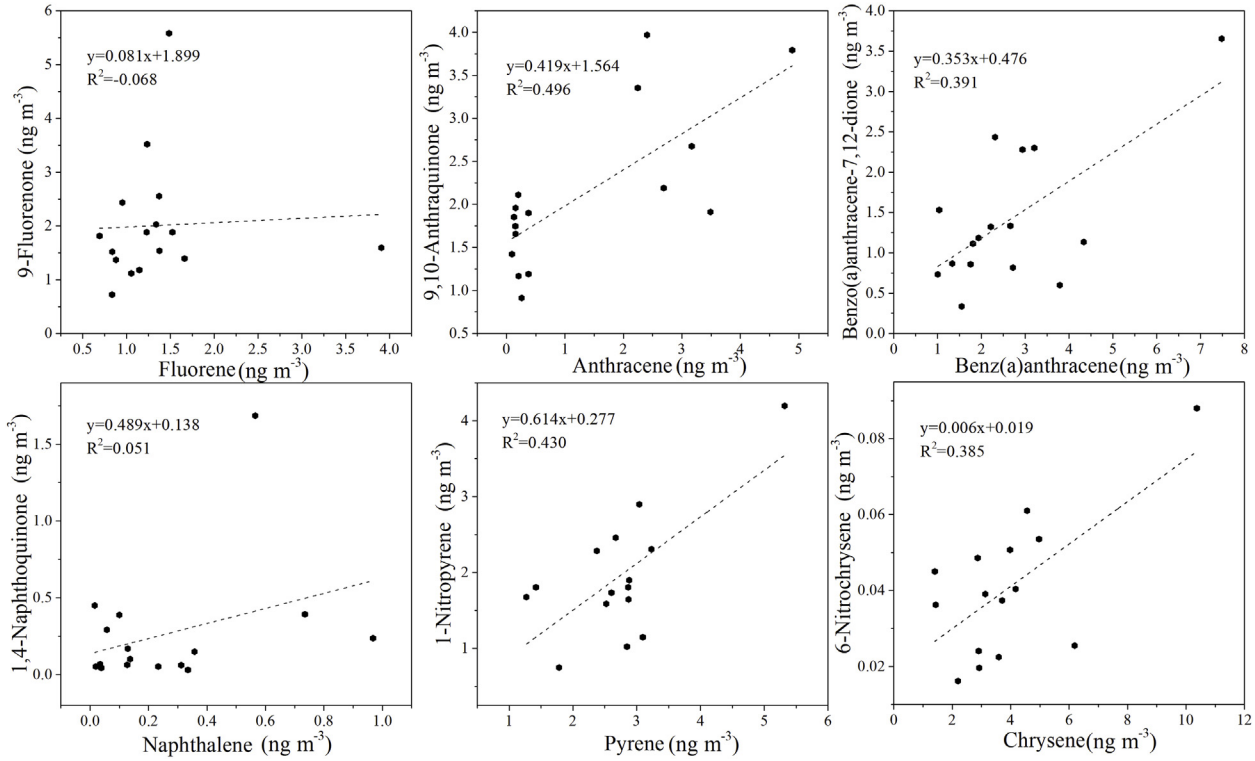


**Fig. 3.** Relationships between concentrations of parent PAHs and their related OPAHs/NPAHs. Coefficients and standard errors were included in the regression equations.

(8.6%–15.4%), and 1,2-ACNQ (4.3%–13.1%). The contributions of indoor and outdoor 1,8-NAP at HS were higher than those at the other locations, and BEZ and BPYRone accounted for larger contributions at QJ. ΣNPAHs concentration was dominated by 2-NFL (25.5%–42.1%), 2-NBP (16.6%–20.5%), and 1-NPYR (11.3%–15.4%). The contributions of individual NPAHs exhibited minimal variation between the sampling locations. The diagnostic ratios of PAHs from three sampling locations (Fig. S2) indicated the mixed influences from wood, coal and petroleum combustion sources. Since the three communities were all located in the urban area of Xi'an city, coal combustion from heating activities in winter was still the dominant sources for the indoor and outdoor environments.

Fig. 3 presents the correlations between selected individual parent PAHs and OPAHs/NPAHs from indoor and outdoor environments. Significant correlations were identified between the concentrations of ANT and 9,10-ANQ ($R^2 = 0.496$) and between PYR and 1-NPYR ($R^2 = 0.430$). Moderate correlations were found between BaA and BANTdione ($R^2 = 0.391$) and between CHR and 6-NC ($R^2 = 0.385$). We inferred that these OPAH and NPAH components (9,10-ANQ, BANTdione, 1-NPYR, and 6-NC) were partially generated from their parent PAHs or emitted directly from same combustion sources with parent PAHs. The typical primary emission sources of PYR, BaA, and CHR are coal and biomass combustion (Chang et al., 2006; Vione et al., 2004). The corresponding OPAHs and NPAHs—which were present in relatively higher concentrations outdoors (except for NPAHs at XY and HS with additional indoor sources) —were predominantly formed through secondary photochemical reactions. The conversion rate of each compound is dependent on atmospheric conditions such as moisture, temperature, and precursor pollutants (Lui et al., 2017). With the increasing number of vehicles in Xi'an city, especially at HS, the primary emissions of vehicles showed higher contributions to the NPAHs (Huang et al., 2014). In addition, the elevated nitric oxide ($NO_x$) concentrations in ambient air is an essential precursor gaseous pollutant for the secondary formation of NPAHs (Li et al., 2015).

### 3.2. Molecular ratio and indoor/outdoor ratio

The ratio of HMW PAHs to LMW PAHs (HMW/LMW) are usually used as indicators to define the sources of PAHs (Xu et al., 2016), and the average ratio was 9.0 ± 2.4 at the three locations. At QJ and XY, the outdoor HMW/LMW ratios (10.0 ± 2.6 and 7.9 ± 2.4) were 11%–14% higher than the indoor ratios (8.6 ± 3.4 and 7.0 ± 1.1). The indoor HMW/LMW ratio at HS (14.1 ± 3.6) was higher than other indoor environments. These results are consistent with other studies in Xi'an and in other cities, with the reported indoor and outdoor HMW/LMW ratios always being > 1 (Bandowe et al., 2014; Han et al., 2015; Xu et al., 2016). LMW PAHs typically exist both in gaseous and particulate phases, whereas the HMW PAHs exist mainly in the particulate phase, especially as fine particles emitted from biomass, coal, and petroleum combustion (Bi et al., 2003). The high HMW/LMW ratios indicate the heavy fossil fuel combustion in winter. Particulate-phase HMW PAHs could more severely affect human health because of the physical and chemical characteristics of the components (Kelly and Fussell, 2012). The high HMW/LMW ratios in the indoor environment at HS could be due to the use of a honeycomb briquette stove during winter for heating, together with the low air exchange rate.

The indoor/outdoor (I/O) concentration ratios of LMW and HMW PAHs, ΣPAHs, ΣOPAHs, and ΣNPAHs are presented in Fig. 4 and Table S3 (Supplementary Materials). The mean I/O ratios of the total PAHs were < 1 and showed minimal variation. The outdoor HMW and LMW concentrations are higher than that indoors, and the I/O ratio of HMW PAHs (0.72 ± 0.20) at HS was approximately 1.9 times higher than that of LMW PAHs (0.39 ± 0.02). The average I/O ratio of OPAHs was 0.61 (range = 0.57–0.64). A high concentration of precursors (e.g., parent PAHs) could facilitate the formation of OPAHs through photochemical reactions (Li et al., 2014). The I/O ratios at XY and HS were > 1 for NPAHs given the low air exchange rate, while the relative high indoor concentrations of $NO_2$ (shown in Supplementary Fig. S3) and HONO may also contribute to the reactions.

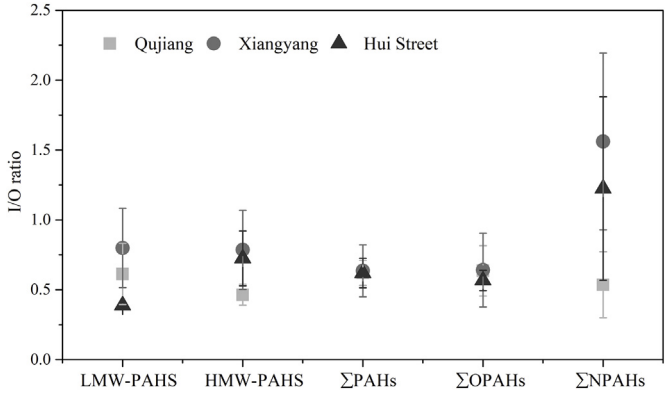


**Fig. 4.** Indoor/outdoor ratios of PAHs, OPAHs and NPAHs.

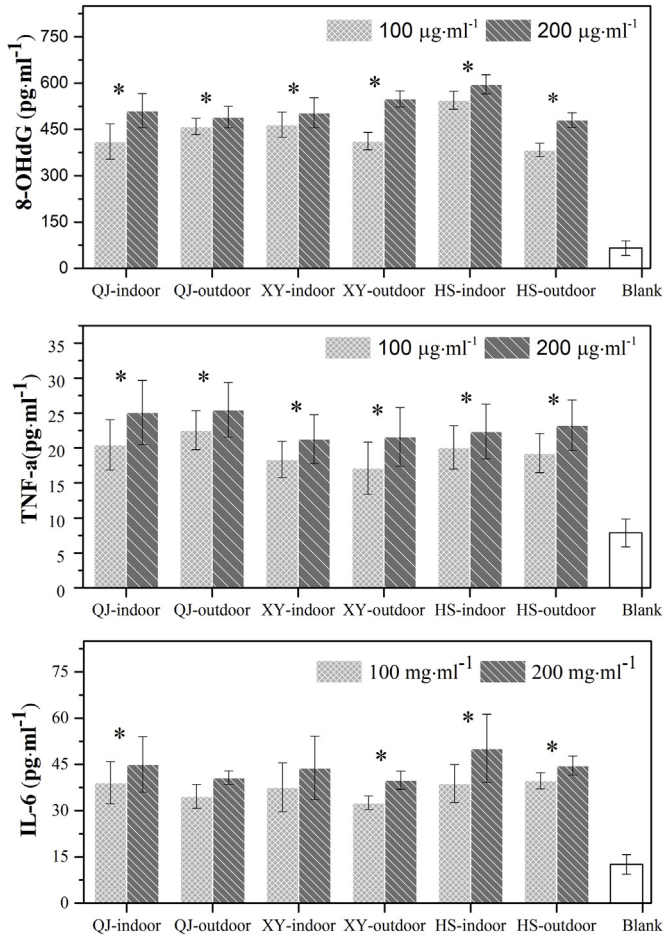


**Fig. 5.** The activity levels of 8-Hydroxy-desoxyguanosine (8-OHdG), interlukin-6 (IL-6) and tumor necrosis factor-α (TNF-α) after A549 cells exposure to $PM_{2.5}$ collected in different environments. *p < 0.05 comparing with control group, QJ: Qujiang, XY: Xiangyang, HS: Hui Street.

### 3.3. Oxidative stress and inflammation reactions in vitro

Fig. 5 presents the oxidative stress and inflammation levels. The 8-OHdG, TNF-α, and IL-6 concentrations in most samples differed significantly compared with the blank samples ($p < 0.05$). In indoor and outdoor environments, the formation of 8-OHdG, TNF-α, and IL-6 was dose-dependent. The concentrations of 8-OHdG in the indoor samples at HS were higher than those at QJ and XY. IL-6 (under a dosage of 200 μg $mL^{-1}$) was highest in the indoor and outdoor environments at HS. TNF-α (under a dosage of 100 and 200 μg $mL^{-1}$) was highest at QJ, followed by at HS and XY, in both indoor and outdoor environments.

8-OHdG is one of the major products in oxidative DNA damage and is an important mutagenic adduct to DNA (Vattanasit et al., 2014). In addition, $PM_{2.5}$ has been positively correlated with 8-OHdG (Sauvain et al., 2011). The formation of 8-OHdG in high concentrations in the indoor environment at HS indicates its high oxidative potential and DNA damage capability. In the relatively closed indoor environment at HS, the residents could easily have inhaled the $PM_{2.5}$ emitted from the honeycomb briquette stove, further inducing 8-OHdG formation (Storr et al., 2013). These toxic organic compounds could easily adhere to the core particles or enter the human lung and induce ROS (Dagher et al., 2005; Nel et al., 2001). The oxidative stress would further change the chromosomes by breaking the DNA strand and by hindering the enzymatic repair mechanisms; these effects would in turn induce genetic amplification and alterations in gene expression (Ovrevik et al., 2010; Totlandsdal et al., 2010). Although the pollution level in the outdoor environment at XY was relatively low, DNA damage caused by the XY outdoor particles was more severe than these at QJ and HS; this may be due to the differences in the contributions of individual PACs to $PM_{2.5}$ (0.12% for XY, 0.11% and 0.10% for QJ and HS), as PM-induced toxicity is affected by size, locale, and composition of the PM samples (Steenhof et al., 2011; Mirowsky et al., 2013). Studies have suggested that PM could induce systemic oxidative stress, with damage to DNA and other biomolecules even at low exposure levels (Sorensen et al., 2003).

In A549 cells, IL-6 and TNF-α levels increased in both indoor and outdoor environments with increasing $PM_{2.5}$ exposure. The outdoor environment at QJ is more influenced by coal combustion in winter, therefore its indoor environment has relatively higher concentrations of PACs by outdoor air penetration. $PM_{2.5}$ emissions from coal combustion (especially PAHs and their derivatives) are potentially associated with high TNF-α release in A549 cells (Chuang et al., 2019). Activation of inflammation is regarded as the first response of the immune system against oxidative damage–induced infections and injuries to the human body (Suhaimi and Jalaludin, 2015). A previous study suggested that the annual lung cancer rate for Chinese population caused by the inhalation of PAHs (with an average BaP concentration of 2.43 ng $m^{-3}$) was $0.65 \times 10^{-5}$, and populations in cities are at a higher risk than are those in rural areas (Zhang et al., 2009). HS exhibited the highest IL-6 levels in both indoor and outdoor environments. The high levels for indoor HS in inflammation markers are comparable with the 8-OHdG trends, highlighting the potential impact of residential heating on human health. The outdoor environment at HS is affected by heavy traffic given its central location in the city. The photochemical reactions of parent PAHs with $NO_x$ could further increase the inflammatory responses of $PM_{2.5}$ to human alveolar epithelial cells. These findings are consistent with those of Dieme et al. (2012): PM from high levels of anthropogenic emissions could have a strong association with lung disease and may induce proinflammatory cytokine production.

### 3.4. Characterization of participants and their lung function

Table S4 presents the results of the lung function tests. Spirometry was applied in the lung function measurements to differentiate the respiratory symptoms reported by the participants (Mortimer et al., 2003). Before the test, data on the personal and lifestyle characteristics of each participant were collected. The lung function parameters associated with the predicted values (%) are common indicators of asthma and chronic obstructive pulmonary disease (Shen et al., 2018). The results revealed normal pulmonary function in the participants. Individual lung function was found to depend on personal and lifestyle characteristics. Participants older than 60 years had lower predicted FVC and $FEV_1$ values than did the other adult and child participants ($p < 0.05$), indicating that lung function declines with age.

**Table 1**
Pearson's correlation coefficients of oxidative and inflammatory activities in various PAC compounds.

| | 8-OHdG | TNF-α | IL-6 |
|---|---|---|---|
| 1-MNAP | 0.557* | 0.087 | 0.065 |
| ACY | −0.286 | 0.309 | 0.624* |
| FLO | −0.181 | −0.132 | 0.597* |
| PHE | 0.603* | 0.626* | −0.251 |
| FLT | −0.131 | 0.575* | −0.375 |
| BaA | 0.766* | 0.321 | −0.365 |
| BaP | 0.760* | 0.339 | −0.315 |
| 9-FLU | 0.056 | −0.244 | 0.485* |
| 9,10-ANQ | 0.047 | −0.208 | 0.489* |
| 5,12-NACQ | −0.406 | 0.547* | 0.364 |

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

### 3.5. Correlation between bioreactivity and chemical components

Pearson correlation coefficients (R) between ROS-inflammatory activities at 200 μg $mL^{-1}$ p.m.$_{2.5}$ exposure and selected $PM_{2.5}$ compounds concentrations (converted to corresponding exposure dose) were calculated. Only those chemical components that demonstrated moderate to strong correlations with ROS-inflammatory responses are presented in Table 1. Moderate positive correlations were observed between 8-OHdG and 1-MNAP ($r = 0.557$, $p < 005$) as well as PHE ($r = 0.603$, $p < 005$). Strong, positive correlations were observed between 8-OHdG and BaA ($r = 0.766$, $p < 005$) as well as BaP ($r = 0.760$, $p < 005$). Moderate positive correlations were observed between TNF-α and PHE ($r = 0.626$, $p < 005$), FLT ($r = 0.575$, $p < 005$), as well as 5,12-NACQ ($r = 0.547$, $p < 005$). Moderate positive correlations were observed between IL-6 and ACY ($r = 0.624$, $p < 005$), FLO ($r = 0.597$, $p < 005$), 9-FLU ($r = 0.485$, $p < 005$), as well as 9,10-ANQ ($r = 0.489$, $p < 005$). None of the NPAH compounds showed moderate to strongly positive correlations in the oxidative-inflammatory response tests. The strong correlations of 8-OHdG with BaP and BaA suggests that these two compounds play influential roles in the DNA damage process. A previous study showed that PAHs and their derivatives in fine PM are important components that cause oxidative stress (Oh et al., 2011). These components can further induce cell cycle arrest and increasing the formation of double nuclei and micronuclei cells (Danielsen et al., 2011). The compounds of PHE, FLT, and 5,12-NACQ were well correlated with TNF-α. In addition, ACY, FLO, 9-FLU, and 9,10-ANQ were well correlated with IL-6. TNF-α is a major indicator of particle-induced inflammation in human lungs (Hetland et al., 2005; Jalava et al., 2009). These results clarify that individual PAH compounds are correlated to inflammatory responses; 9-FLU and 9,10-ANQ are derivatives of their parental PAHs, and the aforementioned findings are consistent with those of previous studies that have stated that PAH derivatives can trigger oxidative stress and inflammatory response (Niu et al., 2017; Ovrevik et al., 2010). Ho et al. (2016b) reported that some individual PAHs, azaarenes, and carbonyl-OPAHs are all correlated with cell viability after exposure to $PM_{2.5}$ emitted from coal combustion. Dergham et al. (2015) concluded that some inorganic and organic chemical compounds are preferentially associated with early oxidative stress, whereas others are responsible for later oxidative damage and cytokine secretion.

### 3.6. Correlation between bioreactivity and lung function parameters

Table 2 summarizes the correlations of the lung function parameters with volume-based bioreactivities (in pg $m^{-3}$, bioreactivity divided by dose and then multiplied by $PM_{2.5}$ mass concentration) induced by indoor $PM_{2.5}$. Outdoor $PM_{2.5}$ concentration was not included considering residents spend majority of time in indoor environment and individual outdoor activities are too complex to track. Significant correlations were observed between $FEF_{25-75}$ and oxidative and inflammatory markers. Moderate negative correlations were observed between $FEF_{25-75}$ and TNF-α ($r = -0.463$, $p < 0.01$) as well as IL-6 ($r = -0.455$, $p < 0.01$). Weak negative correlations were observed between $FEF_{25-75}$ and 8-OHdG ($r = -0.385$, $p < 0.05$). A previous study demonstrated that $PM_{2.5}$ exposure is harmful to the human respiratory system and that $PM_{2.5}$ could cause oxidative stress in pulmonary epithelia cells due to lung function decline (Madureira et al., 2015). A study on personal exposure to gaseous pollutants and $PM_{2.5}$ revealed that lifetime average and first year of life exposure to $PM_{2.5}$ is associated with reduced $FEF_{25-75}$ and $FEV_1$ (Neophytou et al., 2016). Another study reported that Chinese schoolchildren living in high-pollution districts could exhibit hindered $FEV_1$, $FEF_{25}$, and $FEF_{25-75}$ growth (He et al., 2010). The correlations between 8-OHdG and $FEF_{25-75}$ indicate that DNA damage initiated by oxidative stress in the respiratory system is associated with decreased predicted values. The correlations between TNF-α and IL-6 with $FEF_{25-75}$ highlight the crucial roles of inflammatory responses in lung function expression. A previous study showed that long-term exposure to $PM_{2.5}$ could ultimately cause lung function decline (Schultz et al., 2017). It has been proved that high exposure of smoky coal emissions would cause p53 and K-ras mutations in lung tissues (Keohavong et al., 2018). A source apportionment study in Beijing showed high associations between secondary and coal combustion sources and inflammation, indicating the major contribution of coal combustions emissions to short-term lung diseases in China (Liu et al., 2016). The aforementioned results indicate that elevated oxidative and inflammatory levels potentially affect lung function. However, the long-term effect of $PM_{2.5}$ and associations between lung function and the bioreactivities of human epithelial lung cells, together with the potential influencing factors will be required in further analysis.

**Table 2**
Pearson's correlation coefficients of lung function parameters and bioreactivity results.

| | 8-OHdG | TNF-α | IL-6 |
|---|---|---|---|
| FVC (% of predicted) | −0.074 | −0.011 | −0.112 |
| $FEV_1$ (% of predicted) | −0.161 | −0.105 | −0.011 |
| $FEV_1$/FVC (% of predicted) | −0.087 | −0.174 | −0.198 |
| $FEF_{25-75}$(% of predicted) | −0.385* | −0.463** | −0.455** |

a**Correlation is significant at the 0.01 level (2-tailed).
b* Correlation is significant at the 0.05 level (2-tailed).

## 4. Conclusions

The chemical and bioreactivity properties of $PM_{2.5}$ in Xi'an city in China were characterized. The concentrations of PAHs and their derivatives were observed to be higher in urban areas than in suburban areas due to the more concentrated emission sources and anthropogenic activities in the former. In addition, outdoor PAH concentrations were generally higher than indoor concentrations. However, several OPAHs and NPAHs were present in higher concentrations in indoor environments given the low air exchange rate and residential heating systems. Oxidative DNA damage and inflammatory responses dose-dependently increased on exposure to $PM_{2.5}$. The high levels of 8-OHdG and IL-6 at HS is likely due to frequent coal burning in indoor environments. $PM_{2.5}$ in outdoor environment have a relatively stronger impact on oxidative DNA damage and inflammatory responses, except for indoor environments with combustion sources (i.e. coal stove and tobacco smoking). PAH compounds (1-MNAP, ACY, FLO, PHE, FLT, BaA and BaP) were associated with biomarkers; several OPAH compounds (9-FLU, 9,10-ANQ, and 5,12-NACQ) were more correlated to oxidative and inflammatory responses than did their parent PAHs. Increased oxidative and inflammatory levels could result in declined lung function. However, there was a lack of detailed analysis for the adjusting covariates (i.e. age, sex, BMI, passive and active smoking, physical activity, outdoor $PM_{2.5}$ exposure and etc.) in this study due to lung function

sample size limit. The mechanism between PM-induced lung function decline and the exact organic compounds association will warrant further scrutiny.

## Declaration of competing interest

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

## Acknowledgement

This study was supported by the Research Grants Council of the Hong Kong Special Administrative Region China (Project No. CUHK 14212116), the Ministry of Science and Technology of China (2013FY112700) and National Program on Basic Research Project of China (2013FY112500). This manuscript was edited by Wallace Academic Editing.

## Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.atmosenv.2019.116929.

## References

Albinet, A., Leoz-Garziandia, E., Budzinski, H., Villenave, E., 2007. Polycyclic aromatic hydrocarbons (PAHs), nitrated PAHs and oxygenated PAHs in ambient air of the Marseilles area (South of France): concentrations and sources. Sci. Total Environ. 384, 280–292.

Bandowe, B.A.M., Meusel, H., Huang, R.J., Ho, K.F., Cao, J.J., Hoffmann, T., Wilcke, W., 2014. PM2.5-bound oxygenated PAHs, nitro-PAHs and parent-PAHs from the atmosphere of a Chinese megacity: seasonal variation, sources and cancer risk assessment. Sci. Total Environ. 473, 77–87.

Bi, X.H., Sheng, G.Y., Peng, P., Chen, Y.J., Zhang, Z.Q., Fu, J.M., 2003. Distribution of particulate- and vapor-phase n-alkanes and polycyclic aromatic hydrocarbons in urban atmosphere of Guangzhou, China. Atmos. Environ. 37, 289–298.

Charrier, J.G., McFall, A.S., Richards-Henderson, N.K., Anastasio, C., 2014. Hydrogen peroxide formation in a surrogate lung fluid by transition metals and quinones present in particulate matter. Environ. Sci. Technol. 48, 7010–7017.

Chang, K.F., Fang, G.C., Chen, J.C., Wu, Y.S., 2006. Atmospheric polycyclic aromatic hydrocarbons (PAHs) in Asia: a review from 1999 to 2004. Environ. Pollut. 142, 388–396.

Chen, Y., Li, X., Zhu, T., Han, Y., Lv, D., 2017. PM2.5-bound PAHs in three indoor and one outdoor air in Beijing: concentration, source and health risk assessment. Sci. Total Environ. 586, 255–264.

Chuang, H.C., Fan, C.W., Chen, K.Y., Chang-Chien, G.P., Chan, C.C., 2012. Vasoactive alteration and inflammation induced by polycyclic aromatic hydrocarbons and trace metals of vehicle exhaust particles. Toxicol. Lett. 214, 131–136.

Chuang, H.C., Shie, R.H., Chio, C.P., Yuan, T.H., Lee, J.H., Chan, C.C., 2018. Cluster analysis of fine particulate matter (PM2.5) emissions and its bioreactivity in the vicinity of a petrochemical complex. Environ. Pollut. 236, 591–597.

Chuang, H.C., Sun, J., Ni, H., Tian, J., Lui, K.H., Han, Y., Cao, J.J., Huang, R.J., Shen, Z.X., Ho, K.F., 2019. Characterization of the chemical components and bioreactivity of fine particulate matter produced during crop-residue burning in China. Environ. Pollut. 245, 226–234.

Dagher, Z., Garcon, G., Billet, S., Verdin, A., Ledoux, F., Courcot, D., Aboukais, A., Shirali, P., 2007. Role of nuclear factor-kappa B activation in the adverse effects induced by air pollution particulate matter (PM2.5) in human epithelial lung cells (L132) in culture. J. Appl. Toxicol. 27, 284–290.

Dagher, Z., Garcon, G., Gosset, P., Ledoux, F., Surpateanu, G., Courcot, D., Aboukais, A., Puskaric, E., Shirali, P., 2005. Pro-inflammatory effects of Dunkerque city air pollution particulate matter 2.5 in human epithelial lung cells (L132) in culture. J. Appl. Toxicol. 25, 166–175.

Danielsen, P.H., Moller, P., Jensen, K.A., Sharma, A.K., Wallin, H., Bossi, R., Autrup, H., Molhave, L., Ravanat, J.L., Briede, J.J., de Kok, T.M., Loft, S., 2011. Oxidative stress, DNA damage, and inflammation induced by ambient air and wood smoke particulate matter in human A549 and THP-1 cell lines. Chem. Res. Toxicol. 24, 168–184.

Dergham, M., Lepers, C., Verdin, A., Cazier, F., Billet, S., Courcot, D., Shirali, P., Garcon, G., 2015. Temporal-spatial variations of the physicochemical characteristics of air pollution Particulate Matter (PM2.5-0.3) and toxicological effects in human bronchial epithelial cells (SEAS-2B). Environ. Res. 137, 256–267.

Dieme, D., Cabral-Ndior, M., Garcon, G., Verdin, A., Billet, S., Cazier, F., Courcot, D., Diouf, A., Shirali, P., 2012. Relationship between physicochemical characterization and toxicity of fine particulate matter (PM2.5) collected in Dakar city (Senegal). Environ. Res. 113, 1–13.

Ding, J.N., Zhong, J.J., Yang, Y.F., Li, B.G., Shen, G.F., Su, Y.H., Wang, C., Li, W., Shen, H.Z., Wang, B., Wang, R., Huang, Y., Zhang, Y.Y., Cao, H.Y., Zhu, Y., Simonich, S.L.M., Tao, S., 2012. Occurrence and exposure to polycyclic aromatic hydrocarbons and their derivatives in a rural Chinese home through biomass fuelled cooking. Environ. Pollut. 169, 160–166.

Durant, J.L., Busby, W.F., Lafleur, A.L., Penman, B.W., Crespi, C.L., 1996. Human cell mutagenicity of oxygenated, nitrated and unsubstituted polycyclic aromatic hydrocarbons associated with urban aerosols. Mutat. Res. Genet. Toxicol. 371, 123–157.

Han, B., Bai, Z., Liu, Y., You, Y., Xu, J., Zhou, J., Zhang, J., Niu, C., Zhang, N., He, F., Ding, X., 2015. Characterizations, relationship, and potential sources of outdoor and indoor particulate matter bound polycyclic aromatic hydrocarbons (PAHs) in a community of Tianjin, Northern China. Indoor Air 25, 320–328.

He, Q.Q., Wong, T.W., Du, L., Jiang, Z.Q., Gao, Y., Qiu, H., Liu, W.J., Wu, J.G., Wong, A., Yu, T.S.I., 2010. Effects of ambient air pollution on lung function growth in Chinese schoolchildren. Respir. Med. 104, 1512–1520.

Hetland, R.B., Refsnes, M., Cassee, F.R., Lag, M., Dybing, E., Schwarze, P.E., 2005. Cytokine release from alveolar macrophages exposed to ambient particulate matter: relation to size, city, season and metal content. Toxicol. Lett. 158, S80–S81.

Ho, K.F., Ho, S.S.H., Huang, R.J., Chuang, H.C., Cao, J.J., Han, Y.M., Lui, K.H., Ning, Z., Chuang, K.J., Cheng, T.J., Lee, S.C., Hu, D., Wang, B., Zhang, R.J., 2016a. Chemical composition and bioreactivity of PM2.5 during 2013 haze events in China. Atmos. Environ. 126, 162–170.

Ho, K.F., Chang, C.C., Tian, L., Chan, C.S., Musa Bandowe, B.A., Lui, K.H., Lee, K.Y., Chuang, K.J., Liu, C.Y., Ning, Z., Chuang, H.C., 2016b. Effects of polycyclic aromatic compounds in fine particulate matter generated from household coal combustion on response to EGFR mutations in vitro. Environ. Pollut. 218, 1262–1269.

Hong, Z.Y., Hong, Y.W., Zhang, H., Chen, J.S., Xu, L.L., Deng, J.J., Du, W.J., Zhang, Y.R., Xiao, H., 2017. Pollution characteristics and source apportionment of PM2.5-bound n-alkanes in the yangtze river delta, China. Aerosol. Air Qual. Res. 17, 1985–1998.

Huang, B., Liu, M., Bi, X.H., Chaemfa, C., Ren, Z.F., Wang, X.M., Sheng, G.Y., Fu, J.M., 2014. Phase distribution, sources and risk assessment of PAHs, NPAHs and OPAHs in a rural site of Pearl River Delta region, China. Atmos. Pollut. Res. 5, 210–218.

Jalava, P.I., Hirvonen, M.R., Sillanpaa, M., Pennanen, A.S., Happo, M.S., Hillamo, R., Cassee, F.R., Gerlofs-Nijland, M., Borm, P.J.A., Schins, R.P.F., Janssen, N.A.H., Salonen, R.O., 2009. Associations of urban air particulate composition with inflammatory and cytotoxic responses in RAW 246.7 cell line. Inhal. Toxicol. 21, 994–1006.

Jariyasopit, N., Zimmermann, K., Schrlau, J., Arey, J., Atkinson, R., Yu, T.W., Dashwood, R.H., Tao, S., Simonich, S.L.M., 2014. Heterogeneous reactions of particulate matter-bound PAHs and NPAHs with NO3/N2O5, OH radicals, and O-3 under simulated long-range atmospheric transport conditions: reactivity and mutagenicity. Environ. Sci. Technol. 48, 10155–10164.

Kelly, F.J., Fussell, J.C., 2012. Size, source and chemical composition as determinants of toxicity attributable to ambient particulate matter. Atmos. Environ. 60, 504–526.

Keohavong, P., Lan, Q., Gao, W., 2018. p53 and K-ras mutations in lung tissues and sputum samples of individuals exposed to smoky coal emissions in Xuan Wei County, China. Mutat. Res. Genet. Toxicol. Environ. Mutagen. 829, 70–74.

Knecht, A.L., Goodale, B.C., Truong, L., Simonich, M.T., Swanson, A.J., Matzke, M.M., Anderson, K.A., Waters, K.M., Tanguay, R.L., 2013. Comparative developmental toxicity of environmentally relevant oxygenated PAHs. Toxicol. Appl. Pharmacol. 271, 266–275.

Kojima, Y., Inazu, K., Hisamatsu, Y., Okochi, H., Baba, T., Nagoya, T., 2010. Influence of secondary formation on atmospheric occurrences of oxygenated polycyclic aromatic hydrocarbons in airborne particles. Atmos. Environ. 44, 2873–2880.

Lai, C.H., Lee, C.N., Bai, K.J., Yang, Y.L., Chuang, K.J., Wu, S.M., Chuang, H.C., 2016. Protein oxidation and degradation caused by particulate matter. Sci. Rep. 6, 33727.

Li, W., Wang, C., Shen, H.Z., Su, S., Shen, G.F., Huang, Y., Zhang, Y.Y., Chen, Y.C., Chen, H., Lin, N., Zhuo, S.J., Zhong, Q.R., Wang, X.L., Liu, J.F., Li, B.G., Liu, W.X., Tao, S., 2015. Concentrations and origins of nitro-polycyclic aromatic hydrocarbons and oxy-polycyclic aromatic hydrocarbons in ambient air in urban and rural areas in northern China. Environ. Pollut. 197, 156–164.

Li, W., Wang, C., Wang, H.Q.J., Chen, J.W., Shen, H.Z., Shen, G.F., Huang, Y., Wang, R., Wang, B., Zhang, Y.Y., Chen, H., Chen, Y.C., Su, S., Lin, N., Tang, J.H., Li, Q.B., Wang, X.L., Liu, J.F., Tao, S., 2014. Atmospheric polycyclic aromatic hydrocarbons in rural and urban areas of northern China. Environ. Pollut. 192, 83–90.

Lin, Y., Ma, Y.Q., Qiu, X.H., Li, R., Fang, Y.H., Wang, J.X., Zhu, Y.F., Hu, D., 2015. Sources, transformation, and health implications of PAHs and their nitrated, hydroxylated, and oxygenated derivatives in PM2.5 in Beijing. J. Geophys. Res. D Atmos. 120, 7219–7228.

Liu, Q., Baumgartner, J., Zhang, Y., Schauer, J.J., 2016. Source apportionment of Beijing air pollution during a severe winter haze event and associated pro-inflammatory responses in lung epithelial cells. Atmos. Environ. 126, 28–35.

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. Cancer risk from polycyclic aromatic compounds in fine particulate matter generated from household coal combustion in Xuanwei, China. Chemosphere 169, 660–668.

Madureira, J., Paciencia, I., Rufo, J., Ramos, E., Barros, H., Teixeira, J.P., Fernandes, E.D., 2015. Indoor air quality in schools and its relationship with children's respiratory symptoms. Atmos. Environ. 118, 145–156.

Mirowsky, J., Hickey, C., Horton, L., Blaustein, M., Galdanes, K., Peltier, R.E., Chillrud, S., Chen, L.C., Ross, J., Nadas, A., Lippmann, M., Gordon, T., 2013. The effect of particle size, location and season on the toxicity of urban and rural particulate matter. Inhal. Toxicol. 25, 747–757.

Mortimer, K.M., Fallot, A., Balmes, J.R., Tager, I.B., 2003. Evaluating the use of a portable spirometer in a study of pediatric asthma. Chest 123, 1899–1907.

Naumova, Y.Y., Eisenreich, S.J., Turpin, B.J., Weisel, C.P., Morandi, M.T., Colome, S.D., Totten, L.A., Stock, T.H., Winer, A.M., Alimokhtari, S., Kwon, J., Shendell, D., Jones, J., Maberti, S., Wall, S.J., 2002. Polycyclic aromatic hydrocarbons in the indoor and

outdoor air of three cities in the US. Environ. Sci. Technol. 36, 2552–2559.
Nel, A.E., Diaz-Sanchez, D., Li, N., 2001. The role of particulate pollutants in pulmonary inflammation and asthma: evidence for the involvement of organic chemicals and oxidative stress. Curr. Opin. Pulm. Med. 7, 20–26.
Neophytou, A.M., White, M.J., Oh, S.S., Thakur, N., Galanter, J.M., Nishimura, K.K., Pino-Yanes, M., Torgerson, D.G., Gignoux, C.R., Eng, C., Nguyen, E.A., Hu, D.L., Mak, A.C., Kumar, R., Seibold, M.A., Davis, A., Farber, H.J., Meade, K., Avila, P.C., Serebrisky, D., Lenoir, M.A., Brigino-Buenaventura, E., Rodriguez-Cintron, W., Bibbins-Domingo, K., Thyne, S.M., Williams, L.K., Sen, S., Gilliland, F.D., Gauderman, W.J., Rodriguez-Santana, J.R., Lurmann, F., Balmes, J.R., Eisen, E.A., Burchard, E.G., 2016. Air pollution and lung function in minority youth with asthma in the GALA II (Genes-Environments and admixture in latino Americans) and SAGE II (study of african Americans, asthma, genes, and environments) studies. Am. J. Respir. Crit. Care Med. 193, 1271–1280.
Niu, X.Y., Ho, S.S.H., Ho, K.F., Huang, Y., Sun, J., Wang, Q.Y., Zhou, Y.Q., Zhao, Z.Z., Cao, J.J., 2017. Atmospheric levels and cytotoxicity of polycyclic aromatic hydrocarbons and oxygenated-PAHs in PM2.5 in the Beijing-Tianjin-Hebei region. Environ. Pollut. 231, 1075–1084.
Oh, S.M., Kim, H.R., Park, Y.J., Lee, S.Y., Chung, K.H., 2011. Organic extracts of urban air pollution particulate matter (PM2.5)-induced genotoxicity and oxidative stress in human lung bronchial epithelial cells (BEAS-2B cells). Mutat. Res. Genet. Toxicol. Environ. Mutagen. 723, 142–151.
Okuda, T., Okamoto, K., Tanaka, S., Shen, Z.X., Han, Y.M., Huo, Z.Q., 2010. Measurement and source identification of polycyclic aromatic hydrocarbons (PAHs) in the aerosol in Xi'an, China, by using automated column chromatography and applying positive matrix factorization (PMF). Sci. Total Environ. 408, 1909–1914.
Ovrevik, J., Arlt, V.M., Oya, E., Nagy, E., Mollerup, S., Phillips, D.H., Lag, M., Holme, J.A., 2010. Differential effects of nitro-PAHs and amino-PAHs on cytokine and chemokine responses in human bronchial epithelial BEAS-2B cells. Toxicol. Appl. Pharmacol. 242, 270–280.
Perrone, M.G., Gualtieri, M., Ferrero, L., Lo Porto, C., Udisti, R., Bolzacchini, E., Camatini, M., 2010. Seasonal variations in chemical composition and in vitro biological effects of fine PM from Milan. Chemosphere 78, 1368–1377.
Ren, Y.Q., Zhou, B.H., Tao, J., Cao, J.J., Zhang, Z.S., Wu, C., Wang, J.Y., Li, J.J., Zhang, L., Han, Y.N., Liu, L., Cao, C., Wang, G.H., 2017. Composition and size distribution of airborne particulate PAHs and oxygenated PAHs in two Chinese megacities. Atmos. Res. 183, 322–330.
Ringuet, J., Albinet, A., Leoz-Garziandia, E., Budzinski, H., Villenave, E., 2012. Reactivity of polycyclic aromatic compounds (PAHs, NPAHs and OPAHs) adsorbed on natural aerosol particles exposed to atmospheric oxidants. Atmos. Environ. 61, 15–22.
Risom, L., Moller, P., Loft, S., 2005. Oxidative stress-induced DNA damage by particulate air pollution. Mutat. Res. Fundam. Mol. Mech. Mutagen. 592, 119–137.
Salo, P.M., Jiang, X., Johnson, C.A., Yan, L., Avol, E.L., Gong, J., London, S.J., 2004. Indoor allergens, asthma, and asthma-related symptoms among adolescents in Wuhan, China. Ann. Epidemiol. 14, 543–550.
Sauvain, J.J., Setyan, A., Wild, P., Tacchini, P., Lagger, G., Storti, F., Deslarzes, S., Guillemin, M., Rossi, M.J., Riediker, M., 2011. Biomarkers of oxidative stress and its association with the urinary reducing capacity in bus maintenance workers. J. Occup. Med. Toxicol. 6, 13.
Schultz, E.S., Litonjua, A.A., Melen, E., 2017. Effects of long-term exposure to traffic-related air pollution on lung function in children. Curr. Allergy Asthma Rep. 17, 13.
Shi, T.M., Duffin, R., Borm, P.J.A., Li, H., Weishaupt, C., Schins, R.P.F., 2006. Hydroxyl-radical-dependent DNA damage by ambient particulate matter from contrasting sampling locations. Environ. Res. 101, 18–24.
Shen, M.L., Xing, J., Ji, Q.P., Li, Z.H., Wang, Y.H., Zhao, H.W., Wang, Q.R., Wang, T., Yu, L.W., Zhang, X.C., Sun, Y.X., Zhang, Z.H., Niu, Y., Wang, H.Q., Chen, W., Dai, Y.F., Su, W.G., Duan, H.W., 2018. Declining pulmonary function in populations with long-term exposure to polycyclic aromatic hydrocarbons-enriched PM2.5. Environ. Sci. Technol. 52, 6610–6616.
Sorensen, M., Daneshvar, B., Hansen, M., Dragsted, L.O., Hertel, O., Knudsen, L., Loft, S., 2003. Personal PM2.5 exposure and markers of oxidative stress in blood. Environ. Health Perspect. 111, 161–165.
Steenhof, M., Gosens, I., Strak, M., Godri, K.J., Hoek, G., Cassee, F.R., Mudway, I.S., Kelly, F.J., Harrison, R.M., Lebret, E., Brunekreef, B., Janssen, N.A.H., Pieters, R.H.H., 2011. In vitro toxicity of particulate matter (PM) collected at different sites in The Netherlands is associated with PM composition, size fraction and oxidative potential - the RAPTES project. Part. Fibre Toxicol. 8, 15.
Storr, S.J., Woolston, C.M., Zhang, Y.M., Martin, S.G., 2013. Redox environment, free radical, and oxidative DNA damage. Antioxidants Redox Signal. 18, 2399–2408.
Suhaimi, N.F., Jalaludin, J., 2015. Biomarker as a Research tool in linking exposure to air particles and respiratory health. BioMed Res. Int. 10.
Totlandsdal, A.I., Cassee, F.R., Schwarze, P., Refsnes, M., Lag, M., 2010. Diesel exhaust particles induce CYP1A1 and pro-inflammatory responses via differential pathways in human bronchial epithelial cells. Part. Fibre Toxicol. 7, 13.
Vattanasit, U., Navasumrit, P., Khadka, M.B., Kanitwithayanun, J., Promvijit, J., Autrup, H., Ruchirawat, M., 2014. Oxidative DNA damage and inflammatory responses in cultured human cells and in humans exposed to traffic-related particles. Int. J. Hyg Environ. Health 217, 23–33.
Vione, D., Barra, S., De Gennaro, G., De Rienzo, M., Gilardoni, S., Perrone, M.G., Pozzoli, L., 2004. Polycyclic aromatic hydrocarbons in the atmosphere: monitoring, sources, sinks and fate. II: sinks and fate. Environ. Cult. Herit. Chem. 94, 257–268.
Wang, J.Z., Cao, J.J., Dong, Z.B., Guinot, B., Gao, M.L., Huang, R.J., Han, Y.M., Huang, Y., Ho, S.S.H., Shen, Z.X., 2017. Seasonal variation, spatial distribution and source apportionment for polycyclic aromatic hydrocarbons (PAHs) at nineteen communities in Xi'an, China: the effects of suburban scattered emissions in winter. Environ. Pollut. 231, 1330–1343.
Wang, J.Z., Ho, S.S.H., Huang, R.J., Gao, M.L., Liu, S.X., Zhao, S.Y., Cao, J.J., Wang, G.H., Shen, Z.X., Han, Y.M., 2016. Characterization of parent and oxygenated-polycyclic aromatic hydrocarbons (PAHs) in Xi'an, China during heating period: an investigation of spatial distribution and transformation. Chemosphere 159, 367–377.
Wang, W.T., Jariyasopit, N., Schrlau, J., Jia, Y.L., Tao, S., Yu, T.W., Dashwood, R.H., Zhang, W., Wang, X.J., Simonich, S.L.M., 2011. Concentration and photochemistry of PAHs, NPAHs, and OPAHs and toxicity of PM2.5 during the Beijing olympic games. Environ. Sci. Technol. 45, 6887–6895.
Wei, P., Ning, Z., Ye, S., Sun, L., Yang, F., Wong, K.C., Westerdahl, D., Louie, P.K.K., 2018. Impact analysis of temperature and humidity conditions on electrochemical sensor response in ambient air quality monitoring. Sensors 18 (2), 59.
Xu, H.M., Ho, S.S.H., Gao, M.L., Cao, J.J., Guinot, B., Ho, K.F., Long, X., Wang, J.Z., Shen, Z.X., Liu, S.X., Zheng, C.L., Zhang, Q., 2016. Microscale spatial distribution and health assessment of PM2.5-bound polycyclic aromatic hydrocarbons (PAHs) at nine communities in Xi'an, China. Environ. Pollut. 218, 1065–1073.
Zeng, X., Xu, X.J., Zheng, X.B., Reponen, T., Chen, A.M., Huo, X., 2016. Heavy metals in PM2. 5 and in blood, and children's respiratory symptoms and asthma from an e-waste recycling area. Environ. Pollut. 210, 346–353.
Zhang, Y., Tao, S., Shen, H., Ma, J., 2009. Inhalation exposure to ambient polycyclic aromatic hydrocarbons and lung cancer risk of Chinese population. Proc. Natl. Acad. Sci. PNAS., 0905756106.
Zhao, Z.H., Zhang, Z., Wang, Z.H., Ferm, M., Liang, Y.L., Norback, D., 2008. Asthmatic symptoms among pupils in relation to winter indoor and outdoor air pollution in schools in Taiyuan, China. Environ. Health Perspect. 116, 90–97.
Zimmermann, K., Jariyasopit, N., Simonich, S.L.M., Tao, S., Atkinson, R., Arey, J., 2013. formation of nitro-PAHs from the heterogeneous reaction of ambient particle-bound PAHs with N2O5/NO3/NO2. Environ. Sci. Technol. 47, 8434–8442.