

Predictors of Personal Polycyclic Aromatic Hydrocarbon Exposures among Pregnant Minority Women in New York City

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As part of a multiyear birth-cohort study examining the roles of pre- and postnatal environmental exposures on developmental deficits and asthma among children, we measured personal exposures to polycyclic aromatic hydrocarbons (PAHs) among 348 pregnant women in northern Manhattan and the South Bronx, New York. Nonsmoking African-American or Dominican women were identified and recruited into the study. During the third trimester of pregnancy, each subject wore a personal air monitor for 48 hr to determine exposure levels to nine PAH compounds. In this study, we examined levels of exposures to PAHs and tested for associations with potential predictor variables collected from questionnaires addressing socioeconomic factors and day-to-day activities during pregnancy as well as activities and environmental exposures during the 48-hr monitoring period. Reliable personal monitoring data for women who did not smoke during the monitoring period were available for 344 of 348 subjects. Mean PAH concentrations ranged from 0.06 ng/m³ for dibenz[*a,h*]anthracene to 4.1 ng/m³ for pyrene; mean benzo[*a*]pyrene concentration was 0.50 ng/m³. As found in previous studies, concentrations of most PAHs were higher in winter than in summer. Multiple linear regression analysis revealed associations between personal PAH exposures and several questionnaire variables, including time spent outdoors, residential heating, and indoor burning of incense. This is the largest study to date characterizing personal exposures to PAHs, a ubiquitous class of carcinogenic air contaminants in urban environments, and is unique in its focus on pregnant minority women. **Key words:** minority, personal exposure, polycyclic aromatic hydrocarbons (PAHs), traffic, urban, women. *Environ Health Perspect* 112:754–759 (2004). doi:10.1289/ehp.5955 available via <http://dx.doi.org/> [Online 29 January 2004]

Polycyclic aromatic hydrocarbons (PAHs) are a family of organic compounds produced during the incomplete combustion and pyrolysis of organic material such as coal, wood, fuel, garbage, tobacco, and meat. A major source of ambient PAHs in the United States is believed to be motor vehicle emissions, particularly in urban areas. Motor vehicle emissions contribute 46–90% of the mass of individual PAHs in ambient airborne particles in urban areas in the United States and Europe (Dunbar et al. 2001; Harrison et al. 1996; Nielsen 1996). Vehicles reported to have the highest PAH emission rates are diesel engines and gasoline engines without catalytic converters (Harrison et al. 1996; Rogge et al. 1993). In urban environments, industrial operations, waste incinerators, and residential boilers are also important sources of ambient PAHs [Agency for Toxic Substances and Disease Registry (ATSDR) 1995]. In addition to these local sources, regional effects such as long-range transport and the volatilization of previously deposited PAHs from surfaces can be important factors in urban ambient PAH levels (Dimashki et al. 2001).

Many PAHs are known mutagens or animal carcinogens and are therefore ranked as probable human carcinogens (category B2) by the U.S. Environmental Protection Agency (U.S. EPA) Integrated Risk Information System (IRIS 1997). B2 PAHs are present in the air predominantly in particulate form,

associated with respirable particles of < 2.5 μm aerodynamic diameter (PM_{2.5}; Lioy and Greenberg 1990). PAHs emitted by mobile sources have been shown to be more mutagenic than those from other sources such as wood stoves (Cuppitt et al. 1994; Lewtas et al. 1992). Because most particles generated by combustion have aerodynamic diameters < 2.5 μm, they tend to deposit in the pulmonary region, potentially delivering carcinogenic compounds directly to the lung interstitium (Venkataraman and Raymond 1998). This is consistent with findings from epidemiologic investigations that relate mortality associated with air particles to cardiopulmonary disease and lung cancer (Dockery et al. 1993; Pope et al. 2002). Several epidemiologic investigations have shown increased mortality from lung cancer in humans exposed to coke oven emissions, roofing-tar emissions, and cigarette smoke, each of which is a mixture containing many carcinogenic PAHs, and decreased lung function has been observed among rubber factory workers exposed to particle-bound PAHs [ATSDR 1995; Gupta et al. 1994; International Agency for Research on Cancer (IARC) 1984]. In addition to carcinogenic effects, experimental evidence indicates that benzo[*a*]pyrene (BaP) and other PAHs may induce reproductive and immunologic disorders in rodents. (ATSDR 1995).

Northern Manhattan and the South Bronx, New York, are home to primarily

African-American and Hispanic communities that are also relatively economically disadvantaged. There are reasons to believe that economically disadvantaged groups may be at greater risk of health impacts from environmental contaminants, because of either enhanced exposures or greater susceptibility compared with more advantaged groups. Previous studies have suggested that exposures to environmental pollutants such as environmental tobacco smoke (ETS), lead, and industrial pollution are in some cases more prevalent among minorities and low-income individuals (Chuang et al. 1999; Knight et al. 1996; Sheppard et al. 1999; Talbot et al. 1998; Wagenknecht et al. 1993). Economic disparities in a variety of health outcomes have been widely documented (Kaplan and Keil 1993; Townsend et al. 1988), and these may influence underlying susceptibility to environmental insults.

As part of a multiyear birth-cohort study examining the roles of environmental factors in adverse birth outcomes, developmental deficits, and asthma among children, we measured personal exposures to PAHs among pregnant women in northern Manhattan and the South Bronx. Because individual activities are expected to strongly influence personal exposure to PAHs, personal air monitoring is especially relevant for assessing health risks associated with PAH exposures. Personal monitoring integrates exposure across multiple times and locations through which an individual passes in the course of daily activities, and thus has the potential to better characterize human exposure than is possible using stationary ambient or indoor monitors. To date, no data have been reported on the personal exposure to PAHs of minority residences in U.S. cities, a population subgroup that may be both

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Grant support was provided by the National Institute of Environmental Health Sciences (NIEHS)/U.S. Environmental Protection Agency (EPA) P50 ES09600, NIEHS R01 ES10165, and R01 ES08977, the U.S. EPA (NCER STAR Program), the W. Alton Jones, Gladys and Roland Harriman Foundation, Bauman Foundations, and the New York Community Trust.

The authors declare they have no competing financial interests.

Received 27 August 2002; accepted 27 January 2004.

disproportionately exposed and vulnerable to PAHs. In this article, we report airborne PAH personal exposures in a cohort of 344 pregnant women in New York City and evaluate the extent to which a variety of questionnaire-based variables can predict levels of exposure.

Materials and Methods

The subjects included in this analysis are part of an ongoing prospective cohort study of African-American and Dominican women and their newborns conducted by the Columbia Center for Children's Environmental Health (CCCEH). The study began in 1997 to evaluate the effects of prenatal exposures to ambient air pollutants including PAHs and ETS on birth outcomes and neurocognitive development in members of these major minority communities in New York City. The study design does not include a low-exposure "control" cohort from outside the urban area, but rather seeks to characterize between-subject variability in exposures within the study area. Initial results from the analysis of these women's personal exposure to PAHs are presented here and are being used to support epidemiologic investigations conducted by the CCCEH.

Subjects were recruited through the prenatal clinics at New York Presbyterian and Harlem Hospitals and were told that the study was to gain information on the effects of environmental exposures during pregnancy on asthma and on child development. These hospitals generally serve residents of the Washington Heights, Inwood, Harlem, and South Bronx areas of New York City. Recruitment was restricted to women 18–35 years of age who self-identified as African American or Dominican and who were residents of the Central Harlem or Washington Heights/Inwood areas of northern Manhattan or the South Bronx for at least 1 year before pregnancy.

Women were excluded from the study if they smoked cigarettes or used other tobacco products or illicit drugs during pregnancy, if they had diabetes or hypertension or were known to be HIV positive, or if their first prenatal visit occurred after the 20th week of pregnancy, to control for known risk factors of adverse birth outcomes. Of the women screened, 870 of 1,706 (47%) met the eligibility criteria; 70% agreed to participate. Women were considered fully enrolled in the study upon completion of prenatal personal monitoring and questionnaires and if blood samples were collected from the mother and/or newborn at delivery. All subjects provided informed consent, and the Columbia Presbyterian Medical Center Institutional Review Board approved the study.

Two prenatal questionnaires were administered to subjects. A prenatal questionnaire was

administered to collect data on race/ethnicity, socioeconomic status (SES), medical history, lifetime residential history, and history of active and passive smoking as well as occupational history, workplace exposures, and environmental exposures during pregnancy. To explore factors that may be predictive of PAH exposures, in the present study we focused on variables related to demographics (age and race/ethnicity), SES (highest degree earned and household income), and home characteristics (traffic levels near the home, proximity to bus depot, and type of heating system). This questionnaire was administered to each woman in her home by a trained research worker during the third trimester of pregnancy.

To gain additional information on potential sources of PAH exposure encountered during the personal monitoring period, we began administering a second questionnaire in May 1998 at the completion of the 48-hr personal air monitoring; these data were available for 336 of 348 women. Variables from this questionnaire that were included in the present study included exposures to coal products and combustion by-products, active and passive smoking, type and frequency of cooking in the home, home heating, and use of candles and incense during the previous 48 hr. Also, information on the number of hours spent outdoors and means of transportation used while wearing the personal monitor was included.

While in the third trimester of pregnancy, all subjects wore a small backpack containing a personal ambient air monitor during the day and were asked to place the monitor near the bed at night (within breathing zone) for 48 consecutive hours. PM_{2.5} was collected on a precleaned quartz microfiber filter, and semi-volatile vapors and aerosols were collected on a polyurethane foam (PUF) plug backup. The personal air sampling pumps operated continuously at 4 L/min, and an average of 11.2 m³ of air was drawn through the sampler. Each sampling event was characterized as to accuracy in flow rate, time, and completeness of documentation for quality control. Filters and PUF cartridges were extracted together and analyzed using gas chromatography at the Southwest Research Institute. PAH analytes included benz[*a*]anthracene (BaA), benzo[*b*]fluoranthene (BbF), benzo[*k*]fluoranthene (BkF), benzo[*g,h,i*]perylene (BghiP), BaP, chrysene/iso-chrysene, dibenz[*a,h*]anthracene (DahA), indeno[*1,2,3*]pyrene (IP), and pyrene. Immediately after collection, the air monitoring samples were brought to the molecular epidemiology laboratory at the Mailman School of Public Health, inventoried, and frozen. Once each month, air monitoring samples were shipped on ice to Southwest Research Institute, where they were then stored at –12°C. Within 10 days of arrival at the institute, the PUF plug and filter were placed

in a Soxhlet extractor (Corning, Corning, NY), which was spiked with terphenyl-d14 as a recovery surrogate, and extracted with 6% diethyl ether in hexanes for 16 hr and then concentrated the extractant to 1 mL. Extracts were frozen at –12°C before analysis. Each extract was retained until after that woman delivered and was analyzed once the woman was fully enrolled in the study. PAHs were determined in samples using an Agilent 6890 gas chromatograph/5973 mass spectrometer (Agilent Technologies, Palo Alto, CA). Cleaned extracts were injected into a 30 m × 0.25 mm inner-diameter DB-5.625 gas chromatography analytical column (J&W Scientific, Folsom, CA), and the gas chromatography/mass spectrometry instrument was scanned to monitor two selected ions per analyte to achieve low-level detection. Deuterated polyaromatic hydrocarbons were used as internal standards to perform quantification (U.S. EPA 1999). Details on limits of detection and quality assurance methods are presented elsewhere (Whyatt et al. 2002).

Personal monitoring measurements began in February 1998, and data were collected for 348 subjects. Because of problems with equipment, air flow, or timekeeping, three observations were considered unreliable and were not included in the analysis. Data for an additional 18 subjects were flagged because of problems with accuracy in flow rate, time, or completeness of documentation but were considered still useable; statistical analysis was conducted including and excluding these 18 subjects.

Multiple linear regression analysis was used to explore the roles of questionnaire variables as predictors of individual PAH concentrations as well as the sum of all compounds (total PAH) and the sum of all PAH compounds recognized as probable human carcinogens (total B2). Potential predictor variables included those related to demographics, SES, home characteristics, and personal activity patterns. SAS (version 8; SAS Institute, Cary, NC) was used for all the statistical analysis. PAH concentrations were log transformed before statistical analysis to normalize positively skewed distributions. All variables that were univariate significant ($\alpha = 0.05$) were considered for inclusion into the multivariate model. To avoid including collinear covariates in this exploratory analysis, a forward selection method was used for model selection. Univariate significant covariates not included in the forward selection model were evaluated for their influence by adding them into the forward selection model one at a time. The parameter estimate of each covariate was compared with the estimate in the forward selection model; if the parameter estimate changed by 15% or more, the new covariate was left in the model. Because covariate data were missing, the sample size of

the multivariate models was allowed to float. To evaluate the appropriateness of the model and to identify outliers, residual analysis was conducted.

In the regression analysis, education and income were measured as ordinal categorical variables (highest degree earned and strata of annual household income from the previous year). Race/ethnicity was included as a binary variable (African American or other), and subject's age was included as a continuous variable. Employment status during pregnancy and exposure to tobacco smoke, fire, or coal smoke in the 48 hr of monitoring were included as binary variables. Variables measuring the amount of time the heat in the subject's building was on, the use of electric heaters, and whether fuel oil was the primary type of heat were included to describe home heating during personal monitoring. Season was measured as an ordinal categorical variable based on month of monitoring (coded as 0 = JunJulAug, 1 = SepOctNov, 2 = MarAprMay, 3 = DecJanFeb). Binary variables were used to capture whether subjects had fried, broiled, or burned food during the monitoring period as well as whether they had burned incense or lived near a bus depot. Hours spent outdoors and in transit and the level of traffic near home were included as categorical variables, and indicator variables were used to describe the most common type of transit. If fewer than 20 subjects were available in one level of a categorical variable, levels were combined.

Results

Demographic and SES characteristics for the 344 subjects who did not smoke during the 48 hr of air monitoring and for whom there was reliable personal monitoring data are summarized in Table 1. The mean age was 25 years.

Table 1. Subject demographic and socioeconomic characteristics.

Characteristics	Cohort
No.	344
Age [years (mean ± SD)]	25 ± 6
Race/ethnicity [n (%)]	
African American	148 (43.0)
Dominican Caribbean	195 (56.7)
Caribbean	1 (0.3)
Household income [n (%)]	
< \$10,000	150 (43.6)
\$10,001–20,000	99 (28.8)
\$20,001–50,000	65 (18.9)
> \$50,000	12 (3.5)
Not reported	18 (5.2)
Employed ^a [n (%)]	189/344 (54.9)
Sales	42/189 (22.2)
Restaurant	20/189 (10.6)
Office, health care, telemarketing	59/189 (31.2)
School employee	3/189 (1.6)
Factory	16/189 (8.5)
Other or not reported	49/189 (25.9)

^aTotal number of employed women ($n = 189$) is a subset of the overall total ($n = 344$).

Slightly more than half of the subjects reported Dominican (or other Caribbean) race/ethnicity (57%), with the remainder African American. Household income was < \$10,000/year for 43.6% of subjects, and < \$20,000 for 72% of subjects. Of the 344 subjects, 189 (54%) reported being employed, with most jobs in sales and office, health care, or telemarketing.

During pregnancy, 70% of subjects reported spending four or fewer hours outdoors each day. More than 70% of subjects reported spending 7 or more waking hours at home each day. Subways or buses were the most common modes of transportation for 57% of subjects. Driving in private automobiles was reported as the most common transport mode for only 8% of subjects. Other characteristics obtained by questionnaire included cooking fuel: 95% reported cooking with gas stoves, and 11 of 316 (4%) subjects reported cooking on a charcoal grill during pregnancy. Although 38% of women did not know what type of heating was in their building, 53% reported that their heating supply was from fuel oil combustion. No subject reported having a kerosene heater in the home. Nearly half (47%) of the women said they burned candles in the home, and 28% reported burning incense in the home during pregnancy. Although personal smoking was an exclusion criterion for enrollment, data collected at the prenatal questionnaire indicated that seven of the 343 subjects smoked. A total of 143 of 339 (42%) reported that there was at least one smoker in the house. One or two smokers in the home was reported by 32% of subjects, and 4% reported having three to eight smokers in the home. Of 189 subjects with jobs, 31 (16%) reported that there was a smoker in the workplace.

Descriptive statistics for the personal monitoring data are presented in Table 2. Mean PAH concentrations ranged from 0.06 ng/m³ for DahA to 4.1 ng/m³ for pyrene; mean BaP concentration was 0.50 ng/m³. Total B2 indicates the sum of PAH compounds that have been identified as probable human carcinogens by the U.S. EPA Integrated Risk Information System. This includes all compounds except BghiP and pyrene, which

remain unclassified as to human carcinogenicity (IRIS 1997). Two of the 344 women reported that the air monitor was not near them for up to 8 hr during the 48-hr monitoring period; 3 of 344 women reported not being near the monitor for 8–16 hr, and 3 of 344 reported being away from the monitor for 16–24 hr.

In the univariate analysis, age and SES factors, including household income last year and highest educational degree earned, were not significantly associated with PAH concentration. Employment during pregnancy was significantly associated with log[BkF] ($p = 0.04$); however, log[BkF] did not vary significantly with type of employment. Season (coded as 0 = JunJulAug, 1 = SepOctNov, 2 = MarAprMay, 3 = DecJanFeb) was significantly associated with log[BaA], log[BaP], log[BbF], log[BghiP], log[BkF], log[chrysene], log[IP], and log[pyrene], and the log of the sum of B2 PAHs. In addition, mean concentrations of each PAH compound did not vary significantly with whether the subject lived in Manhattan or the Bronx.

The associations between log PAH concentration and questionnaire information collected on activities and indoor and outdoor environmental exposures during the 48 hr of monitoring were analyzed for total PAH, total B2 PAHs, as well as BbF, BghiP, BaA, BkF, BaP, and pyrene. The predictors of log PAH concentrations adjusted for other covariates are presented in Table 3. Season, hours spent outdoors, and African-American race were significantly associated with log[BaA] in the adjusted model. The amount of time the building heat was on substantially changed the parameter estimate for season and was therefore included in the model as a covariate. Season and having burned incense during the 48 hr of monitoring were significantly associated with log[BaP] in the adjusted model; the amount of time building heat was on was added because it substantially changed the parameter estimate of season. Season and burning incense were significantly associated with log[BbF], and the amount of time building heat was on was added because it substantially influenced the estimate of season.

Table 2. Descriptive statistics of personal exposure to PAHs (ng/m³) in New York City.

PAH	No.	Mean ± SD	Minimum	25th Percentile	Median	75th Percentile	Maximum
BaA	344	0.300 ± 0.245	0.031	0.165	0.239	0.346	1.878
BaP	344	0.491 ± 0.646	0.021	0.194	0.326	0.555	6.439
BbF	327	0.585 ± 0.629	0.044	0.269	0.423	0.664	7.259
BghiP	344	1.268 ± 1.908	0.024	0.518	0.789	1.335	18.11
BkF	328	0.161 ± 0.170	0.021	0.051	0.119	0.189	1.594
Chrysen	344	0.380 ± 0.291	0.048	0.218	0.309	0.445	2.845
DahA	344	0.055 ± 0.040	0.015	0.043	0.046	0.054	0.457
IP	344	0.689 ± 0.888	0.024	0.286	0.447	0.757	7.409
Pyrene	344	4.073 ± 6.227	0.695	2.004	2.702	4.17	96.97
Total PAH	344	7.966 ± 9.478	1.514	4.287	5.652	8.923	127.11
Total B2	344	2.625 ± 2.655	0.341	1.284	1.923	3.034	26.700

Chrysen, chrysene/iso-chrysen.

Season, burning incense, and amount of time heat was on were significantly associated with log[BghiP] in the adjusted model. Season was the only significant covariate in the adjusted model for log[BkF], and its parameter estimate was influenced by the amount of time building heat was on and burning incense. The number of hours spent outdoors was the only significant predictor of log[pyrene] and the log of total PAH concentration. Season and burning incense were significantly associated with the log of the B2 PAHs, and amount of time building heat was on was included because it substantially changed the estimate for season. The regression analysis for the multivariate log[BaA] model was repeated with one observation with a particularly large jackknife residual (-4.04) removed from the sample; there was no change in the

regression model selected. Similarly, excluding the 18 observations that had been flagged because of problems with the PAH measurements did not substantially alter the regression models that best describe log PAH concentration.

Discussion

In the present study, the personal exposure of 344 women residing in northern Manhattan and the South Bronx ranged from 0.06 ng/m³ for DahA to 4.1 ng/m³ for pyrene. Mean BaP concentration was 0.50 ng/m³ and mean total PAH was 8.0 ng/m³. In personal exposure studies of other nonoccupational cohorts, reported values of BaP concentration ranged from 0.07 to 4.3 ng/m³ (Buckley et al. 1995; Sisovic et al. 1996; Zmirou et al. 2000). Exposure to total PAH ranged from 1.09 to

24.7 ng/m³, although studies are not easily comparable because different PAH compounds were measured (Sisovic et al. 1996; Zmirou et al. 2000). Personal exposure of cyclists and drivers measured over 1-hr trips along inner-city routes in Amsterdam ranged from 7.5 to 24.7 ng/m³ for total PAH (Van Wijnen et al. 1995). Observed occupational exposures were much higher; typical personal exposures to total PAH ranged from 1.3 to 397 µg/m³, with the highest exposure measured for asphalt paving workers in California (Angerer et al. 1997; Pyy et al. 1997; Van Delft et al. 1998; Watts et al. 1998). It is important to note that the populations monitored in these studies were likely to have differed substantially in terms of demographics, activities, and home characteristics from those monitored in the present study. In addition, differences in how values below the limits of detection were handled can influence the comparability of results from different studies. Ambient background concentrations of PAHs measured in urban settings or other locations near heavy traffic or industrial emissions typically range from 0.5 to 20 ng/m³ (ATSDR 1995; Zmirou et al. 2000).

Summertime concentrations of pyrene measured in this cohort were substantially higher than those measured in the winter. One possible explanation for the higher summertime concentration is that volatilization from road surfaces, soil, and vegetation, which is positively associated with temperature, may be an appreciable source of pyrene in New York City. Air concentrations of most individual PAHs have been found to be negatively correlated with temperature, a result of increased sources of PAHs from heating in the winter or seasonal variation in boundary layer depth. However, significantly higher concentrations of low- and medium-molecular-weight PAHs have been measured in warmer weather in other urban locations, which the authors attributed to volatilization from surfaces with higher temperature (Dimashki et al. 2001; Liroy and Greenberg 1990).

Concentrations of individual PAHs measured in summer and winter for the New York City cohort are compared with those measured in other nonoccupational groups in Table 4. All personal monitoring was conducted on nonsmoking adults, except the Zagreb study, in which up to four smokers may have been included in the sample of 15 subjects. Summertime personal exposures to PAHs in the New York City cohort were generally similar to those measured for individuals in these previous studies; however, exposure to pyrene was particularly high. Most of these studies monitored only particulate PAHs and thus were likely to underestimate concentrations of the more volatile PAHs such as pyrene.

Table 3. Multivariable adjusted predictors of personal exposure to PAHs.

PAH (n)	Predictors in final model	$\beta \pm SE$	p-Value	R ²
BaA (281)	Season	0.11 ± 0.04	0.01	0.07
	Hours per day spent outdoors	0.10 ± 0.04	0.006	
	African American	0.16 ± 0.07	0.03	
	Time building heat was on	0.004 ± 0.04	0.91	
BaP (276)	Season	0.27 ± 0.05	< 0.0001	0.23
	Burning incense	0.33 ± 0.15	0.03	
	Time building heat was on	0.09 ± 0.05	0.06	
BbF (261)	Season	0.10 ± 0.05	0.04	0.10
	Burning incense	0.33 ± 0.14	0.02	
	Time building heat was on	0.08 ± 0.05	0.08	
BghiP (276)	Season	0.18 ± 0.05	0.001	0.16
	Burning incense	0.31 ± 0.15	0.04	
	Time building heat was on	0.10 ± 0.05	0.03	
BkF (262)	Season	0.15 ± 0.05	0.004	0.09
	Burning incense	0.25 ± 0.15	0.09	
	Time building heat was on	0.05 ± 0.05	0.29	
Pyrene (292)	Hours per day spent outdoors	0.15 ± 0.03	< 0.0001	0.06
Total PAH (292)	Hours per day spent outdoors	0.10 ± 0.03	0.002	0.03
Total B2 (276)	Season	0.14 ± 0.04	0.002	0.12
	Burning incense	0.26 ± 0.12	0.04	
	Time building heat was on	0.05 ± 0.04	0.24	

Table 4. Mean PAH concentration measured by personal monitoring (ng/m³).

Season, PAH	New York City	Grenoble, France ^a	Zagreb, Croatia ^b	Phillipsburg, NJ ^c	Milan, Italy ^d
Summer					
BaA	0.22 ± 0.14	0.05 ± 0.03	—	—	—
BaP	0.20 ± 0.18	0.07 ± 0.02	0.13 ± 0.08	0.11	—
BbF	0.36 ± 0.30	0.12 ± 0.06	0.39 ± 0.19	—	—
BghiP	0.56 ± 0.46	0.30 ± 0.21	0.36 ± 0.16	—	—
BkF	0.09 ± 0.07	0.06 ± 0.02	0.13 ± 0.07	—	—
Chrysene	0.30 ± 0.19	ND	—	—	—
IP	0.33 ± 0.26	0.46 ± 0.09	—	—	—
Pyrene	4.45 ± 3.48	ND	—	—	1.67 ± 1.64
Winter					
BaA	0.32 ± 0.25	1.24 ± 2.12	—	—	—
BaP	0.63 ± 0.76	1.05 ± 0.87	4.34 ± 3.07	0.66	—
BbF	0.68 ± 0.81	1.17 ± 0.99	3.82 ± 2.13	—	—
BghiP	1.45 ± 1.95	1.05 ± 0.38	3.40 ± 1.61	—	—
BkF	0.17 ± 0.15	0.52 ± 0.38	2.02 ± 2.44	—	—
Chrysene	0.40 ± 0.3	0.30*	—	—	—
IP	0.82 ± 0.95	2.88 ± 3	—	—	—
Pyrene	3.50 ± 4.21	0.25 ± 0.004	—	—	4.18 ± 2.37

—, PAH compound was not measured in the study.

^aZmirou et al. (2000): 48-hr average; n = 29 summer, n = 32 winter. ^bSisovic et al. (1996): 7-day average; n = 15. ^cBuckley et al. (1995): 24-hr average; n = 18 summer, n = 14 winter; GSD not reported. ^dPastorelli et al. (1999): 24-hr average; n = 37 summer, n = 35 winter. *Only one filter had detectable values of chrysene.

During the winter, this cohort's personal exposure to most PAHs was lower than that measured in the European studies. Personal exposure to BaP measured in Zagreb was approximately 7-fold higher than that measured in New York City, whereas exposure to BaP for subjects in New York City and New Jersey was very similar. Wintertime personal exposures measured in Europe were much higher than their summer counterparts: Wintertime BaP levels were 15- and 33-fold higher in Grenoble and Zagreb, respectively, whereas winter BaP in New York City increased by only 3-fold. Traffic emissions are not expected to vary seasonally, whereas PAH emissions from heating sources are expected to increase substantially during the winter (Dimashki et al. 2001).

Overall, the regression analysis suggested that season and a variety of indoor and outdoor sources of PAH influence the personal exposure of this cohort. Pyrene concentration was an order of magnitude larger than that of most other individual PAHs; therefore, the association found between total PAH and questionnaire variables was largely driven by pyrene. It is worth noting, however, that in each of the final regression models selected, the R^2 values were modest; the models explained 3–23% of the variance in the concentration of the individual PAHs.

The results of the linear regression modeling were somewhat inconclusive regarding the influence of traffic on personal PAH exposures in this cohort. Personal exposure to pyrene, total PAHs, and BaA was associated with time spent outdoors, which may reflect the influence of traffic sources; however this was not the case for most individual PAH compounds investigated. Perhaps most important with respect to health risks posed from carcinogenic PAHs, no questionnaire variables indicating exposure to traffic sources were significant predictors in the model for total B2. This may reflect the limited ability of the questionnaires to measure exposure to traffic sources. Ongoing work by our group is directed at developing more objective measures of local traffic density that can be related to PAH exposures. Previous studies have shown strong local influence of traffic on airborne elemental carbon concentrations in New York City (Kinney et al. 2000; Lena et al. 2002).

Personal monitoring data were collected from 1998 through 2001; therefore, in addition to individual variation in personal exposure, the measurements also include temporal and spatial variation. To a limited extent, temporal variation and meteorologic factors such as boundary layer depth were adjusted for indirectly by including season in the model. However, continuous measurements of background PAH or $PM_{2.5}$ levels at a stationary monitor would have allowed these factors to

be considered more explicitly and individual variation in personal exposure to be better isolated. Because of the high cost of maintaining a stationary PAH monitor, we do not have background PAH measurements, making a comprehensive temporal analysis difficult.

Because of the high cost of analysis, gas-phase and particulate-phase PAHs were extracted together in each sample. However, this creates only a minor limitation in this study because eight of the nine PAHs measured in this study are high-molecular-weight compounds and should exist primarily as particulates. Pyrene, a medium-molecular-weight compound, would have comprised most of the gas-phase sample.

Although personal monitoring is the most accurate measurement of personal exposure, the personal monitoring measurements were taken over 48 hr, a short period of time providing only a snap shot of the exposures to PAHs subjects experienced throughout pregnancy. The exposures encountered during the 48-hr interval may not have been representative of exposures during the subject's entire pregnancy.

To our knowledge, this is the first study in which personal monitoring has been used to characterize the PAH exposure of urban minority populations in the United States. Environmental exposures experienced by minority and low-income populations in urban areas are of particular interest because of the myriad environmental health risks these populations face and their suspected increased susceptibility to them. A particular strength of this investigation is its very large sample size for a study using personal monitoring. Although we have documented widespread exposures to PAHs in this urban population, possible associations between these exposures and adverse health risks are addressed elsewhere (Perera et al. 2003).

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