Polycyclic Aromatic Hydrocarbons in the Indoor and Outdoor Air of Three Cities in the U.S.

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The indoor and outdoor concentrations of 30 polycyclic aromatic hydrocarbons (PAHs) were measured in 55 nonsmoking residences in three urban areas during June 1999—May 2000. The data represent the subset of samples collected within the Relationship of Indoor, Outdoor, and Personal Air study (RIOPA). The study collected samples from homes in Los Angeles, CA, Houston, TX, and Elizabeth, NJ. In the outdoor samples, the total PAH concentrations (ΣPAH) were 4.2–64 ng m⁻³ in Los Angeles, 10–160 ng m⁻³ in Houston, and 12–110 ng m⁻³ in Elizabeth. In the indoor samples, the concentrations of ΣPAH were 16–220 ng m⁻³ in Los Angeles, 21–310 ng m⁻³ in Houston, and 22–350 ng m⁻³ in Elizabeth. The PAH profiles of low molecular weight PAHs (3–4 rings) in the outdoor samples from the three cities were not significantly different. In contrast, the profiles of 5–7-ring PAHs in these three cities were significantly different, which suggested different dominant PAH sources. The signatures of 5–7-ring PAHs in the indoor samples in each city were similar to the outdoor profiles, which suggested that indoor concentrations of 5–7-ring PAHs were dominated by outdoor sources. Indoor-to-outdoor ratios of the PAH concentrations showed that indoor sources had a significant effect on indoor concentrations of 3-ring PAHs and a smaller effect on 4-ring PAHs and that outdoor sources dominated the indoor concentrations of 5–7-ring PAHs.

Introduction

The effect of particulate matter (PM) on indoor air quality has been a subject of extensive research (1, 2). Polycyclic aromatic hydrocarbons (PAHs) comprise only a small fraction of PM; however, they are among the pollutants of concern for human health due to the carcinogenic and mutagenic properties of certain compounds from the PAH class. PAHs arise from a variety of combustion processes and are therefore ubiquitous in the environment. The largest PAH sources to the atmosphere include motor vehicles (3–6), power generation via combustion of coal and oil (3, 7–10), incineration (3, 9), and wood burning (11, 12). In indoor environments, PAHs are generated from cooking (13, 14); smoking (15); burning of natural gas (15, 16); wood (12, 17, 18); candles, and incense (19, 20); and are transported from the outdoors (14, 21).

PAH concentrations in indoor air and their attribution to indoor and outdoor sources have been characterized in a number of studies (10, 14, 20–28). Most studies that investigated the relationship between indoor PAH concentrations and outdoor pollution sources have focused on traffic-related emissions (14, 21, 27, 28). For example, emissions from traffic have been found to be the main outdoor source for the indoor PAH concentrations at urban, semiurban, and suburban locations in the Boston area, MA (14). Few studies examined the indoor/outdoor relationships of PAH concentrations with respect to other types of outdoor sources. Comprehensive assessment of indoor PAH concentrations in urban areas with different climates and their relationship to different types of outdoor emissions sources would significantly contribute to the present understanding of people's exposure to pollutants of indoor and outdoor origin.

This paper presents the results obtained from the subset of samples collected within the Relationship of Indoor, Outdoor, and Personal Air study (RIOPA). The goal of RIOPA was to gain a quantitative understanding of the impact of ambient sources of air pollutants such as VOCs, aldehydes, PM₁₀, and PAHs on indoor air quality and people's exposure by examining the relationships of concentrations of these pollutants in indoor, outdoor, and personal (breathing zone) air. RIOPA included homes in three different climate zones and with a variety of housing characteristics such as house types, air exchangers, household appliances, and activities that can influence indoor air quality. RIOPA was not a population-based study but rather a research effort, which targeted homes located particularly close to sources of air pollution.

Sampling for PAHs within the RIOPA framework was conducted in the indoor and outdoor air; no personal samples were collected for PAHs. The main objective of the PAH component of RIOPA was to characterize residential exposure to PAHs of outdoor origin through the relationship between indoor and outdoor PAH concentrations. The specific objectives of this paper are (i) to assess the indoor and outdoor PAH concentrations in three geographically distinct urban areas characterized by different climate and types of dominant emission sources, (ii) to establish the relationship between the indoor and outdoor PAH concentrations, and (iii) to examine residential exposure to outdoor PAHs.
Methods

**Site Characterization and Sampling Strategy.** Los Angeles County, CA, Houston, TX, and Elizabeth, NJ, were selected for the study to represent urban areas with different climates and different types of emission sources. Los Angeles County, CA (further in the text, Los Angeles), is a metropolis interwoven by many major transportation arteries as well as thousands of local roads that together have a large amount of automobile traffic. Motor vehicles have been found to contribute a significant fraction to the atmospheric concentrations of particulate PAHs within the Los Angeles basin. Samples presented were collected in two different areas located about 50 km apart: West Los Angeles and Santa Clarita, which is located to the northwest of the city of Los Angeles. In West Los Angeles, the target homes were those located near the intersection of the U.S. Interstate routes I-10 and I-405, and in Santa Clarita, homes were selected in the vicinity of the U.S. Interstate route I-5.

Houston, TX, has a highly developed petrochemical industry, which creates a large area source of petrogenic as well as pyrogenic PAHs in the region. The majority of petrochemical plants and oil refineries are located northeast and east of the center of Houston, where the Houston Ship Channel provides access to the Gulf of Mexico. Samples presented were collected in three residential areas located in the industrial zone: Baytown, Pasadena, and Channelview.

Elizabeth, NJ, is located in central New Jersey, west of New York City. The highly developed industrial/commercial/transportation infrastructure of Elizabeth includes a marine port, numerous shipping and loading sites, an oil refinery, the Newark International Airport, and several major roads (the U.S. Federal route 1&9 and the U.S. Interstate route I-95) with a considerable amount of diesel truck traffic. Apart from these sources, in New Jersey, as in other regions with four distinct seasons, home heating with natural gas and oil has been found to contribute to the atmospheric PAH concentrations during the cold months (31). Therefore, in terms of PAH emissions, Elizabeth can be described as a combination of stationary and mobile sources. Sampling was carried out in the areas of high density of commercial traffic: Trumbull, Bayway, and along Route 1&9.

The study was targeted at the worst-case locations in terms of outdoor sources of air pollution. Homes were selected based on their proximity to identified sources, and in those areas, participants were recruited on a volunteer basis. Because smoking had been shown to be the dominant indoor source for PAHs (35), only nonsmokers’ homes were considered for sampling.

Integrating indoor and outdoor air samples of 48h duration were collected simultaneously. Inside the homes, sampling equipment was placed in the main living area of the house at a height of about 1.5 m and at least 1 m from the walls. The outdoor sampler was placed at ~1.5 m height, at least 1 m from the wall of the house, and away from exhaust ducts and light and heat sources. PAH analysis was done for 55 homes (19 homes in Los Angeles, 21 homes in Houston, and 15 homes in Elizabeth) sampled during the period from June 1999 to May 2000.

**Air Sampling.** Samples were obtained using modified MSP samplers. Original MSP samplers (MSP Corporation, Minneapolis, MN) designed for collection of fine particulate matter, PM2.5, were modified by addition of stainless steel cylinder with an enclosed polyurethane foam cartridge (PUF) (diameter 25 mm, height 100 mm) immediately following the filter compartment. Particulate-phase PAHs were collected on 37 mm quartz fiber filters (QFFs), and gas-phase PAHs were retained on the PUFs. Samplers were operated at a flow rate of 10L min⁻¹, leading to average sample volume of 29 m³. Flow rates were measured at the beginning and at the end of each sampling event. Prior to sampling, PUF cartridges were hand-washed in tap water containing Alconox detergent, rinsed in deionized water followed by acetone, then sequentially extracted in Soxhlet apparatus with acetone (24 h) and petroleum ether (24 h), and dried in a vacuum desiccator for 48 h at ambient temperature. Cleaned PUF cartridges were stored. The outdoor sampler was placed at a height of about 1.5 m and at least 1 m from the walls. Particulate-phase PAHs were collected on 37 mm quartz fiber filters (QFFs), and gas-phase PAHs were retained on the PUFs.

To determine the analytical recoveries of the PAHs, the PUFs were extracted statically in glass columns (i.d. 30 mm × 120 mm, with 2 mm Teflon stopcock) for 1 h with 40 mL of the mixture of hot (50°C) hexane and dichloromethane (1:1) (50 mL/sample volume). The extracts were dried to constant weight using PUF samples were spiked with a surrogate standard consisting ofacenaphthene-d₅₀, anthracene-d₅₀, fluoranthene-d₅₀, and benzo[a]pyrene-d₃₂ to determine the analytical recoveries of the PAHs. The PUFs were extracted statically in glass columns (i.d. 30 mm × 120 mm, with 2 mm Teflon stopcock) for 1 h with 40 mL of the mixture of hot (50°C) hexane and dichloromethane (1:1) (50 mL/sample volume). The extracts were dried to constant weight using PUF cartridge was loaded into a deactivated column with 5% deionized H₂O. The column was rinsed with 20 mL of the hot hexane/DCM mixture; rinses were combined with the extracts. Each QFF sample was split in two portions. A 2 cm² portion of each filter was analyzed for organic and elemental carbon content using a thermal/optical carbon analyzer (Sunset Laboratory Inc., version 4.0). The remaining portion of the QFF was spiked with the surrogate standard and extracted twice for 35 min with 25 mL of dichloromethane (DCM) under ultrasonic agitation.

The PUF and QFF extracts were concentrated by rotary evaporation (Buchi RE 111), followed by further concentration under a gentle nitrogen stream, and cleaned on microcolumns (i.d. 5 mm × 100 mm) of silica gel to remove interfering polar compounds. Silica gel (60–200 mesh) was baked at 400°C for 8 h, cooled in a desiccator for 1 h, and deactivated with 5% deionized H₂O. The column was rinsed with 2 mL of hexane/DCM (9:1 by volume). The samples were added to the column and eluted with 8 mL of 9.1 hexane/DCM. Collected samples were reduced to ~0.05 mL by evaporation under a gentle nitrogen stream. An internal standard solution consisting of naphthalene-d₈, phenanthrene-d₁₀, pyrene-d₁₀, and benzo[a]pyrene-d₃₂ was added to concentrated samples; the QFF extracts were concentrated further to ~0.01 mL.

The samples were analyzed on a Hewlett-Packard 6890 gas chromatograph equipped with HP 5973 mass selective detector operated in selected ion monitoring mode. Separation of the compounds was carried out on a high-resolution capillary column (J&W Scientific; i.d. = 0.25 mm, L = 30 m,) with DB-5 as the stationary phase (film thickness 0.25 μm). Helium (chromatographic grade) was used as the carrier gas at a flow rate of 1.2 mL min⁻¹. The pressure in the column was maintained at 9.86 psi. The inlet was operated in the pulsed splitless mode at 300°C; the injection volume was 1 μL. The temperature program was as follows: the initial temperature (50°C) was held for 1 min, after which the temperature was raised using three subsequent temperature ramps (first at 25°C min⁻¹ to 125°C, then 8°C min⁻¹ to 260°C, and finally 3.5°C min⁻¹ to 300°C); the final temperature was held for 10 min. The analysis time for one sample was about 43 min.

**Quality Control.** Several methods were used to ensure the quality of the data. To guarantee unbiased analysis, all samples including field blanks were coded by five-digit numbers, such that no information about sampling date, location of homes, or category (indoor/outdoor/field blank) was known until the results of analysis were validated.
Field blanks were collected in nearly all homes to determine potential contamination of samples during sampling, transport, and storage; clean PUFs and QFFs were brought to the homes and placed unopened next to the samplers (indoors or outdoors) for the duration of sampling, after which they were returned to the laboratory and treated as regular samples. ∑PAH mass measured in the PUF field blanks ranged from 11 to 190 ng, whereas in the PUF samples it ranged from 210 to 18 000 ng. In the QFF field blanks, ∑PAH mass ranged from 0.74 to 9.1 ng, while in the QFF samples it ranged from 2.9 to 750 ng. The method detection limits (MDL) for individual PAHs were defined as 3 times the standard deviation of the mean PAH mass in the field blanks. Because the mean PAH masses in the field blanks collected in different cities were similar and because no significant difference was found between the field blanks collected indoors and outdoors, media-specific MDLs were calculated using all field blanks and were applied to samples collected in each of the three cities. The MDL values for individual PAHs are shown in Table 1 of the Supporting Information.

Laboratory blanks (clean PUFs and QFFs) and reference standards were extracted and analyzed with every 14 samples. The mass of ∑PAHs in the laboratory blanks ranged from < MDL to 8.9 ng for the PUFs and from < MDL to 4.8 ng for the QFFs and accounted for less than 5% of the corresponding PAH mass in the field blanks. Because the mean PAH masses in the field blanks collected in different cities were similar and because no significant difference was found between the field blanks collected indoors and outdoors, media-specific MDLs were calculated using all field blanks and were applied to samples collected in each of the three cities. The MDL values for individual PAHs are shown in Table 1 of the Supporting Information.

Results and Discussion

Occurrence of PAHs in Urban Atmospheric Environments. The outdoor and indoor PAH concentrations defined as the sum of the concentrations of 30 individual PAHs, ∑PAH, in the gas and particulate phases are presented in Figure 1. The geometric mean concentrations (ranges) of individual PAHs in the outdoor and indoor samples are shown in Table 2 of the Supporting Information. The ∑PAH concentrations in the outdoor samples ranged from 4.2 to 64 ng m⁻³ in Los Angeles, from 10 to 160 ng m⁻³ in Houston, and from 12 to 110 ng m⁻³ in Elizabeth. In the indoor samples, the variability in the PAH concentrations was substantially larger than in the outdoor samples. The indoor ∑PAH concentrations ranged from 16 to 220 ng m⁻³ in Los Angeles, from 21 to 310
ng m$^{-3}$ in Houston, and from 22 to 350 ng m$^{-3}$ in Elizabeth. Variations in the PAH concentrations in the outdoor and indoor samples were driven by gaseous compounds, which comprised, on average, 90–97% of the total PAH mass measured in the samples.

The outdoor $\Sigma$PAH concentrations in three cities were significantly different. The gas-phase $\Sigma$PAH concentrations were lowest in Los Angeles samples, whereas the lowest particulate-phase $\Sigma$PAH concentrations were in Houston samples. These differences could be due to different dominant emission sources of PAHs in three cities and temperature-driven differences in gas/particle distribution. Distinct PAH patterns in the outdoor samples collected in Los Angeles, Houston, and Elizabeth required that the indoor and outdoor concentrations of PAHs be analyzed by city.

The relative contribution of individual PAHs to the total (gas + particulate) PAH mass is shown in Figure 2. The largest contribution to the $\Sigma$PAH mass in the outdoor and indoor air in each city was made by phenanthrene, followed by the sum of four methylated derivatives of phenanthrene and anthracene. The PAH profiles of low molecular weight PAHs with 3–4 rings (from dibenzo[a]anthracene to benzo[b]naphtho[2,1-d]thiophene, MW = 184–234) in three cities were not significantly different. In contrast, significant differences existed for high molecular weight PAHs with 5–7 rings (from benzo[b+k]fluoranthene to coronene, MW = 252–300). The percent of 5–7-ring PAHs in the $\Sigma$PAH mass was different in each city, with the lowest percent in Houston, as seen in Figure 2. High molecular weight PAHs in the outdoor air were dominated by benzo[g,h,i]perylene and coronene in Los Angeles and benzo[b+k]fluoranthene in Houston; in Elizabeth, contributions of benzo[b+k]fluoranthene, indeno[1,2,3-cd]pyrene, benzo[g,h,i]perylene, and coronene were approximately equal. Significantly different profiles of 5–7-ring PAHs in the outdoor air suggested different dominant PAH sources in three cities. The signatures of 5–7-ring PAHs in the indoor air in each city were similar to the outdoor profiles, which suggested that indoor concentrations of 5–7-ring PAHs were dominated by indoor sources.

The outdoor PAH concentrations measured in this study are compared with the ambient PAH concentrations reported for the same geographic areas (Table 1). Good agreement exists between the PAH concentrations measured in Houston and Seabrook, TX (30), which is located about 40 km southeast from Houston. The PAH concentrations measured in Elizabeth are comparable with the PAH concentrations measured in Jersey City, NJ (32), which is located about 10 km northeast from Elizabeth. The PAH concentrations measured in Los Angeles during this study are consistent with those measured in 1993 (33), with the exception of phenanthrene for which concentrations were lower during RIOPA.

The coupled indoor and outdoor concentrations of gaseous PAHs measured in this study were comparable with those in Columbus, OH (15), and Taipei, Taiwan (20), and were considerably lower than in Hangzhou, China (34), as seen in Table 1. The indoor and outdoor concentrations of particulate PAHs were similar to PAH concentrations in Huddersfield, England (27), and were, on average, lower than those in Columbus, OH (15), Amsterdam, The Netherlands (21), Pavia, Italy (28), and Taipei, Taiwan (20).

**Relationship between the Indoor and Outdoor PAH Concentrations.** The I/O ratios of PAH concentrations (gas + particulate) measured in this study are presented in Figure 3. The reference line in Figure 3 shows the I/O ratio equal to 1. The I/O ratios greater than 1 indicate that indoor sources make a significant contribution to indoor air concentrations, whereas in the absence of strong indoor sources, the I/O ratios are expected to be close to or lower than 1. The indoor/outdoor relationships for individual PAHs were quite different. In general, the I/O ratios were higher for low molecular...
<table>
<thead>
<tr>
<th>Site Location</th>
<th>Time Frame</th>
<th>n</th>
<th>Phenanthrene Outdoor (ng m$^{-3}$)</th>
<th>Pyrene Outdoor (ng m$^{-3}$)</th>
<th>Benzo[a]anthracene Outdoor (ng m$^{-3}$)</th>
<th>Benzo[a]pyrene Outdoor (ng m$^{-3}$)</th>
<th>Indoor Concentration (ng m$^{-3}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Los Angeles, CA</td>
<td>July 1999 - May 2000</td>
<td>19</td>
<td>8.1 (0.92 - 25)</td>
<td>1.6 (0.12 - 5)</td>
<td>0.20 (0.0072 - 1.8)</td>
<td>0.056 (0.00080 - 1.0)</td>
<td>0.655 (0.00080 - 10)</td>
</tr>
<tr>
<td>Houston, TX</td>
<td>June 1999 - May 2000</td>
<td>21</td>
<td>19 (5.4 - 97)</td>
<td>2.3 (0.87 - 15)</td>
<td>0.15 (0.028 - 13)</td>
<td>0.25 (0.0021 - 0.22)</td>
<td>0.29 (0.00080 - 10)</td>
</tr>
<tr>
<td>Elizabeth, NJ</td>
<td>June 1999 - May 2000</td>
<td>15</td>
<td>26 (6.5 - 60)</td>
<td>3.6 (0.85 - 10)</td>
<td>0.41 (0.11 - 1)</td>
<td>0.14 (0.045 - 0.53)</td>
<td>0.52 (0.00080 - 10)</td>
</tr>
<tr>
<td>Los Angeles, CA</td>
<td>Sept 1993</td>
<td>12</td>
<td>0 (0.0 - 50)</td>
<td>3.3 (0.084 - 19)</td>
<td>0.14 (0.015 - 0.63)</td>
<td>0.50 (0.0006 - 0.30)</td>
<td>0.058 (0.0007 - 0.30)</td>
</tr>
<tr>
<td>Jersey City, NJ</td>
<td>Feb 1995 - Aug 1996</td>
<td>58</td>
<td>5 (3.4 - 140)</td>
<td>2.1 (0.16 - 4.3)</td>
<td>0.55 (0.0052 - 3.1)</td>
<td>0.19 (0.0017 - 1.3)</td>
<td>0.37 (0.0052 - 2.1)</td>
</tr>
<tr>
<td>Philadelphia, PA</td>
<td>May 1998 - Dec 1999</td>
<td>8</td>
<td>8.1 (0.92 - 25)</td>
<td>19 (5.4 - 97)</td>
<td>2.6 (0.87 - 15)</td>
<td>0.15 (0.028 - 13)</td>
<td>0.25 (0.0021 - 0.22)</td>
</tr>
<tr>
<td>Columbus, OH</td>
<td>Winter 1986/1987</td>
<td>8</td>
<td>2.3 (0.92 - 25)</td>
<td>3.6 (0.85 - 10)</td>
<td>0.41 (0.11 - 1)</td>
<td>0.14 (0.045 - 0.53)</td>
<td>0.52 (0.00080 - 10)</td>
</tr>
<tr>
<td>Huddersfield, England</td>
<td>Summer and Spring 1995</td>
<td>13</td>
<td>NA</td>
<td>0.32 (0.015 - 0.63)</td>
<td>0.55 (0.0006 - 0.30)</td>
<td>0.37 (0.0052 - 2.1)</td>
<td>0.68 (0.0007 - 0.30)</td>
</tr>
<tr>
<td>Amsterdam, The Netherlands</td>
<td>Summer and Spring 1995</td>
<td>18</td>
<td>NA</td>
<td>1.9 (0.16 - 4.3)</td>
<td>0.55 (0.0006 - 0.30)</td>
<td>0.37 (0.0052 - 2.1)</td>
<td>0.68 (0.0007 - 0.30)</td>
</tr>
<tr>
<td>Pavia, Italy</td>
<td>Feb and June 1996</td>
<td>8</td>
<td>2.1 (0.16 - 4.3)</td>
<td>0.55 (0.0052 - 3.1)</td>
<td>0.19 (0.0017 - 1.3)</td>
<td>0.37 (0.0052 - 2.1)</td>
<td>0.68 (0.0007 - 0.30)</td>
</tr>
<tr>
<td>Taipei, Taiwan</td>
<td>Summer 1995 and Winter 1995/1996</td>
<td>14</td>
<td>21 ± 4.9</td>
<td>1.7 ± 0.2</td>
<td>0.055 (0.0005 - 0.23)</td>
<td>0.550 (0.0005 - 0.23)</td>
<td>0.550 (0.0005 - 0.23)</td>
</tr>
<tr>
<td>Hangzhou, China</td>
<td>Summer and Fall 1999</td>
<td>8</td>
<td>660 ± 600</td>
<td>NA</td>
<td>20 ± 4.6</td>
<td>1.7 ± 0.2</td>
<td>1.7 ± 0.2</td>
</tr>
</tbody>
</table>

* Geometric mean (range).
* Only concentration of benzo[a]fluoranthene reported.
* Arithmetic mean (range).
* Concentration range for the particle phase only.
* Arithmetic mean ± standard deviation.
* Geometric mean ± standard deviation.
* NA, data not available.
* MDL, data below detection limit.

**Notes:**
- Where C and C$^{\circ}$ are the measured concentrations and geometric means of all homes, respectively, and the intercepts and slopes are the intercept and slope of linear regression. The median I/O ratios for the sum of these PAHs were greater than 1, which suggested that indoor concentrations were lower than those of the outdoors. As an example, the I/O ratio for fluoranthene was equal to or smaller than 1 in 36 of 55 homes. The median I/O ratios of 15 homes were not different from zero at the 95% confidence level.
- Different behaviors of low and high molecular weight PAHs were observed for PAHs with 4 rings (fluoranthene, pyrene, benzo[a]pyrene, benzo[b]fluoranthene, dibenzothiophene, andacenaphtene) and those with 5 rings (benzo[k]fluoranthene, benzo[b]fluoranthene, benzo(i)fluoranthene, benzo[k]fluoranthene).
- High molecular weight PAHs in the indoor air were associated primarily with the respirable particulate matter, whereas low molecular weight PAHs were found in gas phase. Indoor concentrations of these PAHs were controlled by outdoor sources.
- Concentrations of these PAHs were lower than those for the outdoors. As an example, the I/O ratio for fluoranthene was equal to or smaller than 1 in 36 of 55 homes. The median I/O ratios of 15 homes were not different from zero at the 95% confidence level.
- Significant ($p < 0.05$) correlation between indoor and outdoor concentrations of these PAHs was found for 51 of 55 homes. The coefficient of determination ($R^2$) was 0.59 for 51 of 55 homes. The intercepts, b, and the slopes, m, were not different from zero at the 95% confidence level. The intercepts and slopes were not different from zero at the 95% confidence level.
- Different behaviors of low and high molecular weight PAHs were observed for PAHs with 4 rings (fluoranthene, pyrene, benzo[a]pyrene, benzo[b]fluoranthene, dibenzothiophene, andacenaphtene) and those with 5 rings (benzo[k]fluoranthene, benzo[b]fluoranthene, benzo(i)fluoranthene, benzo[k]fluoranthene).
- High molecular weight PAHs in the indoor air were associated primarily with the respirable particulate matter, whereas low molecular weight PAHs were found in gas phase. Indoor concentrations of these PAHs were controlled by outdoor sources.
Figure 4. Substantial scattering observed for 3–4-ring PAHs in each city suggests a complex source/sink relationship of these compounds in the indoor air (Figure 4a). Dispersion of the data points above 1:1 line illustrates the importance of indoor sources for low molecular weight PAHs. In contrast, good correlation between the indoor and outdoor concentrations of 5–7-ring PAHs (Figure 4b), with the majority of data points below 1:1 line, indicates the dominance of outdoor sources. Strength of the indoor/outdoor relationships for PAHs in each city is partly explained by the air exchange rates of the homes (Figure 5). Larger scattering of the data was observed in Houston where the air exchange rates were lower (median AER = 0.46 h⁻¹) than in Los Angeles (median AER = 0.82 h⁻¹) and Elizabeth (median AER = 0.90 h⁻¹). Detailed examination of indoor/outdoor concentrations of PAHs with relationship to air exchange rates, sources, and sinks of PAHs will be done in a subsequent publication.

Table 2. I/O Concentration Ratios for the Sum of the High Molecular Weight PAHs (MW = 252–300) and the Slopes of the Linear Regression of the Indoor and Outdoor Concentrations (n = 8) of These PAHs

<table>
<thead>
<tr>
<th>Los Angeles, CA</th>
<th>Houston, TX</th>
<th>Elizabeth, NJ</th>
</tr>
</thead>
<tbody>
<tr>
<td>home ID</td>
<td>I/O ratio</td>
<td>slope</td>
</tr>
<tr>
<td>CA011</td>
<td>0.54</td>
<td>0.56</td>
</tr>
<tr>
<td>CA013</td>
<td>0.92</td>
<td>0.76</td>
</tr>
<tr>
<td>CA015</td>
<td>1.0</td>
<td>1.1</td>
</tr>
<tr>
<td>CA017</td>
<td>1.7</td>
<td>1.9</td>
</tr>
<tr>
<td>CA018</td>
<td>7.3</td>
<td>7.1</td>
</tr>
<tr>
<td>CA019</td>
<td>0.84</td>
<td>0.95</td>
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<tr>
<td>CA021</td>
<td>0.70</td>
<td>0.74</td>
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<td>0.73</td>
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<tr>
<td>CA040</td>
<td>0.98</td>
<td>0.78</td>
</tr>
<tr>
<td>CA041</td>
<td>1.0</td>
<td>1.0</td>
</tr>
<tr>
<td>CA051</td>
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<td>0.63</td>
</tr>
<tr>
<td></td>
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<td></td>
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<tr>
<td></td>
<td></td>
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</tr>
<tr>
<td>median</td>
<td>0.88</td>
<td>0.80</td>
</tr>
</tbody>
</table>

* Slope is not significant (p > 0.05). a Regression was not done because the outdoor concentrations of six PAHs were below method detection limit.
indoor concentrations of benzo[g,h,i]perylene were 68 ± 2.5% (R² = 0.95) of the outdoor concentrations. For comparison, the equilibrium concentration of PM_{2.5} in indoor air calculated for a home with no strong indoor sources and typical air exchange rate of 0.75 h⁻¹, assuming the indoor particle deposition rate constant of 0.4 h⁻¹, is about 65% of the outdoor concentration (35). In 13 of 15 homes in Elizabeth and 16 of 21 homes in Houston, the indoor concentrations of benzo[g,h,i]perylene were 48 ± 4.4% (R² = 0.78) and 33 ± 4.8% (R² = 0.48) of the outdoor concentrations, respectively; however, scattering of the data in Houston creates less confidence in such estimation.

Acknowledgments

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FIGURE 4. Correlations between the indoor and outdoor concentrations of (a) phenanthrene and (b) benzo[g,h,i]perylene. Empty circles show homes with significant indoor sources, and black circles show homes without significant indoor circles. Dashed lines show 1:1 relationship.

FIGURE 5. Air exchange rates of the homes, h⁻¹. The values greater than 3 were set to equal 3.
Supporting Information Available

Information about analysis, quality control/quality assurance of the method, and the mean (range) concentrations of 30 PAHs (gas and particulate) in the indoor and outdoor air in Los Angeles, Houston, and Elizabeth homes. This material is available free of charge via the Internet at http://pubs.acs.org.

Supporting Information Available for help with the PAH analysis.

Literature Cited


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