

Research Article

Effect of NPAH Exposure on Lung Function of Children in Indoor Coal Combustion Environment

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Received 26 September 2023; Revised 3 April 2024; Accepted 5 April 2024; Published 23 April 2024

Academic Editor: Faming Wang

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Nitropolycyclic aromatic hydrocarbon (NPAH) emissions from the combustion of household solid fuel may cause great harm to public health. Children are one of the most susceptible population groups at risk of indoor air pollutants due to their immature respiratory and immune systems. In this study, a primary school using household coal combustion for heating in winter was selected and forty participants were randomly recruited among schoolchildren. Fine particulate matter samples were collected by both individual portable samplers and fixed middle-flow samplers during the heating and nonheating seasons. The NPAH concentrations in PM_{2.5} samples were analyzed by a gas chromatograph coupled to a mass spectrometer. Potential sources of NPAHs were identified by NPAH ratios as well as principal component analysis. Lung function of children was tested by an electronic spirometer. The relationship between NPAH exposure level and children's lung function was studied. Finally, the cancer risk caused by NPAH inhalation was assessed. The results showed significantly higher individual NPAH exposure level in heating season ($0.901 \pm 0.396 \text{ ng}\cdot\text{m}^{-3}$) than that in nonheating season ($0.094 \pm 0.107 \text{ ng}\cdot\text{m}^{-3}$). Coal/biomass combustion and secondary formation were the potential NPAH sources in heating season. Significantly lower lung function of children was also found in heating season compared with that in nonheating season. As a result of the Monte Carlo simulation, the averaged incremental lifetime cancer risk (ILCR) values from the inhalation of NPAHs in the heating and nonheating seasons were 3.50×10^{-8} and 2.13×10^{-8} , respectively. Our research revealed the association between NPAH exposure and children's lung function and confirmed the adverse effect of indoor coal combustion. The results also indicated that further control strategies on indoor coal combustion are needed to reduce the risk of NPAH exposure and protect children's health.

1. Introduction

Nitrated polycyclic aromatic hydrocarbons (NPAHs) are derivatives of PAHs with at least one nitro functional group on the aromatic ring [1]. NPAHs can be generated from both incomplete combustion processes and by the reaction of parent PAHs with atmospheric oxidants, such as N₂O₅, NO₃, and O₂ radicals in the presence of nitrogen oxides [2, 3]. Researchers have found an association between PAHs and obstructive pulmonary disease, including asthma and weakening respiratory utility because PAHs can be involved in the generation of reactive oxygen species, which induce inflammatory responses in human cells [4–6]. NPAHs have

been proven to be 100,000 times more mutagenic and 10 times more carcinogenic compared to the parent PAHs because of the presence of the nitro functional groups [7, 8]. Toxicological experiments at the cellular level provide evidence that acute 1-nitropyrene (1N-PYR) exposure induces inflammatory responses through activating various inflammatory signaling pathways in mouse lungs and human A549 cells [9]. Evidence of epidemiologic studies also showed that the correlation existed between lung cancers and presence of PAHs and NPAHs in the air of industrialized regions [10]. Therefore, NPAH exposure from inhalation has been a topic of public concern due to its extensive sources and toxicity.

Household coal combustion can be a potential source of NPAHs [11], especially inefficient burning of solid fuels, which caused severe indoor NPAH pollution [12]. For developing countries relying on solid fuel for household cooking and heating, it has been estimated that the concentrations of pollutants including PAHs and NPAHs are two- to fivefold higher indoors than in the outdoor environment due to unvented cooking stoves and poor ventilation [13]. However, household solid fuel was still widely used for heating and cooking in rural areas in North China. The exposure experienced by children who spend most of their time at home may be much higher than the national standards and World Health Organization (WHO) guidelines.

Children are among susceptible population groups, because their respiratory, immune, central nervous, and digestive systems are not fully developed [14], which makes them at higher risks for the potential health effects induced by air pollution [15]. According to the survey of behavior patterns, Chinese children spend up to 10 h day⁻¹ at school environments [16]. Immature respiratory and immune systems and long indoor exposure time make children more susceptible to indoor pollution. Specific indoor air pollutants and lung diseases among children have been reported in previous research [17–21]. Decrements in lung function in children during episodes of air pollution have also been recognized [22–24]. In vivo and in vitro studies have also been done to study the toxicological effect of PM_{2.5}-bound PAHs on the lung [25]. However, personal exposure levels of NPAHs based on individual samplers are seldom reported. Moreover, the association between indoor NPAH concentration and lung functions is still unknown [26].

In this study, a typical school in North China using indoor coal combustion for heating was selected as the research area and children in the school were selected as participants. PM_{2.5} samples were collected in heating and nonheating seasons using both individual portable samplers and fixed middle-flow samplers. NPAHs attached to PM_{2.5} were measured using gas chromatograph-mass spectrometer method. Children's lung function was measured by a portable spirometer. Carcinogenic risk was estimated based on NPAH exposure dose. The aim of this research was to reveal the associations of inhalation dose of NPAHs in PM_{2.5} with lung function parameters. The results will provide information for the adverse health effect control of NPAH exposure on children.

2. Methods

2.1. Research Area and Participants. Shanxi Province, located in the north of China, is a center of coal-related electricity production and chemical industries [6]. Due to the rapid development of coal-related industries, Shanxi is currently facing a serious air pollution problem. Xinzhou City, located in the north of Shanxi, is an important coal-producing area. According to the statistics of 2021, Xinzhou produced 106.9 million tons of raw coal, accounting for 8.89% of Shanxi Province [27]. Xinzhou City belongs to a temperate continental monsoon climate, with severe cold

winter. Period from mid-October to March of the following year is the heating season in Xinzhou. During heating season, indoor coal combustion is the main method of heating for residents. For the school where the research was conducted, coal stove is the main indoor heating equipment in classrooms during heating season. To study the adverse effect of NPAHs on children, a primary school located in the northeast of Xinzhou City was selected as the research area (Figure S1). There were 50 students in the primary school, including 56% (28) boys and 44% (22) girls. The age of the students ranged from 8 to 15 years with an average of 10.3 ± 1.5 years old. Students were enrolled in the inhalation exposure measurement depending on their own willingness. As a result, a total of 18 and 22 children were selected as participants in heating and nonheating seasons, respectively. A total of 11 children participated in two individual sampling sessions in heating and nonheating seasons. The basic information of participants is shown in Table S1.

2.2. Sampling and Analysis. Children's individual exposure level and environmental residual level were estimated based on the particle samples collected by individual portable samplers and fixed middle-flow samplers, respectively. The sampling flow rate of individual portable samplers (Buck Libra Plus LP-4, USA) was 0.2 L min⁻¹. PM_{2.5} samples were collected on quartz fiber filters (prebaked at 450°C for 12 h to remove organic contaminants), and the individual sampling pump was put in a suitable crossbody bag with a tube connecting the PM_{2.5} sampling head. The equipment was carried by the participants to assess their personal inhalation exposure levels. The participants were requested to take the samplers for 48 consecutive hours, with the sampler being kept within one meter while sleeping or using the bathroom. Samples were identified to be valid only when the pumps were in operation when being returned to the school on the second day, with the sampling duration among 22 h to 26 h. A sum of 18 and 22 valid personal samples was collected. Middle-flow samplers were set both in and out of the classroom, using a middle-volume air sampler at a flow rate of 0.1 m³ min⁻¹ with a 2.5 μm inlet. PM_{2.5} was collected on the same setting of sampling material for 48 h. Scrub all the metal frames that are in contact with the filter membrane with the treated absorbent cotton, and scrub a small amount several times. After sampling, put the filter back into the filter box and store the sample in a -20°C refrigerator.

Pretreatment of particle-associated NPAHs followed the established procedure in previous works [28]. Briefly, the filter samples were extracted by a microwave accelerated extraction system (CEM, MARSXpress, USA) and then transferred to a silica/alumina gel column for purification. After adding the deuterated standards of 9-nitroanthracene-d₉, 3-nitrofluoranthene-d₉, and 6-nitrochrysene-d₁₁, the samples were analyzed by a gas chromatograph coupled with a mass spectrometer (GC-MS, Agilent GC 7890B, MSD 977A, USA). The details of the pretreatment and instrumental analysis are provided in SI (Text S1). The NPAHs measured in this study (Figure S2, Table S2)

include 1-nitronaphthalene (1N-NAP), 2-nitronaphthalene (2N-NAP), 5-nitroacenaphthene (5N-ACE), 2-nitrofluorene (2N-FLO), 6-nitrochrysene (6N-CHR), 9-nitroanthracene (9N-ANT), 2+3-nitrofluoranthene (2+3N-FLA), 9-nitrophenanthrene (9N-PHE), 3-nitrophenanthrene (3N-PHE), 1-nitropyrene, 7-nitrobenzo[a] anthracene (7N-BaA), and 6-nitrobenzo[a]pyrene (6N-BaP). The spiked sample recovery tests indicated that recoveries of NPAHs were between 67.6% (6N-BaP) and 89.4% (1N-PYR).

2.3. Lung Function Test. The students' lung function was tested by a professional with an electronic spirometer (Chestgraph HI-101, CHEST Ltd, and Tokyo, Japan). During the test, each participant was asked to keep seated and had their nose clipped after more than 5 minutes of normal breathing. Three lung function parameters, including forced expiratory volume in one second (FEV1), forced vital capacity (FVC), and peak expiratory flow (PEF), were measured three times for each participant, and the highest values for FEV1, FVC, and PEF among three tests were recorded. Besides lung function parameters, the children's tidal volume (TV) (refers to the amount of air inhaled or exhaled each time when breathing calmly) and the breathing rate (BF) were measured using the electronic spirometer to calculate inhalation rate (IR).

2.4. Carcinogenic Risk Assessment. The risk assessment model recommended by US EPA [29] was applied to evaluate the cancer risk of NPAHs detected. The incremental lifetime cancer risk (ILCR) of the studied population attributable to the inhalation of NPAHs was estimated using the following equation:

$$\text{ILCR} = \frac{\text{SF} \times \text{C} \times \text{IR} \times \text{EF} \times \text{ED} \times \text{CF}}{\text{BW} \times \text{AT}}, \quad (1)$$

where SF is the cancer slope factor for BaP inhalation exposure per $\text{mg}\cdot\text{kg}^{-1}\cdot\text{day}^{-1}$, C is the daily inhalation exposure level to BaP_{eq} ($\text{ng}\cdot\text{day}^{-1}$), EF is the exposure frequency ($\text{day}\cdot\text{year}^{-1}$), ED is the exposure duration (year), CF is the conversion factor ($10^{-6}\text{mg}\cdot\text{ng}^{-1}$), BW is the body weight (kg), and AT is the average lifespan for carcinogens. Inhalation rates (IR, m^3d^{-1}) were calculated as

$$\text{IR} = \frac{60 \times 24 \times \text{TV} \times \text{BF}}{1000}, \quad (2)$$

where TV is the children's tidal volume (L) and BF is the breathing rate ($\text{times}\cdot\text{min}^{-1}$).

Benzo(a)pyrene (BaP) is commonly used as a reference chemical for NPAH because of its clearly characterized toxicity. The toxic equivalent factor (TEF) relative to BaP was used to assess the carcinogenic risk for each NPAH. In the end, the BaP equivalent (BaP_{eq} , carcinogenic equivalents) concentration was obtained for each NPAH by multiplying the NPAH concentration with its corresponding TEF. TEF is the toxic equivalency factor of the NPAHs (Table 1). TEF values for NPAHs were obtained from the literature

[30]. The BaP_{eq} based on the BaP toxicity was incorporated using the following equation:

$$\text{BaP}_{\text{eq}} = \sum C_i \times \text{TEF}_i, \quad (3)$$

where C_i is the concentration of the NPAH species in particles and TEF_i is the toxic equivalence factor of the congener of NPAHs, i .

Risk assessment uncertainty and parameter sensitivity were quantitatively assessed by the Monte Carlo simulation. NPAH concentrations, body weight, and IR were treated as variables. Based on the results of the Shapiro-Wilk tests on the field measured data, we assumed that NPAH concentrations can be described by a log-normal distribution. Body weight and IR can be fitted by normal distributions. The probability distribution types and parameters are shown in Table 1. The Monte Carlo simulation is a computational technique used in the probabilistic risk assessment. Since risk estimation based on the Monte Carlo simulation allows considerably greater accuracy in describing the variables' uncertainty, it makes the assessment more informative to risk managers. In this study, the Monte Carlo simulations of 10,000 iterations were used to generate probability distributions describing the range of results. The sensitivities of relative parameters were quantitatively assessed by comparing their Spearman's correlation rank coefficients with ILCR values. The Monte Carlo simulation was conducted by Crystal Ball software.

2.5. Other Analysis. A total of 15 priority-controlled parent PAHs were collected and analyzed at the same time (Figure S3, Table S3). The experiment method has been reported in previous research [28]. The concentration of 14 priority-controlled PAHs was used to study the relationship between parent PAHs and NPAHs. The concentration of pyrene (PYR) was applied to characterize the potential sources. The respondents' height and body weight were obtained through field measurement. Principal component analysis (PCA) was conducted to investigate the possible source of NPAHs. Statistical analysis including PCA, the Kruskal-Wallis test, and the Shapiro-Wilk test was conducted by SPSS 21.0. More details of PCA are described in Text S2.

3. Results and Discussion

3.1. Concentrations of NPAHs and Seasonal Variation. The concentrations of NPAHs in the particulate phase from portable sampler and indoor and outdoor fixed samplers are shown in Figure 1. For individual portable samplers, 12 NPAHs were detected in the heating period, while only 8 kinds of NPAHs were detected in the nonheating period. The concentration of individual total NPAHs (ΣNPAHs) in heating season varied from $0.416\text{ng}\cdot\text{m}^{-3}$ to $1.683\text{ng}\cdot\text{m}^{-3}$ with a mean value of $0.901 \pm 0.396\text{ng}\cdot\text{m}^{-3}$. In contrast, ΣNPAHs in nonheating season varied from 0.022 to $0.394\text{ng}\cdot\text{m}^{-3}$ with a mean value of $0.094 \pm 0.107\text{ng}\cdot\text{m}^{-3}$. As a result of the Kruskal-Wallis test, a significant difference was detected between heating and nonheating seasons

TABLE 1: Parameters used in the Monte Carlo simulation during heating and nonheating seasons.

Definition	Units	Distribution mode	Heating	Nonheating	TEF
2N-FLO	ng·m ⁻³	Log-normal	LN (-5.35, 0.63)	LN (-6.23, 0.73)	0.01
9N-ANT	ng·m ⁻³	Log-normal	LN (-1.01, 0.67)	LN (-4.90, 0.49)	0.0032
9N-PHE	ng·m ⁻³	Log-normal	LN (-4.86, 0.79)	LN (-7.05, 1.21)	0.006
1N-PYR	ng·m ⁻³	Log-normal	LN (-3.14, 1.18)	LN (-4.14, 1.79)	0.1
6N-CHR	ng·m ⁻³	Log-normal	LN (-5.07, 0.59)	LN (-7.70, 1.94)	10
2 + 3N-FLA	ng·m ⁻³	Log-normal	LN (-1.39, 0.25)	LN (-4.60, 0.52)	0.0026
IR	m ³ d ⁻¹	Normal	N (35.1, 15.0)	N (42.1, 12.5)	
EF	Day year ⁻¹	Constant	140	220	
ED	Year	Constant	10	10	
AT	Day	Constant	25550	25550	
BW	kg	Normal	N (29.9, 5.6)	N (29.9, 5.6)	

($p < 0.01$). In addition, NPAHs also exhibited various profiles in heating and nonheating seasons. In heating season, 9N-ANT had the greatest concentration (0.440 ± 0.254 ng·m⁻³) followed by 2+3N-FLA (0.257 ± 0.060 ng·m⁻³). In nonheating season, 1N-PYR had the greatest concentration (0.056 ± 0.096 ng·m⁻³). The individual exposure concentration of \sum NPAHs in heating season was greater than the values reported in Beijing with values of 0.25 ± 3.61 ng·m⁻³, respectively [31].

For fixed samplers, indoor concentration and outdoor concentrations were monitored separately. The averaged indoor \sum NPAH concentrations in heating and nonheating seasons were 0.420 ng·m⁻³ and 0.045 ng·m⁻³, respectively. The outdoor \sum NPAH concentrations in heating and nonheating seasons were 0.633 ng·m⁻³ and 0.039 ng·m⁻³, respectively. Obvious seasonal variation can be found in both two sampling sites. The total indoor \sum NPAH concentrations in this study were comparative in magnitude to data reported in Jinan, with value of 0.458 ng·m⁻³ [32, 33]. Compared with data reported abroad, the outdoor NPAH level in our research was also lower than in a previous study from Nepal dust samples with a mean value of 9.193 ng·m⁻³ [3].

3.2. Influencing Factors of Individual NPAH Exposure Levels

3.2.1. Relationship between Individual and Environment Concentrations. Individual exposure can be influenced by both NPAHs in classroom and outdoor, so the relationship between individual exposure level and environment concentration was analyzed by Spearman's correlation analysis (Figure 2). The results showed that a significant correlation can be found between individual concentration (C_{ind}) and both indoor (C_{indoor}) and outdoor ($C_{outdoor}$) NPAH residual levels ($p < 0.01$) in heating season. Linear regression models were established to quantitatively characterize their relationship. Among the regression models, C_{indoor} ($R^2 = 0.86$) in heating season had the best relationship with C_{ind} , followed by $C_{outdoor}$ in heating season ($R^2 = 0.70$). In nonheating season, although a significant correlation can also be found between environment concentration and individual exposure (Spearman's correlation, $p < 0.01$), the R^2 values of regression models were much lower, which indicated that

the influence of C_{indoor} and $C_{outdoor}$ on C_{ind} was not as obvious as in the heating season. Due to the lack of NPAH content data from participants' homes and other scenarios, it is difficult to accurately calculate the contribution of environmental concentrations inside and outside the classroom to total individual NPAH exposure of children. However, the results of correlation analysis showed that the NPAH concentration profiles of individual and environmental concentrations were similar. It indicated that the school environment, especially the indoor environment, had a significant contribution to individual student exposure, which was consistent with previous research. Downward et al. [34] reported that the indoor burning of coal caused PAH exposure and further led to the high cancer rates in some counties of Yunnan Province in China. Indoor coal burning in winter can be an important cause of high NPAH exposure among schoolchildren.

3.2.2. Influence of Parent PAHs. The reaction of parent PAHs with atmospheric oxidants is a potential source of NPAHs. Therefore, parent PAH concentration can be a factor influencing the individual NPAH exposure level. Correlation analysis was applied to study the potential relationship. According to the heat maps in Figure 3, a close relationship can be found between parent PAHs and NPAHs in heating season, especially parent PAHs with more than four rings have obvious influence on NPAHs. Significant correlations were found between PYR and NPYR ($p < 0.001$). In addition, $p < 0.01$ correlations were found between PYR and the other eight NPAHs. 2N-PYR and 2+3N-FLA are considered the typical chemicals formed in the secondary reaction [35]. Significant positive correlations were detected between 2+3N-FLA and 10 parent PAHs, respectively. A $p < 0.01$ correlation was also found between 2+3N-FLA, 7N-BaA, and their parent PAHs.

Compared with the heating season, less significant correlations were found in the nonheating season. No significant correlation was detected between most NPAHs and their parent PAHs except BaA and CHR. The results indicated a low influence of the parent PAHs on the formation of NPAHs. The reaction of parent PAHs with atmospheric oxidants and diesel combustion is usually considered as two

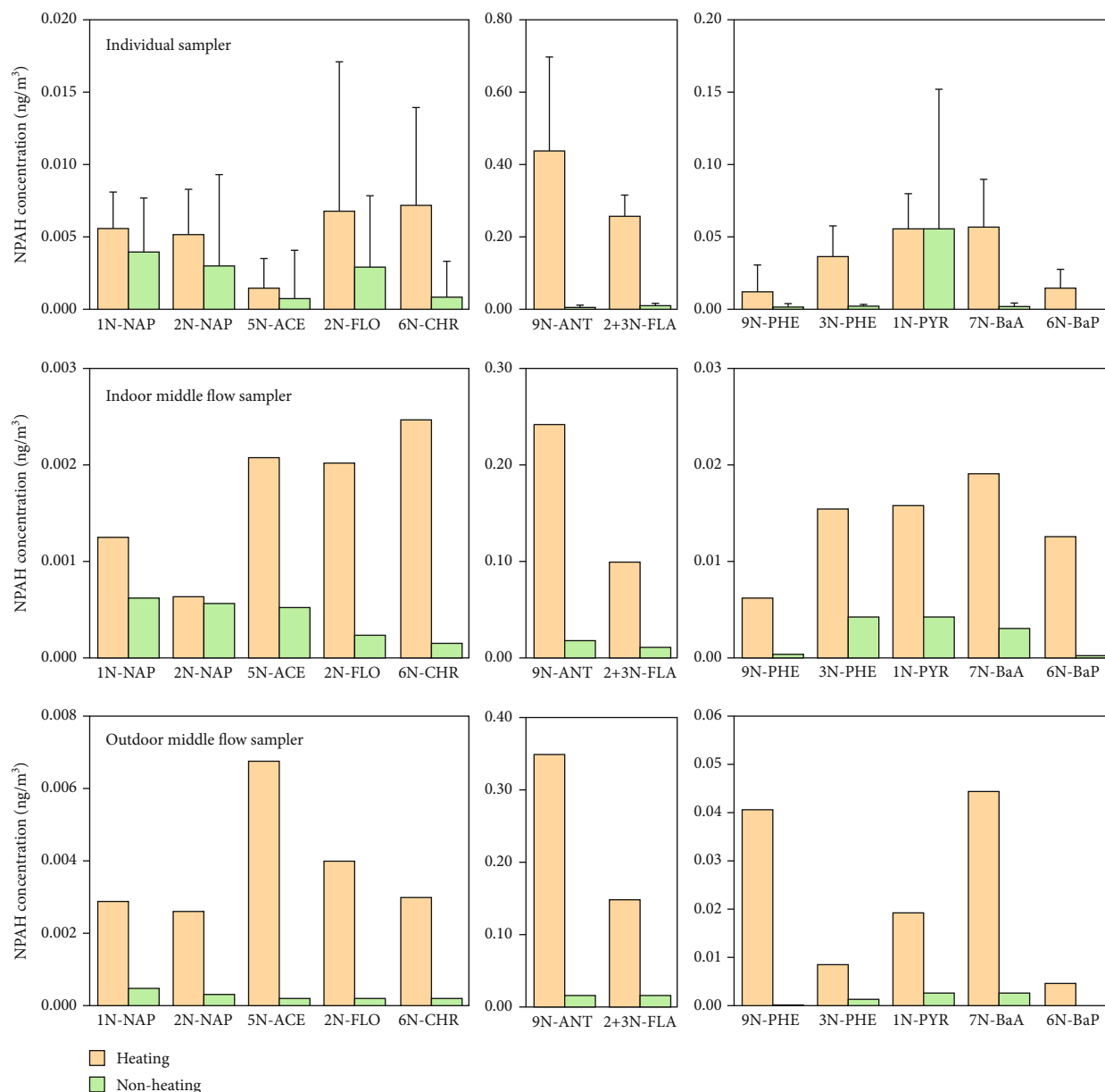


FIGURE 1: NPAH concentrations measured by individual portable sampler and fixed middle-flow sampler during heating and nonheating periods.

major sources of NPAHs. There are limited reports on the direct emission of NPAHs by coal combustion. However, our study suggests that the parent PAHs produced by coal combustion provide the necessary conditions for the secondary generation of NPAHs. The potential sources were further detected by applying the specific concentration ratios and PCA.

3.3. Source Apportionment of NPAHs. Source-specific concentration ratios were usually used to identify the potential sources of NPAHs. Common ratios used for NPAHs include 2+3N-FLA/1N-PYR, 9N-ANT/1N-PYR, and 1N-PYR/PYR. For 2+3N-FLA/1N-PYR, a ratio of 5 is usually defined as the transition point of primary sources versus atmospheric gas-phase formation. Researches have reported that ratios

of 2+3N-FLA/1N-PYR that are less than 5 are typically observed at sites near primary emission sources and ratios that are greater than 5 suggest that the source is the secondary formation of nitro-PAHs [2]. The 2+3N-FLA/1N-PYR of individual samplers in heating season ranged from 3.309 to 5.693 with an average value of 4.428 ± 0.692 . The ratios in nonheating season ranged from 0.022 to 4.277 with an average value of 1.676 ± 1.633 . The ratios in heating season were near 5, which suggested that the secondary reaction was a potential source of the NPAHs, while the values in nonheating season were significantly lower than five, which indicated that the sources of NPAHs at nonheating seasons were mainly generated from primary emissions.

In addition to identifying primary and secondary sources, some ratios are used to identify more specific

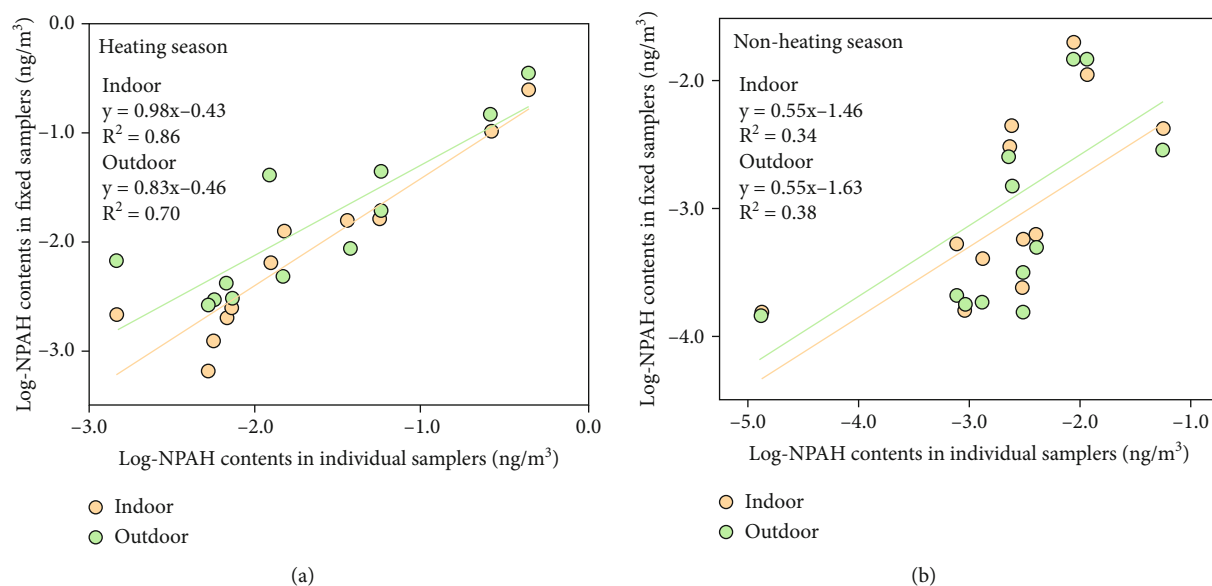


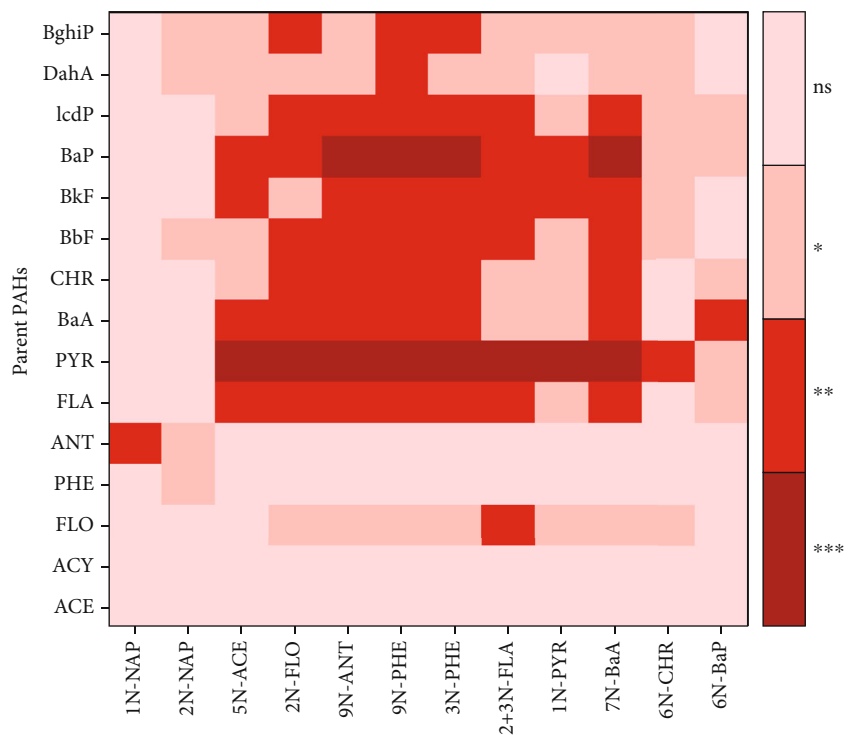
FIGURE 2: Relationship between individual samplers and fixed samplers in (a) heating and (b) nonheating seasons.

combustion sources. It has been proved that 9N-ANT can be produced by biomass burning [36]. 9N-ANT/1N-PYR ratio of more than 10 indicates strong contributions from biomass burning [37]. For 1N-PYR/PYR ratio, it was reported that the ratio of coal stove (0.001) was much smaller than that for diesel exhaust (0.36) [38]. Therefore, the 1N-PYR/PYR ratio can be used as an indicator to determine the potential source of coal stove and vehicle emission. For environment samples (Figure 4), the heating season 9N-ANT/1N-PYR ratios measured by indoor middle-flow sampler were higher than 10 and 1N-PYR/PYR ratios were lower than 0.001. The result indicated a high contribution of coal and biomass combustion in indoor environment during heating season. In contrast, the other 9N-ANT/1N-PYR ratios measured by fixed samplers distributed around 10, which means less contribution of biomass in outdoor environment. The other 1N-PYR/PYR ratios measured by fixed samplers were higher than 0.001 and lower than 0.36, which means a mixture of coal combustion and diesel emission. For individual exposure, the 9N-ANT/1N-PYR ratios of heating season ranged from 2.73 to 12.11, and the ratios decreased rapidly in nonheating season, which means the decrease of biomass contribution. However, the ratios of 1N-PYR/PYR ratios in nonheating season were much higher than those in heating season, which indicated higher diesel emission contribution.

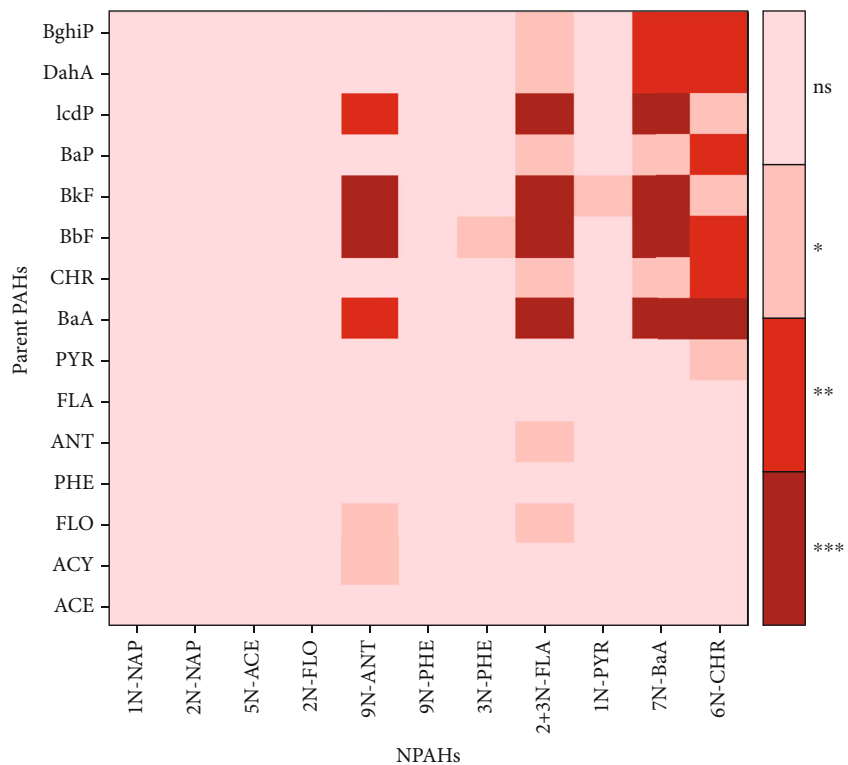
For quantitative investigations of possible sources of PAH contamination, PCA were applied in individual exposure data of both heating and nonheating seasons. As a result, two and three components were extracted in heating and nonheating seasons that represented 78.9 and 88.5 percent of the total variance, respectively. The rotated component matrix is shown in Table 2. PC1 in heating and nonheating seasons accounted for 58.2% and 55.4% of the total variance, respectively. PC1 in both heating and nonheating seasons was heavily weighted by 1N-NAP, 2N-NAP, and 2N-FLO, which were indicators of coal combustion [39]. The emission profile of coal is related to its type.

For the combustion of anthracite, the coal used in the research area Xinzhou, 1N-NAP, was the most abundant species in the emission profile, followed by 2N-NAP and 3N-FLA. In addition, 3N-FLA and 5N-ACE can also be found in the emission of anthracite. This fits the rotated matrix of PC1 in heating seasons well. In the profile of PC1 in nonheating season, 5N-ACE was not detected. This can be attributed to the less consumption of coal in summer. However, coal remains the main fuel for cooking by local residents. PC2 in heating season and PC3 in nonheating season were highly weighted by 1N-PYR, which has been proven to be an indicator of diesel emission [2]. 9N-ANT and 3N-FLA were observed in the combustion of almost all types of wood [36]. Therefore, PC2 in heating season can be judged to present a mixture of biomass combustion and diesel emission. PC3 in nonheating season represents the diesel emission. 9N-ANT and 3N-FLA were highly loaded in PC2 of nonheating season, and it may be a sign of biomass emission in nonheating season. These conclusions of PCA are consistent with the sources identified by specific NPAH ratios. Some evidence suggested that 2+3N-FLA may be generated not directly in combustion but from secondary formation [35]. In this study, we can find that 2+3N-FLA was contained in the profile of coal or biomass combustion. The parent PAHs emitted in the coal combustion, such as FLA, may be a prerequisite for secondary reactions. Therefore, the contribution of secondary reaction may also increase with the increase of coal/biomass combustion. It is difficult to separate the sources of combustion and secondary reactions by PCA.

3.4. Lung Function Levels in Different Seasons. A pulmonary function test was applied in this study to explore the negative effect on children's lung function and obtain the inhalation rates of subjects. The key parameters of lung function of male, female, and total students are shown in Figure 5. The average FVC, FEV1, and PEF of students in Xinzhou were



(a)



(b)

FIGURE 3: Spearman's correlation analysis between parent PAHs and NPAHs during (a) heating season and (b) nonheating season. The heat maps were plotted using the p value or significances (ns: no significances; * $p < 0.05$, ** $p < 0.01$, and *** $p < 0.001$).

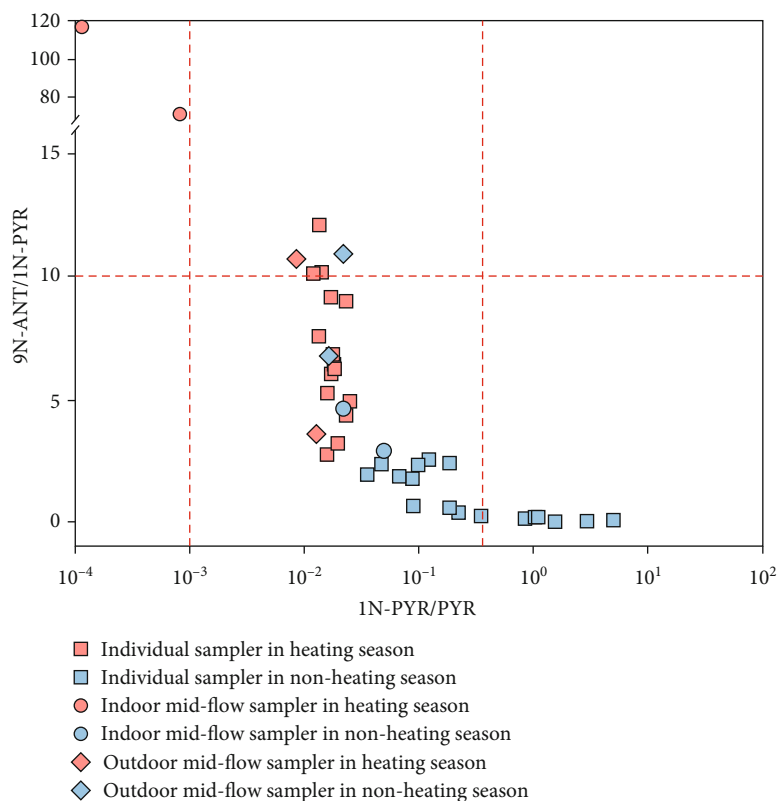


FIGURE 4: Ratios of 1N-PYR/PYR and 9N-ANT/1N-PYR measured by individual portable samplers and fixed middle-flow samplers.

TABLE 2: Rotated component matrix of NPAHs in the particulate phases.

Principal components	Heating season		Nonheating season		
	1	2	1	2	3
1N-NAP	0.145	-0.689	0.951	0.008	0.084
2N-NAP	0.184	-0.758	0.959	0.044	0.153
5N-ACE	0.790	0.431	—	—	—
2N-FLO	0.960	-0.070	0.981	0.075	0.007
9N-ANT	0.522	0.780	0.916	0.309	0.194
9N-PHE	0.953	0.108	0.977	0.072	-0.058
3N-PHE	0.986	0.124	0.235	0.574	-0.383
2+3N-FLA	0.429	0.617	0.251	0.902	0.091
1N-PYR	0.516	0.735	0.197	0.015	0.927
7N-BaA	0.551	0.748	-0.158	0.934	0.040
6N-CHR	0.894	0.109	—	—	—
6N-BaP	0.145	-0.689	—	—	—
Estimated sources	Coal	Diesel and biomass	Coal	Biomass	Diesel
Variance (%)	58.2	20.7	55.4	21.8	11.3

1.7 L, 1.7 L, and 3.8 L·s⁻¹. The total lung function level in this study was similar to the data reported in Taiwan with FVC, FEV1, and PEF values of 1.7 L, 1.6 L, and 3.7 L·s⁻¹, respectively. However, the lung function of children was relatively lower than reported in Hong Kong (FVC was 2.9 L, and FEV1 was 2.6 L) and Mexico abroad (FVC was 2.4 L, and FEV1 was 2.2 L) [40, 41].

Measured lung function parameters were compared with those for healthy children as standard values according to the race and age of children. The severity of chronic obstructive pulmonary disease was classified into several grades according to the ratio between measured function value and predicted value [42]. Ratios ranging from <30%, 30%~<50%, and 50%~<80% were defined as extremely

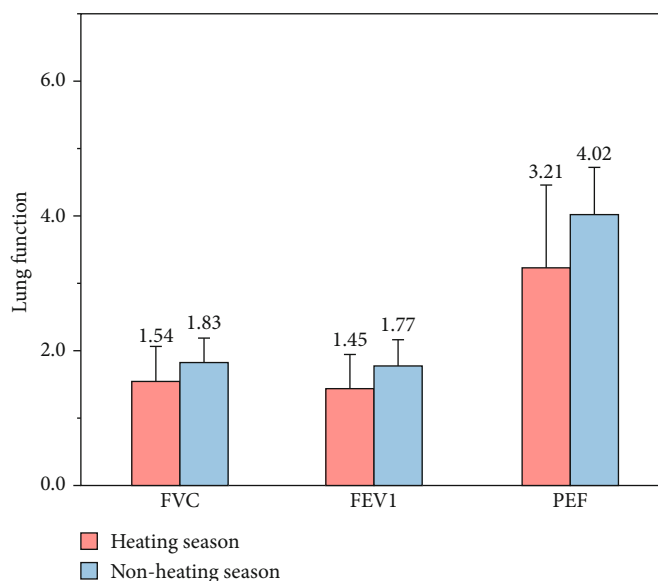


FIGURE 5: Lung function status in heating and nonheating periods. The units for FVC, FEV1, and PEF were L, L, and $L \cdot s^{-1}$, respectively.

TABLE 3: Spearman's correlation analysis results of the lung function and influencing factors.

	Heating season			Nonheating season		
	BW (kg)	$PM_{2.5}$ ($\mu g \cdot m^{-3}$)	NPAH ($ng \cdot m^{-3}$)	BW (kg)	$PM_{2.5}$ ($\mu g \cdot m^{-3}$)	NPAH ($ng \cdot m^{-3}$)
FVC (L)	0.837**	0.624*	0.631*	0.754**	0.369	0.079
FEV1 (L)	0.861**	0.396	0.404	0.756**	0.297	0.072
PEF ($L \cdot s^{-1}$)	0.715**	0.258	0.242	-0.389	0.024	0.008

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

severe, severe, and moderate, respectively. The percentage of children with FVC/FVC predicted lower than 80% was 26.7% and 27.3% during heating and nonheating periods, respectively; the percentage of FEV1/FEV1 lower than 80% was 20.0% and 18.2% during heating and nonheating periods, respectively. Clinical interpretation reveals that FVC can reflect the expiratory period of the larger airway; FEV1 reduction can be associated with increased airway resistance in the airways. The influences of respiratory tract among children can be associated with pollutant exposure in the classroom indoor air.

The relationship between main lung function parameters and three influencing factors including body weight of participants, $PM_{2.5}$ concentration measured by portable sampler, and NPAH exposure level was analyzed by Spearman's correlation (Table 3). As a result, significant positive correlations were found between body weight and most lung function parameters. Taking all participants, it has been proven that indoor coal burning causes an obvious decrease in children's lung function. However, when exploring the relationship between individual lung function and exposure levels, no significant correlation was found either $PM_{2.5}$ or NPAH concentration with lung function parameters. The results can be attributed to two reasons. First, children's lung function parameters are influenced by multiple factors. The effect of personal nutritional level of children (BW) on lung

function may be higher than external conditions such as pollutant exposure. Second, only 40 children were involved in the previous study. The potential relationship may not be detected due to the small sample size.

Significant lung function difference was detected between heating period and nonheating period. FVC, FEV1, and PEF of students during heating period were 1.54 L, 1.45 L, and $3.21 L \cdot s^{-1}$. FVC, FEV1, and PEF of students increased during the nonheating period, which were 1.83 L, 1.77 L, and $4.02 L \cdot s^{-1}$. Comparing the reductions of lung function parameters in heating and nonheating season, it can be estimated that the FVC, FEV1, and DEF had a 15.8%, 18.1%, and 20.1% decrease per $0.80 ng \cdot m^{-3}$ increase in $\sum NPAHs$ in this research. In recent decades, the relationship between pollutants and lung function has been a focus of research. PM is the most concerned pollutant. Previous research has shown a 0.02% decrease in FEV1 per $2.4 mg \cdot m^{-3}$ increase in concentrations of $PM_{2.5}$ [43–46]. Such a decrease in lung function parameters was also noted in studies with personal $PM_{2.5}$ levels [47]. Oxidative stress and inflammation are hypothesized as the main mechanisms through which ambient air pollution can affect lung function [48]. It is believed that the toxicological effect of $PM_{2.5}$ -bound PAHs on the lung through inducing cytotoxicity, oxidative stress, and inflammation has been observed by both in vivo and in vitro studies [25]. Barraza-Villarreal et al. [49] also reported that an

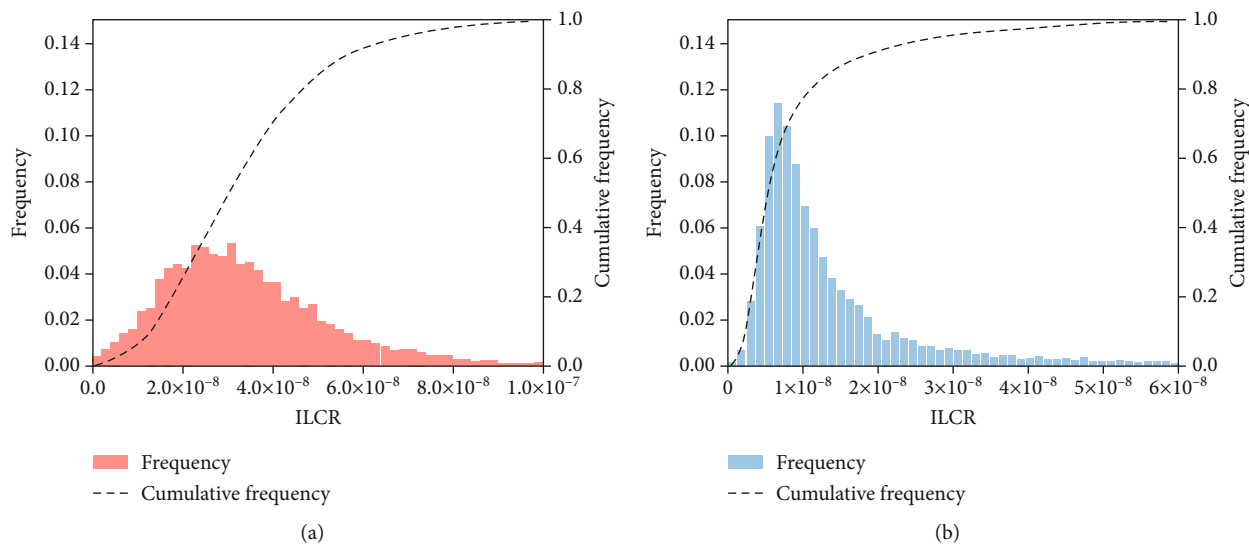


FIGURE 6: Distributions of ILCR for children in (a) heating and (b) nonheating seasons derived using the Monte Carlo simulation.

increase in the concentrations of 2-hydroxyfluorene was significantly negatively associated with FEV1 and FVC. Zhang et al. [50] found that 1 unit increase of 1-hydroxypyrene in urine was negatively associated with 18.8% FVC and 17.3% FEV1. 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 parent PAHs.

3.5. Carcinogenic Risk Assessment. A total of six carcinogenic NPAHs were included in the cancer risk assessment. According to the result of the Shapiro-Wilk test, the IR in two seasons can be described by normal distribution with p values of 0.231 and 0.876, respectively. The Monte Carlo simulations of 10,000 iterations were used to simulate the probability density distributions of carcinogenic risk. The ILCR of the studied population attributing to BaP_{eq} inhalation is illustrated in Figure 6. The mean values of the ILCR for the heating and nonheating seasons were 3.50×10^{-8} (range from 3.04×10^{-11} to 3.14×10^{-7}) and 2.13×10^{-8} (range from 2.20×10^{-10} to 3.59×10^{-6}), respectively, which shows a noticeable higher carcinogenic risk for heating seasons than for nonheating seasons. The effect of heating on exposure is reflected in two aspects. First, indoor coal combustion caused a significant increase in individual exposure. The averaged children's individual BaP_{eq} exposure concentration in heating season was $0.081 \text{ ng}\cdot\text{m}^{-3}$, which is 5.5 times higher than the value of $0.015 \text{ ng}\cdot\text{m}^{-3}$ in nonheating season. Second, the IR of children in heating period was significantly lower than that in nonheating period. In nonheating season, the average IR was $42.08 \text{ m}^3\cdot\text{d}^{-1}$, which was 1.2 times higher than $5.14 \text{ m}^3\cdot\text{d}^{-1}$ in heating season. The slight increase of ILCR is the result of the joint effect of two exposure factors. The ILCR caused by PAH exposure has been studied in our previous research [28]. The ILCR of PAHs for the heating and nonheating seasons were 3.1×10^{-6} and 5.7×10^{-8} ,

respectively. The ILCR of PAHs was two orders of magnitude higher than NPAHs due to their high residual level in the environment. It has been reported by US EPA that ILCR values greater than 10^{-6} indicate a potential risk of carcinogenesis, while the ILCR values greater than 10^{-4} indicate a high risk of cancer. The average total risk due to exposure to NPAHs did not exceed 10^{-6} in two seasons, which indicated a relatively low cancer risk derived by NPAH inhalation.

The sensitivity of parameters is shown in Figure S4. In the heating season, IR was the most sensitive parameter influencing the result (Spearman's rank correlation coefficient = 0.78), followed by the concentration of 6N-CHR and body weight. In the nonheating season, 6N-CHR had the greatest influence on the carcinogenic risk of NPAHs in the nonheating season (0.64), followed by IR and concentration of 1N-PYR.

3.6. Limitations. One limitation of the current study is the small sampling size. The number of participants in this research was relatively small, and the participants in the two seasons were not completely consistent. Only a total of 11 children could take part in both two sampling sessions in heating and nonheating seasons, which may introduce uncertainty in the comparison. Although the sampling size was constrained by limited cost, efforts have been made in participant selection to reduce the uncertainty in the sampling. Questionnaire and household surveys of children were made before sampling, and the participants were chosen according to the results of surveys. Therefore, the participants selected were able to represent the target population. Another limitation is the contribution of the other chemicals associated to particles that were not quantified in this research which requires further analysis.

4. Conclusions

In this study, the individual NPAH exposure levels of children in typical indoor coal combustion were studied. The

results showed that a significantly higher individual NPAH exposure level was found in heating season than in nonheating season. The results of source apportionment indicated that secondary formation is the potential source of NPAHs. Parent PAHs emitted from coal and biomass combustion can be a necessary condition for the secondary formation of NPAHs. The significant influence of increasing NPAH level on the children's function was found in this study. Three lung function indicators including FVC, FEV1, and PEF in heating season were significantly lower than that in nonheating season. The FVC, FEV1, and PEF had a 15.8%, 18.1%, and 20.1% decrease per 0.80 ng·m⁻³ increase in \sum NPAHs. Our research confirmed the association between NPAH exposure and children's lung function decrease. However, the quantitative relationship between NPAH exposure and children's lung function still needs further validation, due to limited sample size. In addition, the determination of metabolites and biomarkers is required to comprehensively evaluate the adverse health effects of children.

Data Availability

The data that support the findings of this study are available from the corresponding author upon reasonable request.

Conflicts of Interest

The authors declare that they have no conflicts of interest.

Authors' Contributions

Beibei Wang was responsible for the conceptualization, methodology, visualization, and investigation and wrote, reviewed, and edited the manuscript. He Huang was responsible for the methodology data curation and investigation. Ning Qin was responsible for the methodology data curation and funding acquisition and wrote the original draft. Wenjing Zhao was responsible for the methodology and data curation. Wang Qin was responsible for the methodology and funding acquisition. Suzhen Cao was responsible for the project administration and supervision. Xing Chen was responsible for the investigation. Xiangyu Xu was responsible for the investigation. Xiaoli Duan was responsible for the project administration, supervision, and funding acquisition.

Acknowledgments

This research was funded by China CDC Key Laboratory of Environment and Population Health, National Institute of Environmental Health, Chinese Center for Disease Control and Prevention (2021-CKL-01), by a special fund of Beijing Key Laboratory of Indoor Air Quality Evaluation and Control (no. BZ0344KF21-06), by the National Natural Science Foundation of China (42177361 and 42007386), and by the National Key Research and Development Program of China (No. 2019YFC1804602).

Supplementary Materials

Table S1: basic information about participants. Table S2: retention time of NPAHs. Table S3: retention time of PAHs. Figure S1: research area in Xinzhou City, Shanxi Province. Figure S2: chromatograms of NPAHs by GC-MS. Figure S3: chromatograms of PAHs by GC-MS. Figure S4: parameter sensitivity for NPAH inhalation incremental lifetime cancer risk assessment relating to children in indoor coal combustion environment. Text S1: pretreatment and instrumental analysis. Text S2: principal component analysis. (*Supplementary Materials*)

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