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Personal and Ambient Exposures to Air Toxics in Camden, New Jersey

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Includes a Commentary by the Institute's Health Review Committee

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ABOUT HEI

The Health Effects Institute is a nonprofit corporation chartered in 1980 as an independent research organization to provide high-quality, impartial, and relevant science on the effects of air pollution on health. To accomplish its mission, the institute

- Identifies the highest-priority areas for health effects research;
- Competitively funds and oversees research projects;
- Provides intensive independent review of HEI-supported studies and related research;
- Integrates HEI's research results with those of other institutions into broader evaluations; and
- Communicates the results of HEI's research and analyses to public and private decision makers.

HEI typically receives half of its core funds from the U.S. Environmental Protection Agency and half from the worldwide motor vehicle industry. Frequently, other public and private organizations in the United States and around the world also support major projects or research programs. HEI has funded more than 280 research projects in North America, Europe, Asia, and Latin America, the results of which have informed decisions regarding carbon monoxide, air toxics, nitrogen oxides, diesel exhaust, ozone, particulate matter, and other pollutants. These results have appeared in the peer-reviewed literature and in more than 200 comprehensive reports published by HEI.

HEI's independent Board of Directors consists of leaders in science and policy who are committed to fostering the public-private partnership that is central to the organization. The Health Research Committee solicits input from HEI sponsors and other stakeholders and works with scientific staff to develop a Five-Year Strategic Plan, select research projects for funding, and oversee their conduct. The Health Review Committee, which has no role in selecting or overseeing studies, works with staff to evaluate and interpret the results of funded studies and related research.

All project results and accompanying comments by the Health Review Committee are widely disseminated through HEI's Web site (www.healtheffects.org), printed reports, newsletters and other publications, annual conferences, and presentations to legislative bodies and public agencies.

ABOUT THIS REPORT

Research Report 160, *Personal and Ambient Exposures to Air Toxics in Camden, New Jersey*, presents a research project funded by the Health Effects Institute and conducted by Dr. Paul J. Lioy, of the Environmental and Occupational Health Sciences Institute, Piscataway, New Jersey, and his colleagues. This report contains three main sections.

The HEI Statement, prepared by staff at HEI, is a brief, nontechnical summary of the study and its findings; it also briefly describes the Health Review Committee's comments on the study.

The Investigators' Report, prepared by Lioy and colleagues, describes the scientific background, aims, methods, results, and conclusions of the study.

The Commentary is prepared by members of the Health Review Committee with the assistance of HEI staff; it places the study in a broader scientific context, points out its strengths and limitations, and discusses remaining uncertainties and implications of the study's findings for public health and future research.

This report has gone through HEI's rigorous review process. When an HEI-funded study is completed, the investigators submit a draft final report presenting the background and results of the study. This draft report is first examined by outside technical reviewers and a biostatistician. The report and the reviewers' comments are then evaluated by members of the Health Review Committee, an independent panel of distinguished scientists who have no involvement in selecting or overseeing HEI studies. During the review process, the investigators have an opportunity to exchange comments with the Review Committee and, as necessary, to revise their report. The Commentary reflects the information provided in the final version of the report.

PREFACE

HEI's Research Program on Air Toxics Hot Spots

INTRODUCTION

Air toxics comprise a large and diverse group of air pollutants that, with sufficient exposure, are known or suspected to cause adverse effects on human health, including cancer; effects on the development of organs and tissues, and damage to the respiratory, immune, neurologic, and reproductive systems. These compounds are emitted by a variety of indoor and outdoor sources, and large numbers of people are exposed to them. Therefore, the compounds are a cause for public health concern, even though the ambient levels are generally low. The low ambient levels are one reason that tools and techniques for assessing specific health effects of air toxics are very limited.

Air toxics are not regulated by the U.S. Environmental Protection Agency (EPA) under the National Ambient Air Quality Standards. However, the EPA is required under the Clean Air Act and its amendments to characterize, prioritize, and address the effects of air toxics on public health and the environment, and it has the statutory authority to control and reduce the release of air toxics. The EPA is also required to regulate or consider regulating air toxics derived, at least in part, from motor vehicles (referred to as mobile-source air toxics [MSATs]) by setting standards for fuels, vehicle emissions, or both. In 2001 the EPA designated 21 high-priority MSATs that needed to be reduced (U.S. EPA 2001a). However, the EPA did not take any specific regulatory action at that time because rules mandating the reduction of sulfur in both gasoline and diesel fuels as a way to decrease particulate matter (PM) in emissions were expected to result in the reduction of several MSATs as well (U.S. EPA 2000, 2001b). Subsequently, the EPA identified eight MSATs that, based on their emissions and reported toxicity, pose the greatest risk to health — benzene, 1,3-butadiene, formaldehyde, acrolein, naphthalene, polycyclic organic matter, diesel PM, and

diesel exhaust organic gases — and mandated the reduction of benzene in gasoline and of hydrocarbons (including MSATs) in exhaust (U.S. EPA 2007). In 2007, HEI published a critical review of the literature on exposure to and health effects associated with these highest-priority MSATs (HEI Air Toxics Review Panel 2007).

In trying to understand the potential health effects of exposure to toxic compounds, scientists often turn first to evaluating responses in highly exposed populations, such as occupationally exposed workers. However, workers and their on-the-job exposures are not representative of the general population, and therefore such studies may be somewhat limited in value.

Another strategy is to study populations living in “hot spots” — areas that have high concentrations of these pollutants owing to their proximity to one or more sources. Some hot spots may have sufficiently high pollutant concentrations to make them suitable locations for studies to determine whether there is a link between exposure to air toxics and an adverse health outcome. Such areas offer the potential to conduct health investigations in groups that are more representative of the general population. Before health effects studies can be initiated, however, actual exposures to pollutants — including their spatial and temporal distributions — in such hot-spot areas must be characterized.

DESCRIPTION OF THE PROGRAM

In January 2003, HEI issued a Request for Applications (RFA 03-1) entitled “Assessing Exposure to Air Toxics,” seeking studies aimed at identifying and characterizing exposure to air toxics from a variety of sources in areas or situations where concentrations were expected to be elevated. The rationale for the RFA was that understanding exposures in hot spots, as

Preface

well as the sources of these exposures, would improve our ability to select the most appropriate sites, populations, and endpoints for subsequent health studies. HEI was particularly interested in studies that focused on the high-priority MSATs.

Five studies were funded under this RFA to represent a diversity of possible hot-spot locations and air toxics. The study by Lioy and colleagues described in this report (Research Report 160) is the fourth of the five to be published. The five studies are summarized below.

“Assessing Personal Exposure to Air Toxics in Camden, New Jersey,” Paul J. Lioy, Environmental and Occupational Health Sciences Institute, Piscataway, New Jersey (Principal Investigator)

In the study presented in this report, Lioy and colleagues measured ambient and personal exposure concentrations of air toxics and fine PM for 107 nonsmoking participants in two neighborhoods of Camden, New Jersey. One, considered to be a hot spot, had a high density of industrial facilities serviced by truck traffic and nearby busy roads. The other, with no industrial sources but near several highways, was considered an urban reference site. The investigators collected four sets of 24-hour personal air samples for the study subjects — in summer and winter and on weekdays and weekend days — and made simultaneous measurements of ambient pollutant concentrations at a fixed monitoring site in each neighborhood. To characterize finer spatial variability in pollutant levels, air toxics levels were measured at multiple sampling sites in each neighborhood during three sampling periods. The investigators also used modeling to estimate the contribution of ambient sources to personal exposure.

“Measurement and Modeling of Exposure to Air Toxics and Verification by Biomarkers,” Roy M. Harrison, University of Birmingham, Birmingham, United Kingdom (Principal Investigator)

In the study described in HEI Research Report 143 (2009), Harrison and colleagues investigated personal exposure to a broad range of air toxics, with the goal

of developing detailed personal-exposure models that would take various microenvironments into account. Repeated measurements of exposure to selected air toxics were made for each of 100 healthy nonsmoking adults who resided in urban, suburban, or rural areas of the United Kingdom, among which exposures to traffic were expected to differ; repeated urine samples were also collected for analysis. Harrison and colleagues developed models to predict personal exposure on the basis of microenvironmental concentrations and data from time–activity diaries; they then compared measured personal exposure with modeled estimates of exposure.

“Assessing Exposure to Air Toxics,” Eric M. Fujita, Desert Research Institute, Reno, Nevada (Principal Investigator)

In the study presented in HEI Research Report 156 (2011), Fujita and colleagues measured the concentrations of PM and MSATs on major California freeways and compared them with corresponding measurements obtained at fixed monitoring stations. The diurnal and seasonal variations in concentrations of selected pollutants and the contribution of diesel- and gasoline-powered vehicles to selected air toxics and elemental carbon were also determined.

“Air Toxics Exposure from Vehicular Emissions at a U.S. Border Crossing,” John Spengler, Harvard School of Public Health, Boston, Massachusetts (Principal Investigator)

The study by Spengler and colleagues, presented in HEI Research Report 158 (2011), assessed concentrations of MSATs surrounding the plaza adjacent to the Peace Bridge, a major border crossing between the United States and Canada, located in Buffalo, New York. Three fixed monitoring sites were used to compare pollutant concentrations upwind and downwind of the plaza. Meteorologic measurements and hourly counts of trucks and cars crossing the bridge were used to examine the relationship between the concentrations of air toxics and traffic density. To study spatial distributions of pollutants, members of the investigative team used portable instruments and a

Global Positioning System device to obtain location-specific, time-stamped measurements as they walked along four routes in a residential neighborhood near the plaza.

**“Air Toxics Hot Spots in Industrial Parks and Traffic,”
Thomas Smith, Harvard School of Public Health,
Boston, Massachusetts (Principal Investigator)**

The study by Smith and colleagues measured levels of air toxics and PM in truck cabins and in 15 truck terminals across the United States. The HEI study was added to an ongoing study, funded by the National Cancer Institute, of the relationship between exposure to diesel exhaust and mortality from lung cancer among dockworkers and truck drivers at more than 200 truck terminals in the United States. Smith and colleagues measured pollutants at upwind and downwind locations around the perimeter of each terminal and at loading docks. The degree of variation at different locations and the influence of wind direction were also evaluated with the goal of identifying the potential impact of truck terminals on the surrounding areas. Continuous sampling was performed inside delivery truck cabins during a work shift. This study has been completed and is currently in the review and publication process. It is expected to be released in early 2012.

HEI is committed to continuing research on air toxics — for example, as part of studies to assess the health outcomes of air quality actions or studies to evaluate the effects of new technologies and fuels. Further information on these programs can be obtained at the HEI Web site (www.healtheffects.org).

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HEI STATEMENT

Synopsis of Research Report 160

Air Toxics Exposures in a Potential Hot Spot in New Jersey

INTRODUCTION

Air toxics comprise a large and diverse group of air pollutants that, with sufficient exposure, are known or suspected to cause adverse effects on human health. The Clean Air Act requires the U.S. Environmental Protection Agency to characterize, prioritize, and address the effects of air toxics on public health and the environment.

Although ambient concentrations of some of these air toxics have been monitored by state or local agencies in some areas, the characterization of personal exposures to air toxics has been limited. And although ambient concentrations are generally low, so-called hot spots might exist where concentrations of one or more air toxics, and consequent exposures of area populations, could be elevated. In 2003, HEI targeted research to identify and characterize potential air toxics hot spots, with the aim of conducting future health studies in these locations.

APPROACH

Dr. Paul J. Liroy of the Environmental and Occupational Health Sciences Institute of Piscataway, New Jersey, and colleagues evaluated ambient and personal exposures to particulate matter with aerodynamic diameter $\leq 2.5 \mu\text{m}$ ($\text{PM}_{2.5}$) and several air toxics in 107 nonsmoking residents of two neighborhoods in Camden, New Jersey. Residents of both neighborhoods were predominantly low-income. The investigators hypothesized that one neighborhood, Waterfront South, was an air toxics hot spot, defined as having elevated concentrations of air toxics compared with those of a nearby area. Waterfront South has numerous industrial sites serviced by heavy truck traffic and is close to major roads; its residents were thus likely to be exposed to air toxics from multiple sources. The other neighborhood, Copewood–Davis, was selected as the control, or comparison, site for the pollutant measurements. It was near Waterfront South and had no industrial sites.

Between June 2004 and July 2006 the investigators collected four sets of 24-hour personal air samples from the study subjects and made simultaneous measurements of ambient pollutant concentrations at a fixed monitoring site in each neighborhood. To assess how pollutant concentrations varied by season, they collected personal and ambient samples in summer and in winter. To assess how mobile sources (particularly truck traffic) contributed to pollutant concentrations, they collected personal and ambient samples on weekdays and weekend days (anticipating that concentrations would be higher on weekdays). To assess finer spatial variability in air toxics concentrations, they also conducted a saturation-sampling substudy in which pollutant measurements were made at 38 monitoring sites in the two neighborhoods.

Liroy and colleagues measured concentrations of multiple air toxics, including volatile organic compounds (VOCs) (especially benzene, toluene, ethylbenzene, *m*- & *p*-xylenes, *o*-xylenes, methyl *tert*-butyl ether [MTBE], chloroform, carbon tetrachloride, and hexane); aldehydes (especially formaldehyde and acetaldehyde); and polycyclic aromatic hydrocarbons (PAHs) (especially naphthalene, phenanthrene, pyrene, and benzo[a]pyrene as representatives of compounds with two to five benzene rings). They also measured $\text{PM}_{2.5}$ concentrations.

In addition to comparing concentrations of air pollutants in Waterfront South with those in Copewood–Davis, the investigators used an alternative definition of a hot spot — i.e., having elevated concentrations of air toxics compared with those of other, more distant areas in New Jersey and across the United States — to compare concentrations of air pollutants in Waterfront South and Copewood–Davis with concentrations at other locations.

KEY FINDINGS AND CONCLUSIONS

HEI's Health Review Committee, which undertook an independent review of the study, thought

the study had made an important contribution to the characterization of possible air toxics hot spots.

The investigators reported that one of the neighborhoods, Waterfront South, had consistently higher ambient concentrations than the other, Copewood–Davis, of PM_{2.5}, toluene, xylenes, and PAHs. Thus, by the investigators' original definition of a hot spot (i.e., having elevated concentrations compared with those of a nearby control, or comparison, area with fewer industrial sites), Waterfront South could be considered a hot spot for these pollutants. However, ambient concentrations in Copewood–Davis of several other pollutants — benzene, MTBE, chloroform, carbon tetrachloride, hexane, and acetaldehyde — were as high as or higher than those in Waterfront South. The Committee generally considered the measurements of the air pollutants to have been accurate and reliable. However, they were concerned about the validity of the absolute concentrations of benzene, formaldehyde, and acetaldehyde, because they were much higher than those reported in other studies, and there appeared to be some specific problems with the measurement method for formaldehyde.

The Committee concurred with the investigators' conclusion that, by their alternative definition of a hot spot (i.e., having elevated concentrations compared with those of other, more distant areas in New Jersey and across the United States), *both* neighborhoods could be considered hot spots for PM_{2.5}, benzene, toluene, xylenes, MBTE, and aldehydes. At the same time, the Committee cautioned that comparisons of pollutant concentrations across studies are difficult because studies differ in averaging periods, sampling and analysis methods, types of sampling sites, and meteorologic conditions (temperature and wind speed and direction are particularly relevant). The Committee also noted that, although ambient concentrations of PAHs were higher in Waterfront South than in Copewood–Davis, they were not higher than those measured at other urban sites in the United States and that therefore, by the alternative definition, neither neighborhood was a hot spot for PAHs.

The Committee concluded that the study had provided useful information on personal exposures in the two neighborhoods. For most of the pollutants, measured personal concentrations were higher than the respective ambient concentrations measured at the study's two fixed monitoring sites, suggesting contributions from sources other than outdoor (i.e., indoor or occupational). In general, variations in personal concentrations did not correspond with variations in ambient concentrations. The Committee considered

that this highlighted an important continuing issue for policy making and future health effects studies of air toxics — namely the difficulty of relating personal exposures to ambient concentrations measured at a central monitoring site.

The Committee found that the investigators' saturation-sampling substudy, in which measurements of ambient air toxics were made in three campaigns at several monitoring sites in each neighborhood, provided valuable information about the spatial variability at small scales of pollutant concentrations that could be compared with information from each neighborhood's fixed monitoring site. The results showed that, even within a possible hot spot, spatial variability in ambient concentrations can be found, suggesting that people in some locations within a neighborhood are likely to be exposed to much higher concentrations than those recorded at a fixed monitoring site in the same neighborhood. This finding again underscores the importance of individualized personal monitoring of pollutants. Lioy and colleagues also provided useful information that showed temporal (weekday versus weekend) and seasonal (summer versus winter) variability in individual pollutant concentrations.

The Committee concluded that in retrospect choosing Copewood–Davis as the control area for the study was not ideal. Although the neighborhood was free of industrial facilities, it was subject to high concentrations of mobile-source emissions from traffic on roads in or near the area as well as of emissions transported from adjacent areas (including Waterfront South). The two neighborhoods were also subject to the same regional meteorologic conditions and the same pollutant plume from Philadelphia (some 20 miles [32 km] away), and both were topographically simple.

In summary, the current study provided valuable information about ambient and personal concentrations of PM_{2.5} and a large number of air toxics and demonstrated elevated ambient concentrations (compared with other areas in New Jersey and across the United States) of some air toxics in both of these lower-socioeconomic-status neighborhoods. At the same time, the findings illustrate the difficulties of defining an area *a priori* as a potential hot spot — or as a control location. The design of future exposure and health effects studies in hot spots will need to take multiple pollutant sources and meteorologic factors into consideration to achieve sufficient contrasts in pollutant concentrations between appropriately chosen hot spots and background locations.

INVESTIGATORS' REPORT

Personal and Ambient Exposures to Air Toxics in Camden, New Jersey

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ABSTRACT

Personal exposures and ambient concentrations of air toxics were characterized in a pollution “hot spot” and an urban reference site, both in Camden, New Jersey. The hot spot was the city’s Waterfront South neighborhood; the reference site was a neighborhood, about 1 km to the east, around the intersection of Copewood and Davis streets. Using personal exposure measurements, residential ambient air measurements, statistical analyses, and exposure modeling, we examined the impact of local industrial and mobile pollution sources, particularly diesel trucks, on personal exposures and ambient concentrations in the two neighborhoods.

Presented in the report are details of our study design, sample and data collection methods, data- and model-analysis approaches, and results and key findings of the study. In summary, 107 participants were recruited from non-smoking households, including 54 from Waterfront South and 53 from the Copewood–Davis area. Personal air samples were collected for 24 hr and measured for 32 target compounds — 11 volatile organic compounds (VOCs*),

four aldehydes, 16 polycyclic aromatic hydrocarbons (PAHs), and particulate matter (PM) with an aerodynamic diameter $\leq 2.5 \mu\text{m}$ ($\text{PM}_{2.5}$). Simultaneously with the personal monitoring, ambient concentrations of the target compounds were measured at two fixed monitoring sites, one each in the Waterfront South and Copewood–Davis neighborhoods. To understand the potential impact of local sources of air toxics on personal exposures caused by temporal (weekdays versus weekend days) and seasonal (summer versus winter) variations in source intensities of the air toxics, four measurements were made of each subject, two in summer and two in winter. Within each season, one measurement was made on a weekday and the other on a weekend day. A baseline questionnaire and a time diary with an activity questionnaire were administered to each participant in order to obtain information that could be used to understand personal exposure to specific air toxics measured during each sampling period. Given the number of emission sources of air toxics in Waterfront South, a spatial variation study consisting of three saturation-sampling campaigns was conducted to characterize the spatial distribution of VOCs and aldehydes in the two neighborhoods. Passive samplers were used to collect VOC and aldehyde samples for 24- and 48-hr sampling periods simultaneously at 22 and 16 grid-based sampling sites in Waterfront South and Copewood–Davis, respectively.

Results showed that measured ambient concentrations of some target pollutants (mean \pm standard deviation [SD]), such as $\text{PM}_{2.5}$ ($31.3 \pm 12.5 \mu\text{g}/\text{m}^3$), toluene ($4.24 \pm 5.23 \mu\text{g}/\text{m}^3$), and benzo[a]pyrene ($0.36 \pm 0.45 \text{ ng}/\text{m}^3$), were significantly higher ($P < 0.05$) in Waterfront South than in Copewood–Davis, where the concentrations of $\text{PM}_{2.5}$, toluene, and

This Investigators’ Report is one part of Health Effects Institute Research Report 160, which also includes a Commentary by the Health Review Committee and an HEI Statement about the research project. Correspondence concerning the Investigators’ Report may be addressed to Dr. Paul J. Liroy, Environmental and Occupational Health Sciences Institute, 170 Frelinghuysen Road, Piscataway, NJ 08854.

Although this document was produced with partial funding by the United States Environmental Protection Agency under Assistance Award CR-83234701 to the Health Effects Institute, it has not been subjected to the Agency’s peer and administrative review and therefore may not necessarily reflect the views of the Agency, and no official endorsement by it should be inferred. The contents of this document also have not been reviewed by private party institutions, including those that support the Health Effects Institute; therefore, it may not reflect the views or policies of these parties, and no endorsement by them should be inferred.

* A list of abbreviations and other terms appears at the end of the Investigators’ Report.

benzo[*a*]pyrene were $25.3 \pm 11.9 \mu\text{g}/\text{m}^3$, $2.46 \pm 3.19 \mu\text{g}/\text{m}^3$, and $0.21 \pm 0.26 \text{ ng}/\text{m}^3$, respectively. High concentrations of specific air toxics, such as $60 \mu\text{g}/\text{m}^3$ for toluene and $159 \mu\text{g}/\text{m}^3$ for methyl *tert*-butyl ether (MTBE), were also found in areas close to local stationary sources in Waterfront South during the saturation-sampling campaigns. Greater spatial variation in benzene, toluene, ethylbenzene, and xylenes (known collectively as BTEX) as well as of MTBE was observed in Waterfront South than in Copewood–Davis during days with low wind speed. These observations indicated the significant impact of local emission sources of these pollutants and possibly of other pollutants emitted by individual source types on air pollution in Waterfront South. (Waterfront South is a known hot spot for these pollutants.) There were no significant differences between Waterfront South and Copewood–Davis in mean concentrations of benzene or MTBE, although some stationary sources of the two compounds have been reported in Waterfront South. Further, a good correlation ($R > 0.6$) was found between benzene and MTBE in both locations. These results suggest that automobile exhausts were the main contributors to benzene and MTBE air pollution in both neighborhoods. Formaldehyde and acetaldehyde concentrations were found to be high in both neighborhoods. Mean (\pm SD) concentrations of formaldehyde were $20.2 \pm 19.5 \mu\text{g}/\text{m}^3$ in Waterfront South and $24.8 \pm 20.8 \mu\text{g}/\text{m}^3$ in Copewood–Davis. A similar trend was observed for the two compounds during the saturation-sampling campaigns. The results indicate that mobile sources (i.e., diesel trucks) had a large impact on formaldehyde and acetaldehyde concentrations in both neighborhoods and that both are aldehyde hot spots. The study also showed that $\text{PM}_{2.5}$, aldehydes, BTEX, and MTBE concentrations in both Waterfront South and Copewood–Davis were higher than ambient background concentrations in New Jersey and than national average concentrations, indicating that both neighborhoods are in fact hot spots for these pollutants.

Higher concentrations were observed on weekdays than on weekend days for several compounds, including toluene, ethylbenzene, and xylenes (known collectively as TEX) as well as PAHs and $\text{PM}_{2.5}$. These observations showed the impact on ambient air pollution of higher traffic volumes and more active industrial and commercial operations in the study areas on weekdays. Seasonal variations differed by species. Concentrations of TEX, for example, were found to be higher in winter than in summer in both locations, possibly because of higher emission rates from automobiles and reduced photochemical reactivity in winter. In contrast, concentrations of MTBE were found to be significantly higher in summer than in winter in both locations, possibly because of higher evaporation rates from gasoline in summer. Similarly,

concentrations of heavier PAHs, such as benzo[*a*]pyrene, were found to be higher in winter in both locations, possibly because of higher emission rates from mobile sources, the use of home heating, and the reduced photochemical reactivity of benzo[*a*]pyrene in winter. In contrast, concentrations of lighter PAHs were found to be higher in summer in both locations, possibly because of volatilization of these compounds from various surfaces in summer. In addition, higher concentrations of formaldehyde were observed in summer than in winter, possibly because of significant contributions from photochemical reactions to formaldehyde air pollution in summer.

Personal concentrations of toluene ($25.4 \pm 13.5 \mu\text{g}/\text{m}^3$) and acrolein ($1.78 \pm 3.7 \mu\text{g}/\text{m}^3$) in Waterfront South were found to be higher than those in the Copewood–Davis neighborhood ($13.1 \pm 15.3 \mu\text{g}/\text{m}^3$ for toluene and $1.27 \pm 2.36 \mu\text{g}/\text{m}^3$ for acrolein). However, personal concentrations for most of the other compounds measured in Waterfront South were found to be similar to or lower than those than in Copewood–Davis. (For example, mean \pm SD concentrations were $4.58 \pm 17.3 \mu\text{g}/\text{m}^3$ for benzene, $4.06 \pm 5.32 \mu\text{g}/\text{m}^3$ for MTBE, $16.8 \pm 15.5 \mu\text{g}/\text{m}^3$ for formaldehyde, and $0.40 \pm 0.94 \text{ ng}/\text{m}^3$ for benzo[*a*]pyrene in Waterfront South and $9.19 \pm 34.0 \mu\text{g}/\text{m}^3$ for benzene, $6.22 \pm 19.0 \mu\text{g}/\text{m}^3$ for MTBE, $16.0 \pm 16.7 \mu\text{g}/\text{m}^3$ for formaldehyde, and $0.42 \pm 1.08 \text{ ng}/\text{m}^3$ for benzo[*a*]pyrene in Copewood–Davis.) This was probably because many of the target compounds had both outdoor and indoor sources. The higher personal concentrations of these compounds in Copewood–Davis might have resulted in part from higher exposure to environmental tobacco smoke (ETS) of subjects from Copewood–Davis.

The Spearman correlation coefficient (R) was found to be high for pollutants with significant outdoor sources. The R 's for MTBE and carbon tetrachloride, for example, were > 0.65 in both Waterfront South and Copewood–Davis. The R 's were moderate or low (0.3 – 0.6) for compounds with both outdoor and indoor sources, such as BTEX and formaldehyde. A weaker association ($R < 0.5$) was found for compounds with significant indoor sources, such as BTEX, formaldehyde, PAHs, and $\text{PM}_{2.5}$. The correlations between personal and ambient concentrations of MTBE and BTEX were found to be stronger in Waterfront South than in Copewood–Davis, reflecting the significant impact of local air pollution sources on personal exposure to these pollutants in Waterfront South.

Emission-based ambient concentrations of benzene, toluene, and formaldehyde and contributions of ambient exposure to personal concentrations of these three compounds were modeled using atmospheric dispersion modeling and Individual Based Exposure Modeling (IBEM)

software, respectively, which were coupled for analysis in the Modeling Environment for Total Risk (MENTOR) system. The compounds were associated with the three types of dominant sources in the two neighborhoods: industrial sources (toluene), exhaust from gasoline-powered motor vehicles (benzene), and exhaust from diesel-powered motor vehicles (formaldehyde). Subsequently, both the calculated and measured ambient concentrations of each of the three compounds were separately combined with the time diaries and activity questionnaires completed by the subjects as inputs to IBEM–MENTOR for estimating personal exposures from ambient sources. Modeled ambient concentrations of benzene and toluene were generally in agreement with the measured ambient concentrations within a factor of two, but the values were underestimated at the high-end percentiles. The major local (neighborhood) contributors to ambient benzene concentrations were from mobile sources in the study areas; both mobile and stationary (point and area) sources contributed to the ambient toluene concentrations. This finding can be used as guidance for developing better emission inventories to characterize, through modeling, the ambient concentrations of air toxics in the study areas. Modeled ambient concentrations of formaldehyde significantly underestimated the actual concentrations; they could only account for 4% to 20% of the ambient measurements. This appeared to have been caused by the fact that current countywide emission inventories of mobile sources of formaldehyde are significantly lower than the actual number of vehicles, particularly trucks (275–825 truck trips/day), passing through or near the two neighborhoods. The estimated percentage contributions of personal exposures from ambient sources to the three compounds were generally higher in the Waterfront South hot spot than in the Copewood–Davis urban reference area. This finding demonstrates the hot spot characteristic of there being higher local ambient source impacts on personal exposures than there are in other areas. Non-ambient sources of benzene and toluene were also found to be significant contributors to personal exposures to these compounds for the population studied.

In conclusion, our study demonstrated that both the Waterfront South and Copewood–Davis neighborhoods are in fact hot spots for various air toxics, depending on the sources of air pollution in each area. Simultaneous measurements of personal and ambient concentrations of air toxics in a hot spot provided important data for examining the impact of local ambient emission sources on personal exposures and hence on potential health risks. Moreover, information obtained from our saturation-sampling campaign increased understanding of the spatial distribution

of air toxics and identified sources of concern in hot spots. Finally, this was the first study to incorporate ambient and exposure measurements and an innovative exposure-modeling approach into the same effort. The sampling and modeling approaches implemented here will provide valuable tools for future air pollution and exposure research on hot spots.

INTRODUCTION

Ambient air toxics contain numerous compounds, including VOCs, aldehydes, and PAHs. These can be emitted from a single source or a wide variety of stationary and mobile sources. Mobile sources, particularly trucks and buses, are the largest sources contributing to ambient air toxics in urban areas. Epidemiologic studies have shown an association between exposure to elevated concentrations of air toxics and adverse health effects, including respiratory and cardiovascular diseases as well as carcinogenic, neurologic, reproductive, and developmental effects (Leikauf 1992; Dockery et al. 1993; Norback et al. 1995; Bascom et al. 1996; Hagen et al. 2000; Delfino 2002).

Higher exposures and associated health risks are suspected for subpopulations that live in areas with mixed urban and industrial sources of air toxics, i.e., hot spots for air toxics. In the early 1980s the California legislature implemented a program for air toxic hot spots that focused on identifying facilities potentially having localized health risk impacts and on reducing the extent of the health risks below the level of significance. The legislature defined a hot spot as “an area where the concentration of air toxics is at a level where individuals may be exposed to an elevated risk of adverse health effects” (California Air Resources Board 1987). More specifically, a hot spot is a small area (e.g., a community or neighborhood) with dense sources of air toxics. Average or peak concentrations of one or more air pollutants in a hot spot are much higher than those in surrounding areas and than state and national concentrations. We used these definitions to develop our strategies for studying Waterfront South and Copewood–Davis.

Recently, several studies were conducted to characterize personal exposures to air toxics. The impact of indoor and outdoor sources as well as other important factors in determining personal exposure to air toxics were also examined in one or more of the studies (the Relationship of Indoor, Outdoor and Personal Air [RIOPA] study [Weisel et al. 2005]; the Toxic Exposure Assessment, a Columbia–Harvard [TEACH] study [Kinney et al. 2002]; a study conducted in Baltimore by Buckley and colleagues [2005; Payne-Sturges et al. 2004]; and the European Exposure Assessment

Project [EXPOLIS] [Jantunen et al. 1998]). Each showed that mobile sources were the primary contributors to many air toxics in urban areas. The relationships among personal, indoor, and outdoor concentrations varied greatly for the various air toxics, depending on the sources of the species and on personal activities. The studies also provided evidence of exposure to a wide range of toxic air pollutants among minority and disadvantaged populations as a result of their proximity to a variety of air pollution sources that could be associated with a hot spot. Personal time-activity patterns were found to have significant impacts on personal exposure.

There are, however, significant gaps in understanding the health risks associated with exposure to ambient air toxics. First, measurements of ambient concentrations of most air toxics are nonexistent or insufficient in most communities to evaluate health risks, and the identification and study of hot spots is only beginning to develop important databases. Second, routine ambient air monitoring is usually conducted at a single fixed site, which might not accurately capture local or micro-environmental high-concentration exposures in presumed hot spot areas. The risks resulting from exposures to air toxics in such hot spots might therefore be underestimated when using the

available monitoring data and model predictions (U.S. Environmental Protection Agency [U.S. EPA] 2006b). Third, large spatial and temporal variations in air toxics are expected in hot spots because of the large number of co-located emission sources in or near them (Sweet and Vermette 1992; Hung et al. 2005; Vardoulakis et al. 2005; Smith et al. 2007). To date, however, this local variability has not been thoroughly investigated, and as a consequence an individual's exposure to air toxics might be mischaracterized in a hot spot. Fourth, the measurement and quantification of personal exposures to most air toxics are still limited, and the contribution of ambient air pollution to personal exposure in hot spots has not been quantitatively determined. Lack of such information limits the ability to accurately estimate the health risks resulting from exposures to ambient air toxics (U.S. EPA 2006a), thereby hampering regulatory agencies' abilities to effectively define and implement strategies that reduce the true risks to human health associated with long-term or periodic exposures to air toxics.

Camden, New Jersey, has been considered a hot spot for more than 25 years (Harkov et al. 1983, 1984; Liroy 1990; U.S. EPA 2006a,b). In the 1980s the issue of

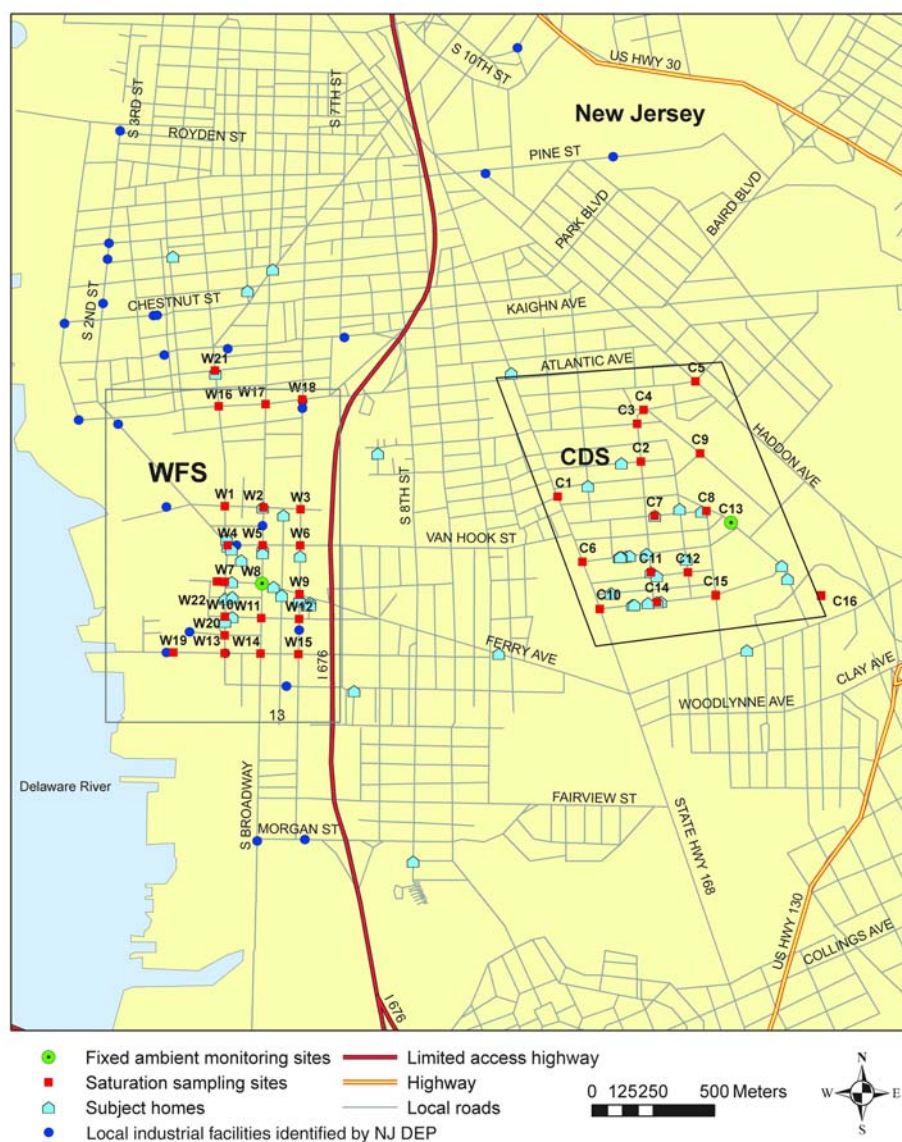


Figure 1. Locations of fixed ambient monitoring sites, saturation sampling sites, subjects' homes, and industrial facilities in the Waterfront South (WFS) and Copewood-Davis (CDS) neighborhoods of Camden, New Jersey.

air toxics in Camden was recognized in the Airborne Toxic Element and Organic Substances (ATEOS) study, completed between 1981 and 1983 (Harkov et al. 1983, 1984; Liroy 1990). The sources of air toxics in Camden include local mobile sources, local industrial and manufacturing sources, and urban sources originating in Philadelphia. The principal industrial sources of air toxics are located in and near the Waterfront South neighborhood (Figure 1). Waterfront South

is in the southwest of Camden and is a rectangular area measuring about 1200 m north to south and 800 m west to east (Figure 1). According to a report from the New Jersey Department of Environmental Protection (NJDEP) (2005), 26 industrial and manufacturing facilities are sources of air toxics in and near Waterfront South (Table 1), including municipal waste and sewage treatment facilities, recycling plants, and metal processing facilities. Given the number

Table 1. Point Sources of Air Pollutants in Waterfront South and Copewood–Davis (NJDEP 2005)

ID	Facility	Type of Operation	Main Pollutants Emitted
PS1	Camden County Municipal Utilities Authority	Sewage treatment facility	PM, MTBE, BTEX, chloroform, carbon tetrachloride, formaldehyde, PAHs
PS2	Mafco	Spice and extract manufacturing	PM, propylene glycol, ammonium
PS3	Art Metalcraft	Electroplating	Hydrogen cyanide, metals
PS4	PSE&G Camden Coal Gas	Electric power generation	Benzene, toluene, formaldehyde
PS5	Georgia Pacific (Domtar Gypsum)	Gypsum product manufacturing	Hexane, benzene, toluene, formaldehyde, metals
PS6	Container Recyclers of Camden	Metal container manufacturing	Xylenes, titanium dioxide
PS7	STE LAR Textiles (Camden Gas and Mantle, Welbach)	Superfund site	Not available
PS8	American Minerals, Inc.	Metal ore mining	PM
PS9	Hospital Central Services Inc. Laundry	Laundry service	PM, metals
PS10	Camden County Resource Recovery Association	Materials recovery	PM, formaldehyde, PAHs
PS11	St. Lawrence Cement	Cement plant	PM, metals
PS12	Colonial Processing	Welding and soldering equipment manufacturing (paint appl.)	PM, xylenes, hexane
PS13	Comarco	Pork processing	PM, lead
PS14	Broadway Finishing	Industrial paint shop	Toluene, xylenes, methyl ethyl ketone
PS15	SL Surface Technologies	Electroplating	PM, metals
PS16	Camdett	Industrial inorganic chemicals, NEC (alumina)	Ammonia
PS17	Camden Cogeneration	Fossil fuel electric power generation	PM, ammonia
PS18	F.W. Winter	Secondary smelting, refining, and alloying of nonferrous metal	PM, metals
PS19	State Metal Industries	Secondary smelting, nonferrous metals	PM, hexane, toluene, dioxins, metals
PS20	CWS Industries	Electroplating, plating, polishing	PM, cadmium
PS21	Duro Plating Co.	Electroplating	Cadmium, hydrogen cyanide
PS22	Camden Iron & Metal (The Pier)	Recyclable material wholesaler	Toluene, hexane, metals
PS23	Steve's Auto Parts	Car scrapping, automotive body repair, painting	PM, gasoline emissions
PS24	Plastic Consulting & Mfg. Co.	Coating, engraving, allied services	PM, metals, VOCs
PS25	Teideken Bros. Auto Body	Automotive body, paint, and interior repair and maintenance	Methyl isobutyl ketone
PS26	Cam Core	Secondary aluminum smelter	PM, toluene, hexane, metals
PS27	Peerless Castings	Aluminum foundry	PM, toluene, hexane, ethylene

of sources in the neighborhood, the character and quantity of air toxics released locally will not be constants in space or time. In addition, it has been estimated that 275 to 825 trucks per day travel and idle in Waterfront South to serve local industry. Further, two major roads — Interstate 676 (I-676) and Route 168 (NJ-168) (Figure 1) — are located near the eastern boundary of Waterfront South. I-676 is a major commuting route between New Jersey and Philadelphia and has a traffic volume of ~80,000 vehicles/day (New Jersey Department of Transportation 2006). NJ-168 is one of the principal local roads in the Camden area; it connects downtown Camden and the Benjamin Franklin Bridge with Philadelphia and has a traffic volume of ~25,000 vehicles/day.

Waterfront South has a population of approximately 1,700, consisting mainly of low-income residents; they are 57.8% black, 27.2% Hispanic–Latino, and 23% white or others (Table 2, U.S. Census 2000). Almost all homes in Waterfront South are located less than 200 m from at least one stationary source of air toxics. The quantity of air toxics in the neighborhood is increased further by the diesel exhaust emitted by the trucks traveling through and idling in the neighborhood. Residents can therefore be expected to experience high exposures to air toxics. However, there had been no actual measurements of neighborhood or personal concentrations of air toxics in Waterfront South. Lack of such data limits the ability to quantify health risks from exposure to air toxics and prevents the identification and prioritization of the ambient sources that need to be mitigated to reduce such health risks.

Copewood–Davis is a residential area located 1 km east of Waterfront South and was used in the study as an urban reference site. The rationale for the selection of the site is discussed below.

STUDY OBJECTIVES AND SPECIFIC AIMS

The main objectives of this 3-year study were (1) to characterize personal and ambient exposures to air toxics in Waterfront South and the Copewood–Davis neighborhood and (2) to assess the impact of local stationary and mobile sources on the air quality of the neighborhood and on personal exposures. The specific aims of the study were the following:

1. To quantify personal exposures to air toxics for people living in Waterfront South and Copewood–Davis;
2. To quantify ambient neighborhood concentrations of air toxics in the two neighborhoods;
3. To characterize spatial variations in VOCs and aldehydes in both locations;
4. To examine differences in concentration and composition between the targeted air toxics in the two neighborhoods;
5. To examine temporal (weekdays versus weekend days) and seasonal (summer versus winter) variations in the targeted air toxics in the two neighborhoods;
6. To examine associations between ambient air pollution and personal exposures; and
7. To assess the impact of local sources of air toxics, particularly diesel emissions, on personal exposures and air quality in the two neighborhoods based on measurements of air toxics with support from exposure modeling, using our source-to-dose IBEM–MENTOR modeling system (Georgopoulos et al. 2005; Georgopoulos and Liou 2006). Our modeling was intended to do the following:
 - To apply IBEM–MENTOR modeling to a hot spot in Camden, New Jersey, and evaluate the viability of the application by comparing its results with personal and ambient measurements collected in the field study;

Table 2. Demographic Data for Camden and the State of New Jersey^a

	Black (%)	Hispanic (%)	Median Household Income (\$)	Individuals Below Poverty (%)
New Jersey State	13.6	12.5	55,136	8.5
Camden County	18.1	9.7	48,097	10.4
Camden City	53.3	38.8	23,421	35.5
Waterfront South	57.8	27.2	22,417 ^b	33.8 ^b
Copewood–Davis	69.3	25.6	NA	NA

^a U.S. Census 2000.

^b NJDEP, 2005.

NA indicates not available.

- To substantiate the impact of local industrial and mobile air toxic sources on personal exposures and local air pollution in Waterfront South and Copewood–Davis; and
- To establish the confidence or system refinements needed to apply IBEM–MENTOR to future population exposure and human health assessments in hot spots.

In addition to these aims, research was also conducted to evaluate a GeoLogger global positioning device (GeoStats, Atlanta, GA) for recording personal movements and to assess the application of global positioning system (GPS) technology to exposure research. The information collected by the GeoLogger was exploratory in nature and was therefore not incorporated into the current report; GeoLogger results are shown in Appendix F.

METHODS AND STUDY DESIGN

Personal and neighborhood ambient concentrations of air toxics were measured in both Waterfront South and Copewood–Davis from June 2004 through July 2006. Our target pollutants included 11 VOCs, four aldehydes, 16 PAHs, and fine PM (i.e., PM_{2.5}) (Table 3). They were selected because each was known to have sources (e.g., emissions of air toxics from diesel trucks, a waste combustor, and paint application) in or near Waterfront South. One hundred and seven subjects were recruited for the

study from nonsmoking homes, 54 from Waterfront South and 53 from Copewood–Davis. Four 24-hr integrated air samples were collected for each subject: two in summer and two in winter, one each per season on a weekday and on a weekend day. Because of concern about potential exposure to ETS in these communities, nicotine (a marker of ETS) was also measured in the personal air samples. During each sampling period, between two and six subjects were monitored on the same sampling day. Ambient concentrations of the target compounds were measured at a fixed monitoring site in each neighborhood at the same time as the personal monitoring. There were 92 days of ambient air sampling in all. A baseline questionnaire (Appendix C) and a time diary (Appendix D) with an activity questionnaire (Appendix E) were administered during each sampling period to obtain information about the demographics of the participants, housing characteristics and locations, facilities and usage, and personal exposure activities. Given the substantial number of stationary sources of air toxics in Waterfront South, a spatial variation study consisting of three saturation-sampling campaigns was conducted to characterize the spatial variation in VOCs and aldehydes across the study areas.

To examine differences in the concentrations and composition of the selected personal and ambient air toxics between the two neighborhoods and to evaluate the potential impact of local air toxic sources on these concentrations, the personal and ambient samples collected in our suspected hot spot (Waterfront South) were compared

Table 3. Target Compounds

Fine PM	VOCs	Aldehydes	PAHs
PM _{2.5}	Methyl <i>tert</i> -butyl ether (MTBE) <i>n</i> -Hexane Benzene Toluene Ethylbenzene <i>m</i> - & <i>p</i> -Xylenes <i>o</i> -Xylene Styrene Chloroform Carbon tetrachloride 1,3-Butadiene	Formaldehyde Acetaldehyde Acrolein Propionaldehyde	Naphthalene Acenaphthylene Acenaphthene Fluorene Phenanthrene Anthracene Fluoranthene Pyrene Chrysene Benzo[<i>a</i>]anthracene Benzo[<i>k</i>]fluoranthene Benzo[<i>b</i>]fluoranthene Benzo[<i>a</i>]pyrene Indeno[1,2,3- <i>cd</i>]pyrene Dibenzo[<i>a,h</i>]anthracene Benzo[<i>ghi</i>]perylene

with those collected in Copewood–Davis, using basic statistics and mixed-effects models. (Details of the data analysis are presented below, in the Statistical Analysis and Modeling Application section.) Given the fact that most facility operations are more active and that truck traffic is heavier on weekdays than on weekend days, the samples collected on weekdays were compared with those collected on weekend days to examine temporal variations in personal and ambient air concentrations. Seasonal variations were also characterized, by comparing results for summer and winter, in order to explore the consequences of the differing amounts of time people spend in outdoor and indoor environments and of the differing degrees of outdoor air penetration into buildings (caused principally by open versus closed windows and doors). Finally, the relationships between ambient concentrations and personal exposures were explored, and the ambient contributions to personal total exposures were estimated using a mixed-effects model analysis.

Ambient concentrations of benzene, toluene, and formaldehyde were modeled using atmospheric dispersion models, and personal concentrations of these three compounds were estimated using IBEM–MENTOR (Georgopoulos et al. 2005; Georgopoulos and Liroy 2006). Both measured and model-estimated ambient concentrations were used for the calculation of personal concentrations. The amounts of time spent in each microenvironment as reported in the time–activity diaries were also used for estimating personal exposures. The modeling results were then compared with the measured personal and ambient air concentrations to assess the impact of local sources of outdoor industrial and mobile air toxics on local air pollution and personal exposures in the two neighborhoods and to identify significant sources of the air toxics.

Investigators from the Environmental and Occupational Health Sciences Institute (EOHSI) (a joint program of Rutgers University and the University of Medicine and Dentistry of New Jersey [UMDNJ]) and collaborators (not funded by the grant) from the NJDEP worked jointly to conduct the study. Below are details of our study approach, sample collection and measurement methods, and the data and model analysis methods used in the study.

SUBJECT RECRUITMENT

Before beginning subject recruitment, our proposed field protocol and informed-consent form were reviewed and approved by the UMDNJ Institutional Review Board (UMDNJ IRB #4464). The informed-consent procedures met government guidelines. Only subjects from non-smoking households were selected for participation in the study. During the first home visit, the study was explained

to each participant, and a completed consent form was obtained from the participant (or, in the case of minors, his or her parents or guardians) prior to entry into the study.

A variety of activities and approaches were used by the research team to recruit subjects in the two neighborhoods, including contact with local community leaders, working with a local liaison, attending local community events, advertising in local newspapers, and word of mouth. Recruitment details are presented in the following subsections.

Community Connections

Our approach to the Waterfront South and Copewood–Davis neighborhoods included gaining support for the project from both community and religious leaders. Beginning in December 2003, local community leaders and church groups were contacted, and the objectives and aims of the study were explained to them. Drs. Liroy and Fan and other team members held a number of individual and group meetings with leaders from Heart of Camden, Sacred Heart Church, South Camden Citizens in Action, Hope Gospel Church, and other organizations. In some cases, these meetings were also attended by scientists from the NJDEP (such as Joann Held or Dr. Linda Bonnanno) and by community residents. In addition, Dr. Liroy had occasional meetings with the pastor of Sacred Heart, a Catholic church that was a focal point of activities in Waterfront South. In these ways the study's purpose, goals, and measurement plans were introduced and strong working relationships were built, giving our study significant support in the two neighborhoods. The pastor of Sacred Heart, for example, allowed us to use church property as the fixed monitoring site for Waterfront South and actively assisted in our recruitment efforts. John Neal, an undergraduate student at Rutgers University–Camden and a resident of Waterfront South, was hired to work as a local liaison for the study. His enthusiasm and interest significantly augmented our subject recruitment. John also disseminated study information to local residents, providing information about local activities and events, such as the Sayers Street block party in Copewood–Davis, and assisted in field sampling work.

Local Meetings and Events

Members of the project team attended several community meetings and events related to environmental issues in Waterfront South and Copewood–Davis. The meetings and events were sponsored by the Camden County Health Department, Camden Area Health Education Center, Virtual Health, Camden Healthy Futures Committee, and the City of Camden. In September 2004, Dr. Liroy presented the study plan to the New Jersey Clean Air Council, which had

been developing a strategy to help the state coordinate efforts for improving the environment and economic opportunities in Camden. He also presented information about the area's air pollution and health issues. Through these activities, an extensive outreach network was established that helped in the identification and recruitment of study subjects from both neighborhoods.

Direct Contact with Local Residents

Subjects were also recruited by means of direct contact with local residents. Research team members handed out flyers (with a list of the study's objectives and pictures of the sampling pack) to residents in the two neighborhoods. Each individual was given a brief synopsis of the study. Our field team member also carried a sampling pack (Figure 2) and demonstrated how it would be used on the personal monitoring days if the individual agreed to participate in the study. We had a high acceptance rate using this approach, and many residents signed up for the study immediately after the meeting.

Advertisement in Local Newspapers

With the help and support of a community leader in Copewood–Davis, an article about the study was published in a local newspaper. We were also invited by local group leaders to present our study to residents in both Waterfront South and Copewood–Davis. Dr. Fan and other team members, for example, were invited to the Sayrs Street Block Party in May 2004 and to a Little League Baseball game in July 2004 in Copewood–Davis. During these events, we were offered special space to advertise our study, and our team members developed closer relationships with the residents. We had tremendous success with subject recruitment during the events; more than 30 people in Copewood–Davis signed up for the study.

Word of Mouth

A few very enthusiastic participants took the liberty of acting as volunteer recruiters, informing friends and family members about the study and in fact generating a satisfactory response rate for new inquiries.

In summary, subjects were successfully recruited through the activities and approaches described above, and the recruiting goal of the study was met. Detailed information about the subjects will be presented below, in the Experimental Results and Discussion section.

SURVEY INSTRUMENT: QUESTIONNAIRES

A baseline questionnaire, time diary, and activity questionnaire developed for the RIOPA studies (Weisel et al. 2005)

were modified for use in the current study by altering and shortening questions to reflect the study's objectives. Our baseline questionnaire (Appendix C) included sections on household and participant characteristics; demographics of the participant; family income; housing characteristics



Figure 2. Sampling backpack on a subject. Front view (top) shows the personal exposure monitors on the shoulder straps and the pump in the backpack. Back view (bottom) shows the GeoLogger receiver hanging outside the backpack.

and location, facilities, and usage; personal exposure activities before the study period; and a number of questions about the participant's respiratory health. The time diary and activity questionnaire (Appendix D and Appendix E) included a 24-hr activity log to record time spent in each microenvironment and a series of detailed questions about activities and the use of selected consumer products. The questionnaires were translated into Spanish, and a Spanish-speaking EOHSI field staff member was available for each household where Spanish was the native language. The baseline questionnaire was completed during the first home visit, and the time diary and activity questionnaire were filled out by the participant for each sampling period. Data on demographics, time spent in various microenvironments, contact with ETS, and indoor sources of air toxics reported in the questionnaires were analyzed and are included in this report.

FIXED MONITORING SITES

Because of the small size of the two neighborhoods (i.e., 800 m east to west and 1200 m north to south for Waterfront South and 600 m east to west and 1000 m north to south for Copewood–Davis; Figure 1), a single fixed monitoring site was selected in each neighborhood to measure concentrations of air toxics in it. There were four major criteria for the monitoring sites: (1) ambient air pollutant concentrations measured at each site had to be representative of the neighborhood's air pollutant concentrations, (2) the site had to be in or near the neighborhood, (3) there had to be no nearby field sources of air pollution (< 10 ft away), and (4) the site had to be easy to reach yet have a secure spot to position the samplers. The fixed monitoring sites selected for the two neighborhoods are described below.

Fixed Monitoring Site in Waterfront South

There are 26 industrial and manufacturing facilities as well as large numbers of mobile sources of air toxics (particularly diesel trucks serving local industry) in Waterfront South. Wishing to maximize the fixed monitoring site's proximity to as many of these sources as possible, we selected Sacred Heart Church as the site location for the neighborhood (Figure 3). The church is on Ferry Avenue, near the center of Waterfront South. No stationary sources are located within 50 m of it. In addition, the church is within one block of the intersection of Ferry and Broadway, the entrance point for trucks traveling through the neighborhood. Diesel emissions are one of the major concerns for local residents; measurements collected from the site could thus capture the impact of diesel exhaust on the neighborhood's air pollution and be used to help address the community's concern. The monitor was placed

on the first-floor porch of the church in a protected area, and the sampling port was directed toward Ferry Avenue. Before starting the main study, ambient air was sampled for a week at the site (and at the fixed monitoring site in Copewood–Davis [see below]) to ensure that the sites were in fact suitable.

Fixed Monitoring Site in Copewood–Davis

The NJDEP's existing ambient monitoring site at the intersection of Copewood and Davis streets was selected as the fixed monitoring site for the Copewood–Davis neighborhood in our study (Figure 3). Copewood–Davis is a residential area located ~1000 m east of Waterfront South (Figure 1). It includes approximately 6,200 residents and is composed mainly of low-income households; the residents are 64% black, 15% Hispanic–Latino, and 21% white or others (U.S. Census 2000). They are similar in socioeconomic status to the residents of Waterfront South. There are no identifiable nearby industrial facilities (< 1000 m) in Copewood–Davis. The NJDEP's existing site was located near the residences of potential study subjects, allowing for easy access, and had no nearby stationary air pollution sources. It is worth noting that Copewood–Davis is surrounded by major roads, including NJ-168 to the west (< 100 m away) and Haddon Avenue to the east (< 100 m away). Traffic volume on Haddon Avenue was ~8,000 vehicles/day. About 80% of the subjects in Copewood–Davis lived within three blocks (< 500 m) of NJ-168. Because the prevailing wind in Camden is from the west, it was expected that mobile-source pollution as well as pollution transported from Waterfront South would affect the personal and ambient air toxic concentrations measured in Copewood–Davis. For these reasons, Copewood–Davis was considered an urban reference site rather than a background area. Nevertheless, there was an advantage to using this location as an urban reference site. The NJDEP has been routinely sampling for volatile hazardous air pollutants in Copewood–Davis for more than 10 years as part of the U.S. EPA's Urban Air Toxics Monitoring Program (NJDEP 2005). Copewood–Davis was also included in the ATEOS study, conducted by EOHSI members in the 1980s (Harkov et al. 1983, 1984; Weisel et al. 2005). The results of the current study can thus be directly compared with those obtained by the NJDEP and from the ATEOS study.

MEASUREMENT OF AIR POLLUTANTS

Sampling Setup for Personal and Ambient Air Sample Collection

Our target compounds were PM_{2.5}, VOCs, aldehydes, and PAHs (Table 3). PM_{2.5} was collected using Leland Legacy personal samplers with 37-mm Teflon filters (SKC

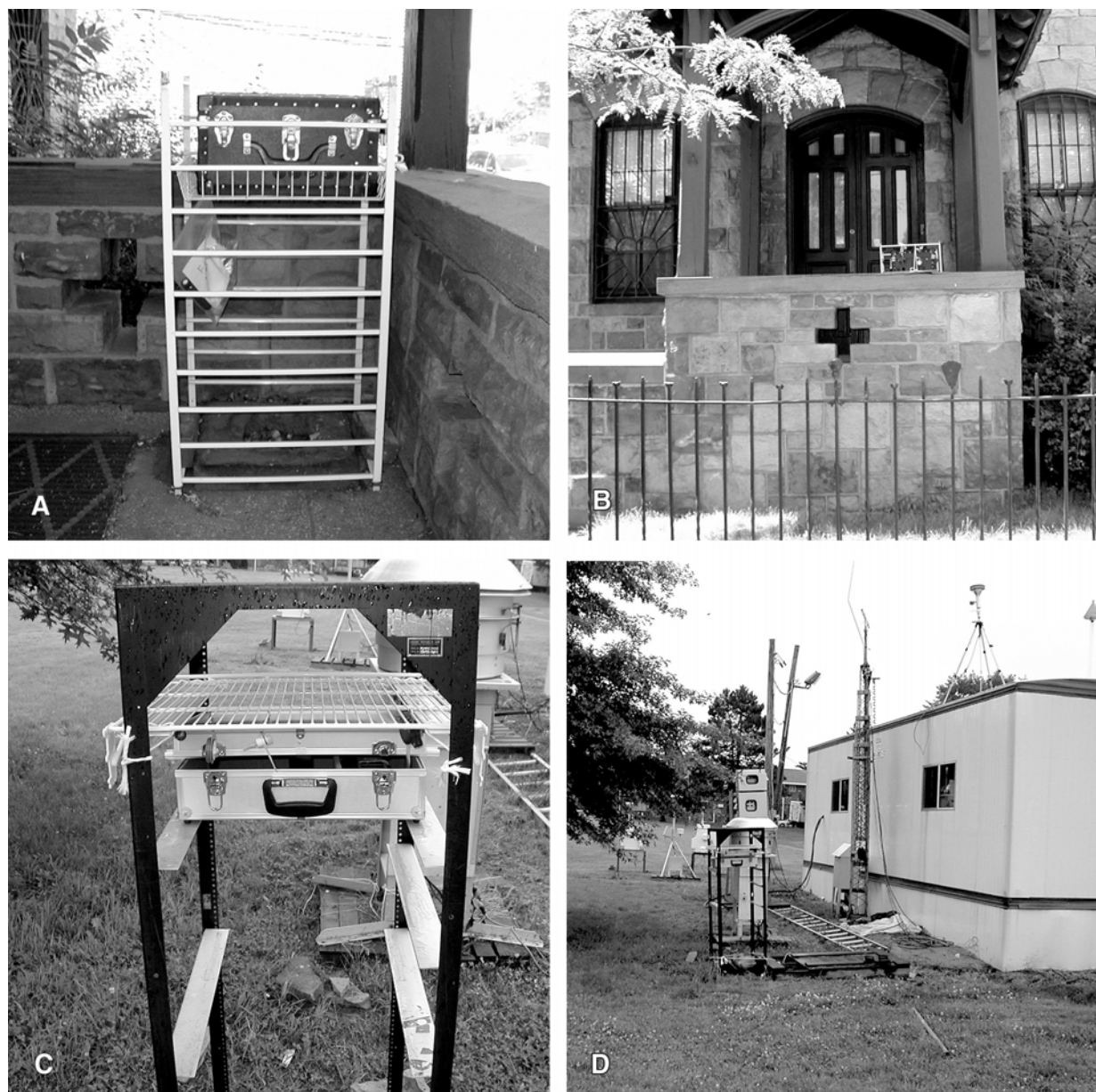


Figure 3. The fixed ambient monitoring sites. (A) Sampling setup and pumps at (B) Sacred Heart Church in Waterfront South. (C) Sampling setup and pumps at (D) NJDEP sampling cabin in Copewood–Davis.

Inc.). The VOCs and aldehydes were collected using passive samplers — organic vapor monitor (OVM) badges (model 3500, 3M) for the VOCs (3M 1993; Weisel et al. 2005) and passive aldehydes and ketones sampler (PAKS) cartridges for the aldehydes (Zhang et al. 2000). The $PM_{2.5}$ filters were also used to collect particle-phase PAHs; gaseous PAHs were collected on two polyurethane foam (PUF) sorbent sample tubes connected in series downstream of the filters. The sampling duration was 24 hr.

Ambient samples for Waterfront South and Copewood–Davis were collected simultaneously at our fixed monitoring

sites in the two neighborhoods. The fixed monitors were deployed on 6-ft sampling racks that were held down securely with ropes and weighting bricks in order to withstand high winds. A plastic covering fastened to the top of each rack provided a barrier against rain and snow. An SKC pump was secured in a covered box, and tubing was connected from the pump to the sampling head through a hole in the side of the box. The OVMS, PAKS cartridges, and ambient sampling head were connected to wire bars on the rack just under the plastic cover protecting the samplers from weather.

The same types of samplers were also placed in the personal air sampling backpacks (Figure 2). An SKC or BGI personal sampling pump (BGI Inc.) was placed inside the backpack, and the various samplers (an OVM badge, PM sampling head with PUF cartridges, and PAKS cartridge) were attached to the shoulder straps of the pack so that the sampling inlets were as close as possible to the subject's breathing zone. The weight of the sampling pack was about 5 lb and was accepted for use by all subjects. During each sampling day, the backpacks were prepared ahead of time in the NJDEP trailer and readied for deployment. This reduced down-time and inconvenience to subjects.

Before the start of the main study, in July 2004, the sampling protocols, performance of the sampling equipment, and analytic methods were evaluated. Details of the sampling and analytic methods, final sampling and analytic conditions, and results of quality assurance and quality control (QA–QC) for the sampling and analytic methods are presented below.

Measurement of PM_{2.5}

PM_{2.5} was collected using an SKC Leland Legacy sampling pump with an SKC PM_{2.5} impactor housing and a 37-mm Teflon filter (SKC Inc.). The flow stability, noise, and battery life of the pump were evaluated at 4 and 10 L/min for a sampling period of 24 hr. Before each sampling trip, the pump was calibrated and tested for leaks. The flow rate was measured using a Dry-Cal meter (TSI Inc.) before and after each sampling period. The SKC pump worked well when operated at 4 L/min for 24 hr, with a flow variation of < 10%. A greater flow variation (< 20%) was observed when the pump was operated at 10 L/min. The pump stopped running during some sampling periods because of the large pressure drop when operating at 10 L/min. For these reasons, a flow rate of 9 L/min was used for ambient air sampling. The PM_{2.5} impactor, which was designed for a flow rate of 10 L/min, was modified to ensure a proper cut-size at 9 L/min. In addition, the pump was noisy when operated at ~10 L/min. The noise level was reduced when the pump was placed in a pouch specially designed for it but was still too high to be suitable for personal sampling. The pump was therefore operated at 4 L/min for personal sampling. A PM_{2.5} impactor designed for operation at 4 L/min was used for personal air sampling. Before the field study, two tests were conducted in the laboratory to compare the PM_{2.5} concentrations obtained by means of the SKC sampling system with those obtained by means of the U.S. EPA's reference sampling method. The measured differences in PM_{2.5} concentrations were less than 25%.

After samples were collected, their PM_{2.5} mass was determined gravimetrically following the protocols of the U.S. EPA's Quality Assurance Handbook, Vol. II, Part II (U.S. EPA 2008). Samples with more than 4.8 hr of missing sampling time (i.e., ~20% of the 24-hr sampling period) were considered invalid and were excluded from analysis. The EOHSI weighing room has an independent climate-control system in which the temperature is set at $20 \pm 1^\circ\text{C}$ and relative humidity is set at $35 \pm 5\%$. Filters were conditioned in the weighing room for at least 24 hr at this temperature and relative humidity before weighing. A digital analytic microbalance (model MT5, Mettler Toledo) was used to weigh each filter before and after sampling. (The balance has a sensitivity of 1 μg .) If the difference between the two values was greater than 5 μg , the filters were reweighed. The tolerance for triplicate weighing of filters was $\pm 5 \mu\text{g}$ between any two values. Lab and field blank samples were tested for QA–QC purposes. Typically, these filter weights were less than $\pm 10 \mu\text{g}$. The average PM_{2.5} mass collected was between 100 to 400 μg ; the variability in PM_{2.5} mass concentrations contributed from field blanks was thus less than 10%.

Measurement of VOCs

OVM badges were used to collect personal and ambient VOCs. Our target compounds included 1,3-butadiene, MTBE, hexane, chloroform, carbon tetrachloride, benzene, toluene, ethylbenzene, *m*- & *p*-xylenes, *o*-xylene, and styrene. The analytic procedures for measuring the target VOCs collected using badges have been previously described by Chung and colleagues (1999) and Weisel and colleagues (2005). To summarize, the charcoal pad from each sample was removed from the badge and placed into a clean, labeled 2-mL amber glass vial. One mL of acetone–carbon disulfide mixture was added to the vial and sonicated for 45 min in an ice-water bath. After sonication, a 200- μL aliquot of the extract was pipetted into another amber glass vial with a 250- μL conical glass insert containing 10 μL of 200- $\mu\text{g}/\text{mL}$ internal standard solution (deuterated toluene and *d*₈-toluene). A 1- μL sample extract was then analyzed by gas chromatography–mass spectroscopy (GC–MS) (model 6890 GC and 5973 MSD, HP). The analytic column was a Restek Rtx-624 60 m \times 0.25 mm with 1- μm thickness. The GC oven temperature was held at 40°C for 8 min, raised $8^\circ\text{C}/\text{min}$ to 200°C , and then held at 200°C for 2 min. The injector temperature was 180°C , and the carrier-gas flow rate was 1.5 mL/min.

The target compounds were quantified using a calibration curve consisting of six levels of VOC calibration standards prepared freshly from certified standard solutions.

The calibration curve and concentration calculations were based on the ratio of the response of each target compound to that of the deuterated toluene. Good linearity and excellent precision were obtained for all the compounds tested. The coefficient of determination (R^2) of each calibration curve was greater than 0.99, and the relative SD of the seven repeated injections of a midlevel standard was less than 7% for all compounds except 1,3-butadiene (< 20%).

The extraction efficiency for each VOC was evaluated at the beginning of the study and determined following procedures recommended by 3M. A known amount of the target-compounds mixture was spiked onto three blank OVM badges, and the capped badges were equilibrated at room temperature for 24 hr before extraction and analysis. The extraction efficiencies obtained for each VOC from this evaluation were similar (% difference < 10%) to those reported by 3M. The efficiencies reported by 3M were therefore used for calculation in the study.

Three types of blank samples (i.e., the lab and field blanks for the OVM badges as well as solvent blanks) were examined for QA–QC purposes. Difficulties were encountered in analyzing hexane because of a high solvent background. The problem was resolved after switching the suppliers of the solvent (acetone, Spectral grade, #AX0110) to EM Science (from J.T. Baker) and of the carbon disulfide to Aldrich Chemical (42464-1) (from Fischer Scientific). Lab blank VOC concentrations were typically less than 5 ng/badge. To control for the potential variability caused by these background VOCs from the badges, one field blank was deployed during each field sampling trip; the results are shown in Table 4. Because there were batch-to-batch variations in blank VOC concentrations, all the samples collected from a given sampling trip were processed together to minimize the variability associated with sample-processing steps. In addition, the mean of each batch was used for blank subtraction.

Method detection limits (MDLs) were determined as three times the SD of the field blanks. For compounds that were not detected in the field blanks, the MDL was calculated as three times the SD of seven replicate injections of a low-level standard. Except for 1,3-butadiene and hexane, MDLs for the badges ranged from 0.1 to 1 $\mu\text{g}/\text{m}^3$ for a 24-hr sampling duration (Table 4).

Measurement of Aldehydes

PAKS cartridges (Herrington et al. 2005) were used to collect the four aldehydes (i.e., formaldehyde, acetaldehyde, propionaldehyde, and acrolein). Sampler preparation and analysis were performed in our laboratories at EOHSI–UMDNJ. The original version of this method was used successfully and described in the RIOPA study

(Weisel et al. 2005). In summary, a PAKS cartridge is a diffusive tube-type sampler that uses a dansylhydrazine (DNSH)-coated silica-based bonded C_{18} sorbent to passively collect airborne aldehydes. The aldehydes react with the DNSH to form their corresponding hydrazone derivatives, which are retained on the sorbent. The derivatives are subsequently extracted and quantified using a high-performance liquid chromatography (HPLC)–fluorescence technique. In the current study we used an improved version of this method, described below.

Our analytic instrument included an HPLC system (Waters 600E System Controller, Waters 717 Autosampler, Waters 470 Programmable Fluorescence Detector, and Waters Nova-Pak C_{18} column [3.9×300 mm, 60 Å, 4 μm]) and a guard cartridge (Nova-Pak, 4 μm , 60 Å, C_{18} Guard-Pak, Supelco); HPLC analytic conditions are described in Table 5. Aldehyde–DNSH derivatives were identified and quantified using their corresponding carbonyl–DNSH derivatives (standards), which were prepared *in situ* by placing PAKS cartridges in a dynamic gas-calibration system containing known concentrations of aldehydes for 12 hr. The PAKS cartridges were then extracted, and a series of calibration standards (eight levels) were generated by diluting the first standard with acetonitrile. The R^2 of the calibration standard curves was greater than 0.99 for all four aldehydes.

At the beginning of the study, elevated concentrations of formaldehyde and acetaldehyde were detected in the lab blanks. The cause of this problem was investigated, and it was found that the elevated concentrations of formaldehyde were caused primarily by the hydroquinone added to the coating solution to improve the reaction efficiency of acrolein. The hydroquinone was therefore removed from the coating solution before commencing the main study, in summer 2004. Blank concentrations for formaldehyde and acetaldehyde dropped from an average of ~ 1.12 and 0.84 $\mu\text{g}/\text{cartridge}$ to an average of ~ 0.17 and 0.54 $\mu\text{g}/\text{cartridge}$, respectively. Further investigation revealed that the acetic acid used in the PAKS coating solution also contributed a fair amount of acetaldehyde to the PAKS lab blanks. This was caused by the fact that the acetic acid solution contained acetaldehyde residues from the manufacturing process. Citric acid was therefore used instead of acetic acid, and average acetaldehyde concentrations in the lab blanks were further reduced to 0.24 $\mu\text{g}/\text{cartridge}$ (Table 4). The PAKS method was put into use in January 2005 and was used throughout the remainder of the study (i.e., until July 2006). Background aldehyde concentrations were checked regularly throughout the field study on solvent, lab, and field blank cartridges to ensure that there had been no contamination during the preparation or storage of the cartridges.

Table 4. Method Detection Limits and Field Blanks

	MDL ^a	Field Blanks	
		Average	SD
Aldehydes	µg/m ³	<i>N</i> = 74 (µg/cartridge)	
Formaldehyde	0.53	0.17	0.089
Acetaldehyde	0.28	0.24	0.147
Acrolein	1.06	0.013	0.015
Propionaldehyde	0.46	0.005	0.009
VOCs	µg/m ³	<i>N</i> = 90 (µg/sample)	
1,3-Butadiene	1.97	0.018	0.010
MTBE	0.28	0.001	0.007
Hexane	1.69	0.030	0.110
Chloroform	0.05	0.001	0.001
Carbon tetrachloride	0.06	0.000	0.001
Benzene	0.45	0.017	0.062
Toluene	0.81	0.000	0.020
Ethyl benzene	0.22	0.006	0.016
<i>m</i> - & <i>p</i> -Xylenes	0.53	0.016	0.028
<i>o</i> -Xylene	0.18	0.005	0.009
Styrene	0.20	0.002	0.004
PAHs	ng/m ³	<i>N</i> = 89 (ng/filter or PUF)	
Naphthalene	0.95	3.29	10.8
Acenaphthylene	0.36	0.79	0.7
Acenaphthene	0.10	0.23	2.1
Fluorene	0.44	2.06	9.7
Phenanthrene	0.77	2.45	8.8
Anthracene	0.03	0.02	4.6
Fluoranthene	0.16	0.59	9.2
Pyrene	0.12	0.24	2.4
Benzo[<i>a</i>]anthracene	0.03	0.04	0.7
Chrysene	0.03	0.03	2.6
Benzo[<i>b</i>]fluoranthene	0.03	0.02	0.2
Benzo[<i>k</i>]fluoranthene	0.01	0.01	0.2
Benzo[<i>a</i>]pyrene	0.06	0.04	0.2
Indeno[1,2,3- <i>cd</i>]pyrene	0.01	0.00	0.2
Benzo[<i>ghi</i>]perylene	0.00	0.00	0.2
Dibenzo[<i>a,h</i>]anthracene	0.02	0.01	0.2
Fine PM	µg/m ³	<i>N</i> = 77 (µg/sample)	
PM _{2.5}	2.0	−3	9

^a Method detection limits (MDLs) were estimated as 3 times the SD of the field blank for PM_{2.5}, VOCs, and PAHs. A sampling time of 24 hr at 10 L/min was used to estimate MDLs for PM_{2.5} and PAHs. For aldehydes, MDLs were derived from the SD of 7 repeat analyses of a low-concentration calibration standard.

Table 5. HPLC Analytic Conditions for the Measurement of Aldehydes

	% A (pH 7.80): Water–Acetonitrile– Tetrahydrofuran 80/10/10 v/v with 0.68 g/L KH ₂ PO ₄ and 3.48 g/L K ₂ HPO ₄	% B (pH 8.70): Water–Acetonitrile– Tetrahydrofuran 40/30/30 v/v with 0.68 g/L KH ₂ PO ₄ and 3.48 g/L K ₂ HPO ₄
0 min	100	0
30 min	70	30
60 min	60	40
80 min	60	40
85 min	100	0
Flow rate	1 mL/min	
Injection volume	20 μ L	
Detector	Excitation wavelength 250 nm	Emission wavelength 525 nm

Given these facts — and the potential for batch-to-batch variability caused by other parameters (such as variations in sampling temperature or sample storage time) — PAKS field blanks were deployed during every field sampling trip, and all samples and blanks collected on the same trip were processed together. Sample concentrations were corrected for the field blank concentrations obtained on the same sampling day in order to account for any variability in the field blank concentrations. The noise in the PAKS method was thereby reduced to the analytic noise. The PAKS MDLs were defined as three times the SD of seven repeat analyses of a low concentration standard (Table 4).

Measurement of PAHs

Particulate PAHs were collected on a 37-mm Teflon filter, and gaseous PAHs were collected using two PUF cartridges connected in series. The PUF holders and cartridges were house-made, 2 cm in outer diameter \times 10 cm in length. Before sample collection, the PUFs were cleaned using the U.S. EPA's TO-13 method. The sampling flow rate was 4 L/min for personal sampling and \sim 9 L/min for ambient sampling.

The target compounds were naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benzo[a]anthracene, chrysene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[a]pyrene, indeno[1,2,3-*cd*]pyrene, benzo[ghi]perylene, and dibenzo[a,h]anthracene. After sample collection, the filters and PUFs were Soxhlet-extracted with dichloromethane for 16 hr. Before extraction, deuterated PAHs (including naphthalene-*d*₈, phenanthrene-*d*₁₀, pyrene-*d*₁₀, and benzo[a]pyrene-*d*₁₂) were added as surrogates for monitoring the loss of PAHs during sample processing. The extract was then concentrated to 50–100 μ L under a gentle nitrogen stream. After spiking with a known amount of acenaphene-*d*₁₀ and anthracene-*d*₁₀ as internal

standards, one μ L of the sample extract was analyzed for PAHs using a Varian 3900 GC with a Saturn 2000 MS ion-trap detector and MS workstation software. The analytic column used for PAH separation was a VF-5ms (30 m \times 0.25 mm with a film thickness of 0.25 μ m). The analytic conditions were optimized to achieve the best separation and sensitivity. In summary, the GC oven was held at 45°C for 1 min, raised 20°C/min to 150°C, 5°C/min to 200°C, and 20°C/min to 300°C, and then held at 300°C for 5 min. The injector temperature was 300°C, and the injection volume was 1 μ L in splitless-injection mode. The carrier-gas flow rate was 1.2 mL/min. All the target compounds were well resolved under these conditions. Seven levels of PAH calibration standards were used for the calibration curves, which were constructed using the ratio of the responses between the target PAHs and the internal standards; the resulting *R*²'s were greater than 0.995. The PAH concentrations and surrogate recoveries were quantified based on the calibration curves. The PAH concentrations were corrected with the recoveries of the surrogates. These took into account the loss of PAHs during sample processing. In accordance with the physicochemical properties of the various PAHs, the recovery of naphthalene-*d*₈ was applied to correct the concentrations of naphthalene, acenaphthylene, acenaphthene, and fluorene; the recovery of phenanthrene-*d*₁₀ was used to correct the concentrations of phenanthrene and anthracene; the recovery of pyrene-*d*₁₀ was used to correct the concentrations of pyrene, fluoranthene, benzo[a]anthracene, and chrysene; and the recovery of benzo[a]pyrene-*d*₁₂ was used to correct the concentrations of benzo[a]pyrene and the rest of the compounds. The precision of the instrument was determined by performing seven repeat analyses of a midlevel calibration standard. The SD of these seven injections was < 25%. The percent recovery for each of the surrogates was 63.3% \pm 38.9% for naphthalene-*d*₈, 88.2% \pm 29.6% for phenanthrene-*d*₁₀, 66.2% \pm 28.4% for pyrene-*d*₁₀, and

54.7% \pm 33.2% for benzo[a]pyrene- d_{12} . Samples with average recoveries of < 30% were excluded from the analysis.

Three types of blank samples (i.e. solvent blanks, and lab and field blanks of the filter and PUF) were examined. In general, no PAHs were detected in the solvent blank or lab blanks. Five- to seven-ring PAHs were not detected in either the filter or PUF field blanks. A typical naphthalene level of \sim 30 ng, equivalent to \sim 2 ng/m³ naphthalene for sampling at 10 L/min for 24 hr, was detected in both filter and PUF field blanks, and less than 10 ng (equivalent to < 0.7 ng/m³ of PAHs in a sampling volume of 10 m³) of 2- to 4-ring PAHs was detected in the PUF blanks. The MDLs were derived from the field blanks collected over the course of the study (Table 4).

Nicotine Measurement for ETS Exposure Evaluation

Many smokers and smoking households were found in Waterfront South and Copewood–Davis during the first year of the study. Although only people living in nonsmoking households were recruited for the study, it was impossible to control for ETS exposures in the general neighborhood or among the subjects' friends. Given the concern that ETS can interfere with the evaluation of personal exposure to local sources of air toxics, measurements of nicotine in personal air samples were added to the study.

Because nicotine is a semi-VOC, it can be collected by the same sampling system used for the collection of gas and particle PAHs (Office of Population Censuses and Surveys, Social Survey Division 1997). Laboratory tests were conducted to evaluate the recovery of nicotine using the sample processing and analysis protocols developed for our PAH samples. For samples that contain ETS, detection of nicotine is possible in a larger volume of extract (hundreds of μ L) because nicotine is present at a much higher concentration (\sim 100 μ g/m³) than PAH concentrations (Sterling 1996; Nebot et al. 2005). However, to analyze for trace-level PAHs (< ng/m³) (Naumova et al. 2002; Marr et al. 2004; Ohura et al. 2004), the sample extract needs to be cleaned and concentrated to a small volume (10 to 50 μ L). Given these considerations, PUF sample extracts were first concentrated to 500 μ L for nicotine analysis. After nicotine analysis, the extract was cleaned using an aluminum column and further concentrated to 50 μ L for PAH analysis. Filter samples were cleaner than PUF samples and did not require aluminum-column cleaning. The filter-sample extracts were therefore concentrated to 100 μ L, and nicotine and PAH were analyzed simultaneously using GC–MS.

Nicotine was successfully separated from PAHs and deuterated PAHs using the GC–MS protocol developed for PAH analysis. The analytic detection limit (three times the signal-to-noise ratio) for nicotine was 0.5 ng, equivalent to

43.4 ng/m³ in air. Nicotine is usually present at μ g/m³ concentrations in ambient air, suggesting that our analytic method was sufficiently sensitive to detect nicotine in personal air samples. The recovery of nicotine from both filter and PUF samples was > 80%. For a subset of samples, two PUF cartridges connected in series were extracted and analyzed separately to determine the potential breakthrough of nicotine during the sampling process. A subset of ambient air samples was also analyzed for nicotine to determine the background nicotine concentrations in each neighborhood. Nicotine concentrations were determined in 234 personal samples using the PAH sample extracts. The measurements of nicotine were used in analysis to control for the interference of ETS exposure on personal exposure to ambient sources of air toxics.

SPATIAL VARIATION STUDY

Given the prevalence of stationary and mobile sources of air toxics in and near Waterfront South, most residents of the neighborhood actually live within 200 m of one or more sources. To judge from emissions inventories, moreover, the distribution of the air toxics released from these sources was expected to be uneven across the neighborhood. During the main exposure study, however, only a single fixed monitoring site was operated in each neighborhood; the one in Waterfront South was not able to capture the spatial variation in the neighborhood's air toxics and, as a consequence, might have underestimated the impact of individual sources on ambient concentrations of air toxics — which could have led to mischaracterization of individuals' personal exposures. Given these concerns, a spatial variation study consisting of three saturation-sampling campaigns was conducted in 2005 to identify the spatial variations in the air toxics and better quantify the impact of local sources of air toxics on personal exposures. Information obtained from these campaigns was also intended to help identify significant local sources of air toxics in both neighborhoods and to help in developing effective strategies to control the principal sources of concern.

Saturation-Sampling Sites

The locations of industrial facilities and subjects' houses in the two neighborhoods are shown in Figure 1. Locations that had more than one subject living in them are only shown once in the figure. Based on the location of stationary sources, main roads, and subjects' homes, it was determined that grid-based sampling would be appropriate to capture the spatial variation of air toxics in Waterfront South. In Copewood–Davis there are no nearby industrial facilities (< 1000 m), but the neighborhood is flanked by two busy streets, NJ-168 and Haddon Avenue (Figure 1). There is

also a train station with a parking lot. We therefore attempted to select sampling sites that could capture mobile sources and the spatial variation in air toxic concentrations in Copewood–Davis.

Sample accessibility and sampler security were also considered in the final selection of sampling sites, and two site visits were made to the area.

A total of 22 sampling sites were selected in Waterfront South in order to capture sufficient variability in relation to the proximity of local sources, especially the major industrial sources (Figure 1). Most of the sampling sites were located at street intersections. The distance between sites was approximately 150 to 250 m. Some sites were located within ~50 m of selected stationary sources (i.e., industrial facilities) in order to examine the maximum impact of the sources on local ambient concentrations of air toxics. Sites W12 and W15, for example, were located close to Colonial Processing (a manufacturer of welding and soldering equipment), site W3 was near I-676 and next to Steve's Auto Parts, site W19 was near Pinto Brothers Recycling (a paperboard mill) and a railway line, and site W20 was at the intersection of Winslow and South 4th streets, where heavy traffic was observed during site visits.

A total of 16 sampling sites were selected in Copewood–Davis. The number of samplers per unit of study area was similar for Waterfront South ($28/\text{km}^2$) and Copewood–Davis ($27/\text{km}^2$), which allowed direct comparison of the spatial variation between the two neighborhoods. Some sampling sites were along the main roads, and one (site C16) was in the train-station parking lot (near the southeast corner of Copewood–Davis) (Figure 1). These sites were selected to capture spatial variation in the air pollutants generated by motor vehicles. Most of the study subjects lived in the southern half of Copewood–Davis. The rest of the sampling sites were therefore located in the southern half, too, in order to enhance the utility of the collected data in evaluating the personal exposure characterizations made in the main study.

Saturation-Sampling Campaigns

The three saturation-sampling campaigns were conducted in 2005, two in summer (~48-hr samples from July 20 to 22 and ~24-hr samples from August 17 to 18) and one in winter (~48-hr samples from December 20 to 22). The planned sampling period had been 72 hr in order to smooth out diurnal and temporal variations, but the sampling had to be terminated early because of a strong storm in the sampling periods.

Meteorologic data were obtained from the National Climatic Data Center's Meteorological Airfield Report database of surface weather observations. Hourly meteorologic

data — primarily wind speed, wind direction, ambient temperature, and relative humidity — were obtained from the Philadelphia International Airport weather station (Weather Bureau Army Navy identification number 13739), located about 13 km southwest of our sampling sites. The meteorologic data were extracted to match each individual 24- or 48-hr sampling period. Wind speed, temperature, and relative humidity values were averaged over each sampling period and are summarized in Table 6.

VOCs and aldehydes were measured at all 38 saturation-sampling sites. OVM badges and PAKS cartridges were used at all sites. The samplers were deployed on 6-ft sampling racks at the fixed monitoring sites and on street signs or along fences at the saturation-sampling sites, at a sampling height of about 4 to 6 ft. The samples were analyzed using the same methods as those presented above in the sections on VOC and aldehyde measurement.

STATISTICAL ANALYSIS AND MODELING APPLICATION

Data Processing and Validation

All data, including questionnaires and meteorologic records, were entered into a database. Sampling information was checked for QA against the original field sampling record by a second person. The validation criterion for $\text{PM}_{2.5}$ and PAH samples was that the change in flow rate from the beginning to the end of the sampling period had to be less than 15%. The collection time had to be longer than 20 hr (~80% of the target duration). Field and analytic comments, including comments about compliance with personal and ambient sampler protocols, were also used to identify invalid samples. Invalid data were flagged in the database and were not used in the final analyses.

GeoLogger GPS devices were used to track the activities of randomly selected subjects. A total of 60 such records were obtained from these subjects. Though the GPS device was not precise enough to track indoor activity, data from it showed that the majority of the tested subjects did wear their sampling pack when they traveled. These data were found to be useful in the model evaluations of personal exposures.

Statistical Analyses

Descriptive analyses were performed to characterize the distributions of measured personal and ambient concentrations and to compare personal and ambient concentrations. Scatter plots of personal versus ambient air concentrations were completed to provide qualitative insight into the influence of outdoor sources on personal exposures. Because the measurements obtained from the study were highly right-skewed, R 's were calculated to

Table 6. Meteorologic Conditions During the Three Saturation-Sampling Campaigns (2005)

Start Time	End Time	Wind Speed (m/s) (Average Minimum–Maximum)	Wind Direction (Blowing From)	Temperature (°C) (Average Minimum–Maximum)	Relative Humidity (%) (Average Minimum–Maximum)	Cloud Cover (Tenths Cloudy)
July 20 9:00 AM	July 22 10:00 AM	3.22 (0 to 6.17)	West (32%) Southwest (32%) Northwest (14%) Others (22%)	28 (22 to 33)	54.8 (33 to 82)	4.5/10
August 17 10:00 AM	August 18 1:00 PM	3.58 (2.06 to 5.66)	Northwest (29%) East (18%) Northeast (18%) West (14%) Others (19%)	26 (21 to 29)	52.5 (39 to 66)	3.7/10
December 20 10:00 AM	December 22 2:00 PM	4.41 (0 to 8.23)	West (51%) Northwest (30%) Others (19%)	−0.6 (−4.4 to 3.3)	46.2 (19 to 66)	5.4/10

quantify associations between personal and ambient concentrations of each compound; this approach can deal with variables that are measured on different interval scales in order to ensure that the correlations are not biased by outliers.

To address the study's objectives and specific aims, analyses using mixed-effects models were performed to formally test and estimate (1) differences in personal and ambient concentrations between Waterfront South and Copewood–Davis and (2) associations between personal exposures and ambient air pollution. The first set of analyses examined differences between locations, using a mixed model with location, day of the week (i.e., weekday or weekend day), and season (i.e., summer or winter) as predictors. A random effect for day accounted for correlations between paired measurements taken simultaneously at the two locations. Interactions between factors were included in the model if they were significant. The analyses were designed to compare the characteristics of air pollution between the hot spot (Waterfront South) and the urban reference area (Copewood–Davis). A second set of analyses examined associations between personal exposures and ambient concentrations, including interactions with season or day of the week, using a nested analysis of variance (hierarchical linear model). (Mixed-effects models control both possible covariates as fixed effects and intra-subject correlations.) Four different models were applied to the data in order (1) to control location, season, and day of the week; (2) to examine the personal versus

ambient associations for each location separately; (3) to examine the personal versus ambient associations for each season separately; and (4) to examine the personal versus ambient associations for weekdays and weekend days separately. All models treated sampling date, subject, and season within subject as random effects. Each model yielded a slope, reflecting the change in personal exposure per unit change in ambient concentration, and a *P* value and provided a predicted personal exposure concentration for each subject measurement based on parameters fitted by the model. An R^2 was obtained by regressing the measured exposure against the modeled exposure, which could be used to interpret how much of the variation in personal exposure could be explained by each model. The confounding effects of ETS exposure on associations between personal and ambient concentrations for all target compounds were evaluated by incorporating the measured nicotine concentrations into the regression analyses (for the subgroup with available nicotine measurements). Measurements from duplicate samples (see Sample Collection, Data Completeness, and QA–QC section, below) were not included in the data analysis. SAS version 9 was used to perform the analysis.

For concentrations below the MDL, half of the MDL was used as censored data so that the dataset could be used for statistical analysis. When more than 60% of the data points were below the MDL, only graphical or descriptive analyses were performed. Quantile–quantile plots and Shapiro–Wilk tests were used to examine the normality of

the concentrations for the target compounds before performing the statistical analyses described above. Except for ambient $PM_{2.5}$ concentrations, measurements for all the targeted compounds had many extreme right-skew concentrations. A log transformation was therefore made on the concentration data before using the mixed model in those cases. The distributions of all the measurements were approximately log-normal ($P > 0.05$). Examples of quantile–quantile plots for log-transformed ambient (except ambient $PM_{2.5}$) and personal data for $PM_{2.5}$, benzene, formaldehyde, and phenanthrene are presented in Figures 4 through 7. The results for location and other comparisons (except ambient $PM_{2.5}$) reported in the study, then, were all based on log-transformed data.

For the spatial variation study, descriptive analyses were performed to calculate the mean and SD of the VOCs and aldehydes obtained from all sampling sites during each sampling period. Given the small sample sizes and highly skewed data, the nonparametric Wilcoxon rank sum test was used to compare the spatial variability and the mean of the VOCs for Waterfront South and Copewood–Davis. To examine spatial variability, the differences in variation

between the two neighborhoods were tested by applying the Wilcoxon rank sum test to the absolute deviations of each measurement from the medians for each day and location. Spearman correlation analyses were also performed among all species to help provide information identifying potential sources of VOCs in the two neighborhoods.

Model Applications

An essential component of the study was using the IBEM–MENTOR system to estimate ambient and personal exposure concentrations of air toxics from ambient sources (Georgopoulos et al. 2005; Georgopoulos and Liroy 2006). Benzene, toluene, and formaldehyde were selected for modeling — benzene because it was a ubiquitous automobile air toxic, toluene because it was found to have both automobile and stationary source emissions in Waterfront South, and formaldehyde because of the very high local concentrations of it measured throughout the course of the

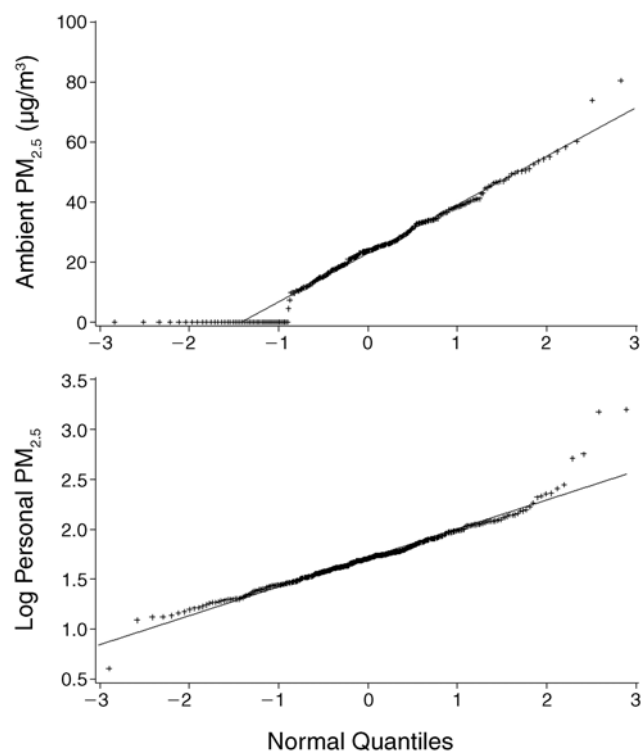


Figure 4. Quantile–quantile plots for (top) original ambient data and (bottom) log-transformed personal data for $PM_{2.5}$. Invalid and suspicious ambient data were replaced with zeroes (see points at bottom of figure).

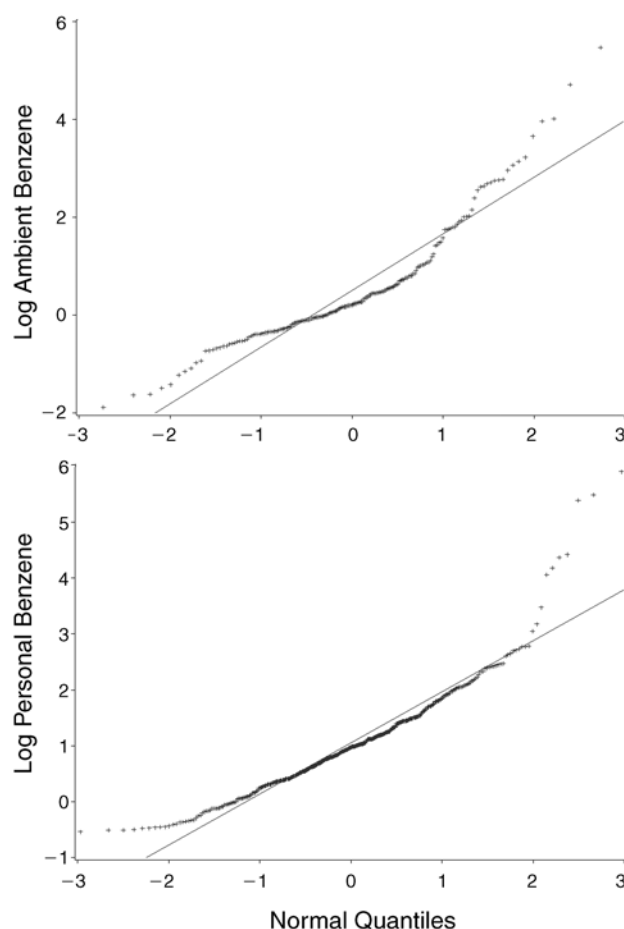


Figure 5. Quantile–quantile plots for (top) log-transformed ambient data and (bottom) log-transformed personal data for benzene.

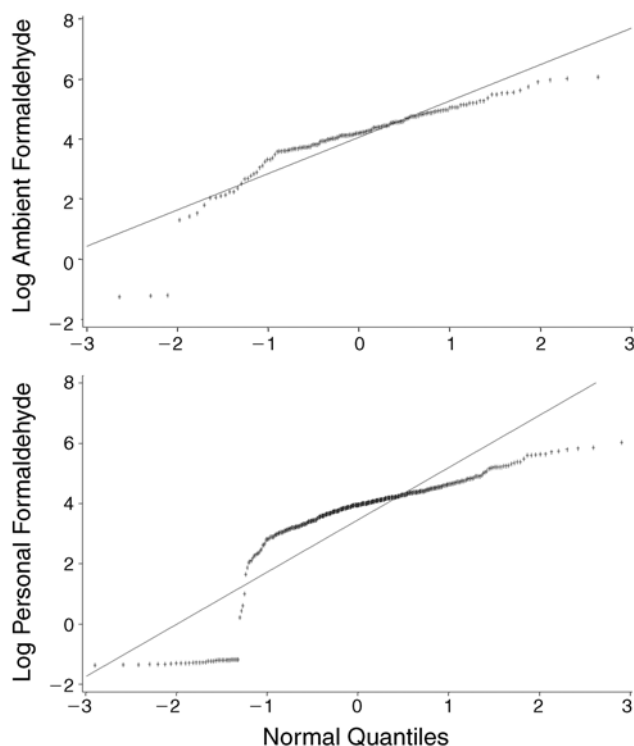


Figure 6. Quantile-quantile plots for (top) log-transformed ambient data and (bottom) log-transformed personal data for formaldehyde.

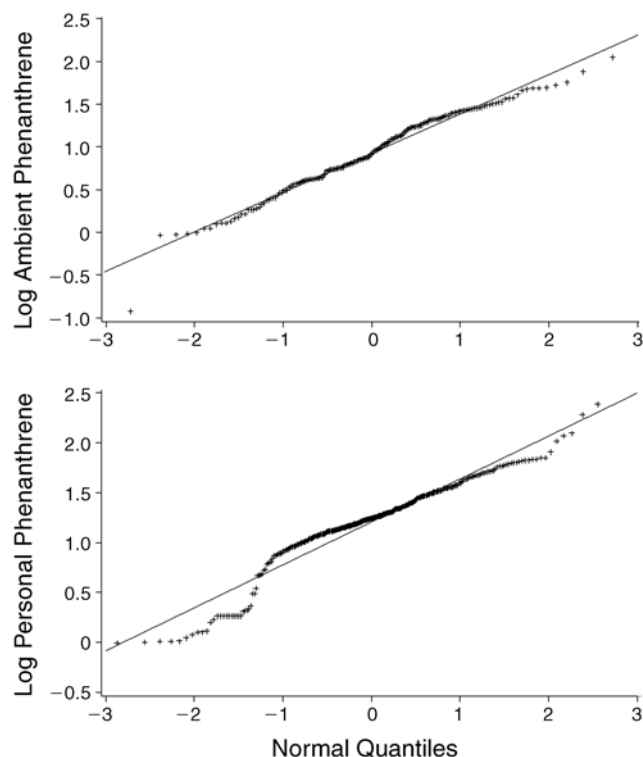


Figure 7. Quantile-quantile plots for (top) log-transformed ambient data and (bottom) log-transformed personal data for phenanthrene.

study. The resulting IBEM-MENTOR estimates were then compared with actual neighborhood measurements and personal air concentrations in order to assess the impact of local stationary and automobile sources on personal exposures and air pollution in the two neighborhoods.

Modeling of Ambient Air Concentrations

Ambient concentrations of benzene, toluene, and formaldehyde at the subjects' residences were estimated using the two widely used Gaussian plume atmospheric dispersion models: (1) the Industrial Source Complex Short Term Version 3 (ISCST3) (U.S. EPA 1995) and (2) the American Meteorological Society-U.S. EPA Regulatory Model (AERMOD) (Strum et al. 2004). Our source emissions data and local meteorologic data were used as inputs to these models for calculating emissions-based ambient concentrations of air toxics. Point, area, mobile nonroad, and mobile onroad emissions data were taken from the 2002 National Emissions Inventory (NEI) (U.S. EPA 2002) for benzene, toluene, and formaldehyde according to an emissions modeling domain that included both Philadelphia County, Pennsylvania, and Camden County, New Jersey. The domain was determined by joining the buffer zones of census tracts located within a 25-km radius of each sampled

home in the field study. Table 7 shows the annual total emissions amounts and percentages for each of the three air toxics and four source types in the two counties. Mobile (nonroad and onroad) sources accounted for 76% and 71% of total emissions for benzene and toluene, respectively; stationary (point and area) sources accounted for the rest. For formaldehyde, mobile sources accounted for 92% of total emissions.

Because mobile and area source emissions are reported as annual county totals in the NEI 2002 database, fine-scale (such as census-tract level) spatial allocation of these emissions was necessary to estimate impacts on a neighborhood. Because the four source types are also reported as annual county totals in the NEI 2002 database, temporal allocation of the emissions was necessary as well, to estimate hourly emission rates (and hence seasonal and day-of-the-week variations). The Emissions Modeling System for Hazardous Air Pollutants Version 3 (Strum et al. 2004) was used to process the NEI 2002 emissions data. The county-level emissions were spatially apportioned into the census tracts within each county by using spatial allocation factors. Mobile onroad emissions, for example, can be allocated using the geographic distribution of roadway miles. The

Table 7. Percentages of Annual Total Emissions^a

Pollutant Source Category	Emissions (%)	Emissions (Tons per Year)
Benzene		
Point	6	83.8
Area	18	237.8
Mobile nonroad	25	323.7
Mobile onroad	51	669.5
Toluene		
Point	6	238.3
Area	23	847.9
Mobile nonroad	20	735.0
Mobile onroad	51	1885.4
Formaldehyde		
Point	6	50.9
Area	2	13.9
Mobile nonroad	41	323.3
Mobile onroad	51	401.5

^a Including all sources inside 25-km-radius circles centered on homes sampled for the 2002 National Emissions Inventory in Philadelphia County, Pennsylvania; and Camden County, New Jersey.

geographic distributions of the emissions sources used in our modeling analyses are shown in Figures 8 through 10.

Meteorologic variables included wind speed, wind direction, stability category, and mixing height. Hourly surface meteorologic data were taken at the Philadelphia International Airport, the station closest to the study area. These data were preprocessed through PCRAMMET (Byun and Ching 1999) and AERMET (Strum et al. 2004) software to generate the meteorologic inputs for ISCST3 and AERMOD, respectively. Wind rose plots summarizing information about wind speeds and directions are shown in Figure 11 for the study's four seasonal and day-of-the-week sampling combinations (i.e., summer weekday, summer weekend day, winter weekday, and winter weekend day). Southwest or south winds dominated in summer; northwest or west winds dominated in winter. Mixing heights in winter were generally lower than in summer.

These meteorologic conditions were typically fairly uniform across the metropolitan area, except during frontal passages, and should therefore be generally representative of conditions in Camden.

Four atmospheric dispersion scenarios were modeled to estimate ambient concentrations of the three air toxics at the central sites and the subjects' residences:

- Scenario 1: ISCST3 predicted ambient concentrations for the two fixed monitoring sites (i.e., in Waterfront South and Copewood–Davis),
- Scenario 2: AERMOD predicted ambient concentrations for the two fixed monitoring sites,
- Scenario 3: ISCST3 predicted ambient concentrations for the subjects' residences, and
- Scenario 4: AERMOD predicted ambient concentrations for the subjects' residences.

All four scenarios estimated the ambient concentrations of each air toxic for time periods that corresponded to the actual field-measurement sampling times. In order to evaluate the performance of the models, the simulation results from the first two scenarios were compared with the actual measurements collected at the two fixed monitoring sites. A comparison between the first two and the last two sets of simulation results provided information on the adequacy of the ambient concentrations predicted at the fixed monitoring sites in representing actual ambient concentrations in the two neighborhoods.

For comparison with the ambient measurements collected at the two fixed monitoring sites, ISCST3 and AERMOD simulations were conducted, generating time-series profiles of estimated hourly average ambient concentrations for the corresponding 24-hr sampling periods. These 24 hourly estimates were then averaged to compare them with the corresponding 24-hr integrated ambient measurements. Because photochemical assessment monitoring station (PAMS) measurements were also available at the Copewood–Davis fixed monitoring site for the summers of 2004 to 2006, a search was conducted of the PAMS measurements to find sampling periods that matched those of the ambient measurements collected at the Copewood–Davis site. The search results revealed that there were in fact 17 matched sets of measurement periods for ambient concentrations of benzene and toluene. Cross comparisons were then conducted among the ISCST3 predictions, Copewood–Davis ambient measurements, and PAMS measurements for benzene and toluene in the 17 sets to check for consistency. Because the PAMS measurements had the same hourly resolution as the ISCST3 predictions, they were also used to further evaluate ISCST3's ability to reproduce the diurnal profiles of hourly ambient benzene and toluene measurements. For comparison with the Copewood–Davis measurements, the hourly PAMS measurements were averaged to produce data with matching temporal scales.

Modeling of Personal Air Concentrations

Calculated and measured outdoor concentrations were used individually with subjects' time–activity diaries as inputs to the MENTOR system to estimate personal exposures resulting from outdoor sources. The IBEM implementation of the MENTOR system was used to model the three air toxics. IBEM uses a person-oriented modeling formulation (i.e., driven by the attributes and activities of exposed real

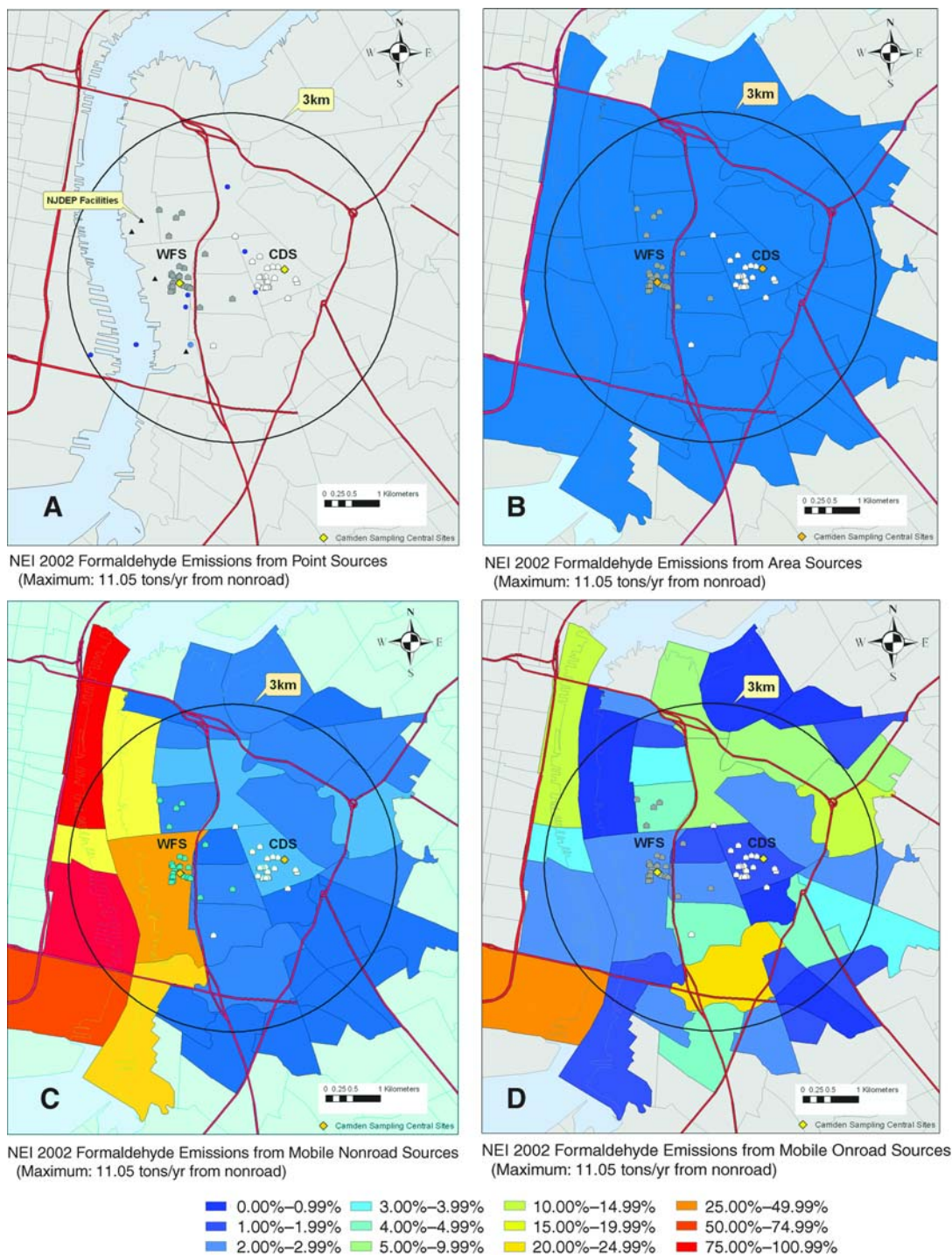


Figure 8. Annual formaldehyde emissions from all sources within a 3-km radius of the centroid of the sampling homes in Camden, showing the spatial distribution and strength of (A) point sources, (B) area sources, (C) mobile nonroad sources, and (D) mobile onroad sources. County-level NEI 2002 data were processed using the Emissions Modeling System for Hazardous Air Pollutants Version 3 (Strum et al. 2004); mobile and area sources were allocated per census tract. Emission rates were calculated as the maximum emission rate multiplied by the percentages shown. The percentages are represented by the various colors. Black triangles indicate local industrial facilities identified by the NJDEP. Blue circles indicate formaldehyde emission sources identified in NEI 2002. Gray pentagons indicate subject homes in Waterfront South. White pentagons indicate subject homes in Copewood–Davis. Yellow diamonds indicate the fixed monitoring sites in the two neighborhoods.

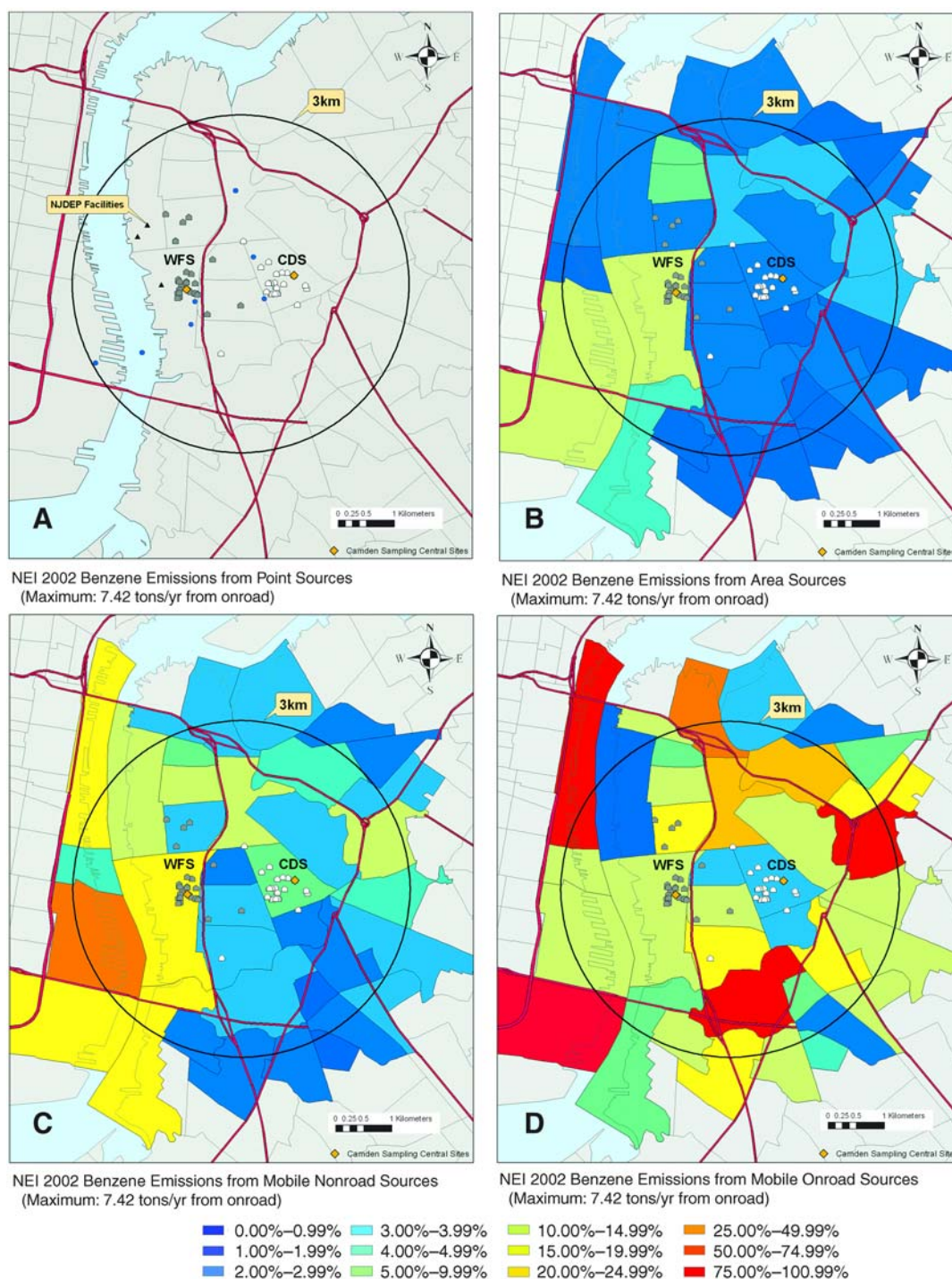


Figure 9. Annual benzene emissions from all sources within a 3-km radius of the centroid of the sampling homes in Camden, showing the spatial distribution and strength of (A) point sources, (B) area sources, (C) mobile nonroad sources, and (D) mobile onroad sources. County-level NEI 2002 data were processed using the Emissions Modeling System for Hazardous Air Pollutants Version 3 (Strum et al. 2004); mobile and area sources were allocated per census tract. Emission rates were calculated as the maximum emission rate multiplied by the percentages shown. The percentages are represented by the various colors. Black triangles indicate local industrial facilities identified by the NJDEP. Blue circles indicate formaldehyde emission sources identified in NEI 2002. Gray pentagons indicate subject homes in Waterfront South. White pentagons indicate subject homes in Copewood–Davis. Yellow diamonds indicate the fixed monitoring sites in the two neighborhoods.

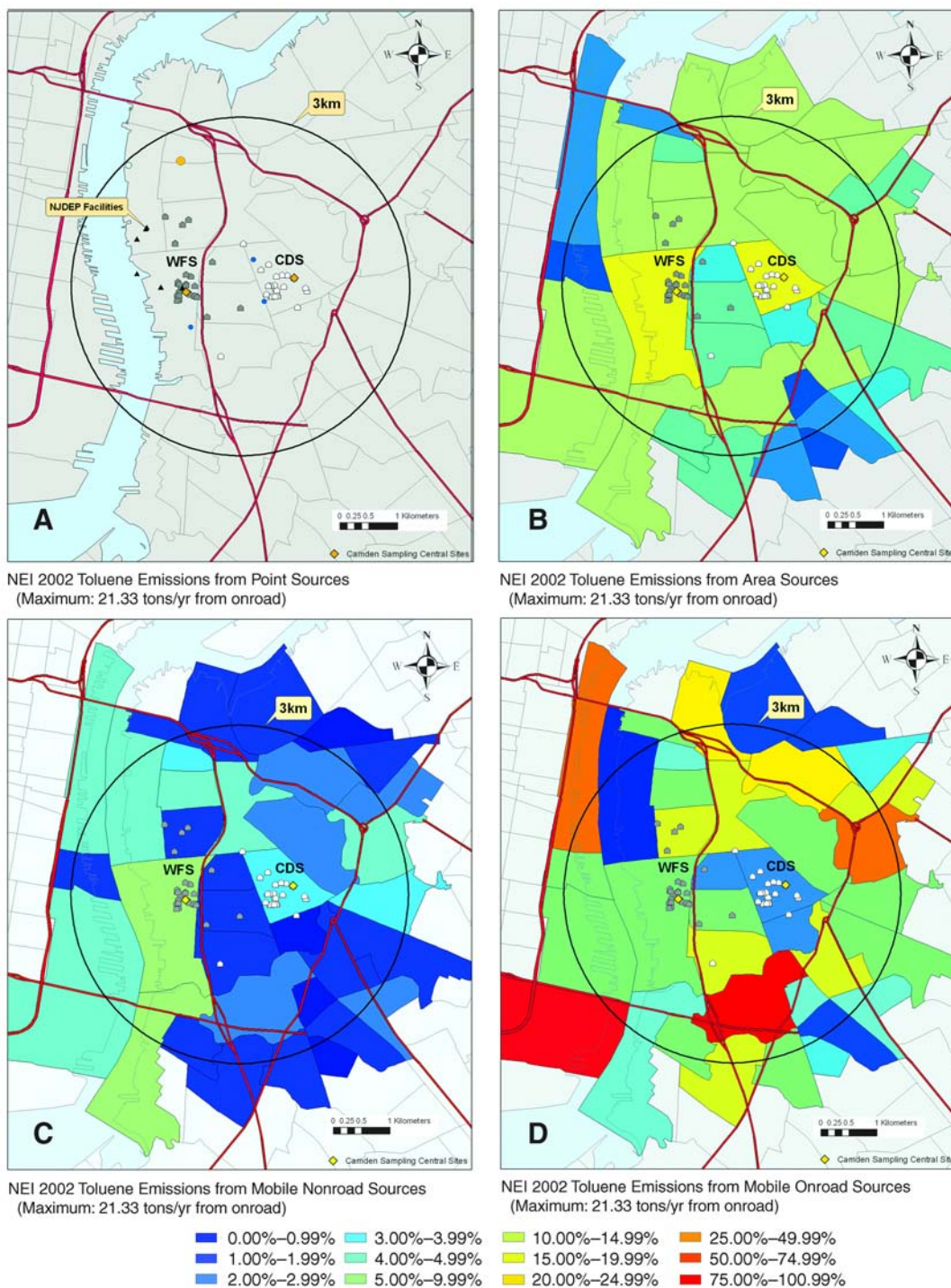


Figure 10. Annual toluene emissions from all sources within a 3-km radius of the centroid of the sampling homes in Camden, showing the spatial distribution and strength of (A) point sources, (B) area sources, (C) mobile nonroad sources, and (D) mobile onroad sources. County-level NEI 2002 data were processed using the Emissions Modeling System for Hazardous Air Pollutants Version 3 (Strum et al. 2004); mobile and area sources were allocated per census tract. Emission rates were calculated as the maximum emission rate multiplied by the percentages shown. The percentages are represented by the various colors. Black triangles indicate local industrial facilities identified by the NJDEP. Blue circles indicate formaldehyde emission sources identified in NEI 2002. Gray pentagons indicate subject homes in Waterfront South. White pentagons indicate subject homes in Copewood–Davis. Yellow diamonds indicate the fixed monitoring sites in the two neighborhoods.

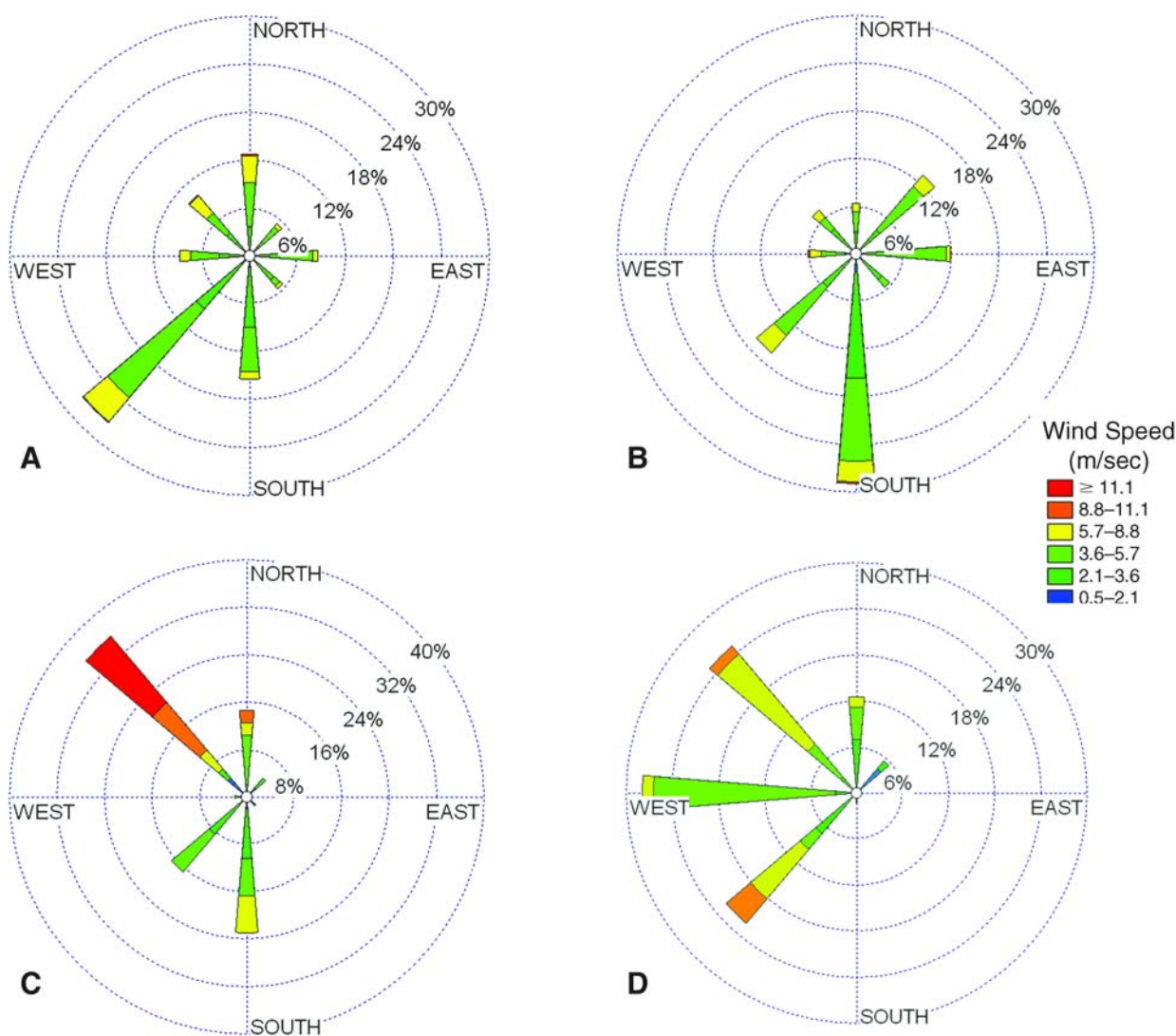


Figure 11. Wind rose plots summarizing wind speeds and directions for the study's four sampling combinations — (A) summer weekdays, (B) summer weekends, (C) winter weekdays, and (D) winter weekends — between April 2004 and July 2006. Spoke lengths indicate the frequency of wind from the given directions. Data were obtained from Philadelphia International Airport's meteorologic station. (Wind directions shown are the directions from which the wind was blowing.)

or virtual individuals). It uses information relevant to real individuals (such as demographics, housing characteristics, and time-activity information) to estimate personal exposures following the source-to-exposure modeling sequence, producing specific exposure and dose estimates for each individual. Variability among the IBEM exposure estimates was used to increase confidence in establishing the population's exposure and to improve the characterization of hot spots. In our study, only time-activity information (the amount of time spent in each microenvironment [Appendix G]) was used in the model.

In order to evaluate the model's performance, the modeled personal and ambient exposures were compared with the corresponding personal and outdoor measurements. The results for ambient modeling were used to identify the strengths and weaknesses of using NEI 2002 source emissions data for local-scale atmospheric dispersion modeling in characterizing hot spots. The results for personal exposure modeling were used to show the contribution of ambient sources to personal exposures as well as to identify data needs or gaps (e.g., indoor source contributions).

Our methods and modeling processes are described in detail in Appendix H.

Table 8. Demographic Data for Study Participants^a

	Waterfront South		Copewood–Davis	
	Adult	Child	Adult	Child
Number	37	17	34	19
Age (yr) ^b				
Mean	41	13	39	14
Minimum	22	9	20	8
Maximum	80	18	62	19
Gender				
Male	14	9	9	6
Female	23	8	25	13
Total	37	17	34	19
Cultural background ^c				
White	10	1	1	
African-American	18	10	20	18
American Indian				
Asian or Pacific islander		3		
Hispanic white	1			
Hispanic black	1	1		
Hispanic other	6	6	10	1
Total	36	21	31	19
Highest level of education completed				
No schooling completed or kindergarten only	1			
Primary or middle school	2	12	3	12
Some high school	5	4	7	6
High school graduate	14	1	9	1
Some college or technical school	5		7	
Undergraduate degree received	6		4	
Some graduate school				
Graduate degree received				
Total	33	17	30	19
Work status				
Adult working full time	6		9	
Adult working part time	11		1	
Student and working	1	1	2	
Student not working	1	16		16
Self-employed working at home or homemaker	3		4	
Out of work just now but usually employed	5		4	1
Retired	5		2	
Disabled or unable to work				
Child attending summer camp				2
Total	32	17	22	19

^a Missing values were not included.

^b Age was determined as of 2005.

^c Some subjects gave multiple answers to the question about cultural background.

EXPERIMENTAL RESULTS AND DISCUSSION

DEMOGRAPHIC INFORMATION AND SUBJECT RETENTION

In total, 107 subjects participated in our personal air sampling study; this met the recruitment goal of 100 subjects. Demographic data on the subjects (based on information collected from the baseline questionnaire, such as gender, age, ethnicity, education level, and work status) are summarized and presented in Table 8. Of the 107 subjects, 37 adults and 17 children were from the Waterfront South neighborhood, and 34 adults and 19 children were from Copewood–Davis. To evaluate the impact of local air pollution on personal exposures, we intentionally recruited subjects who spent most of their time (> 12 hr a day) in the area during the 24-hr sampling period. Of the 71 adult participants, 48 were women and 23 were men. According to census data, then, the study population did not fully represent the demographic and socioeconomic status of the entire population of the city of Camden.

As shown in Table 9, 80 of the 107 subjects completed all four sampling days, six completed three sampling days, 12 completed two sampling days, and nine completed only one, meaning that 93% of the planned personal measurements were made successfully. Relocation was the primary reason for discontinuing participation.

SAMPLE COLLECTION, DATA COMPLETENESS, AND QA–QC

Total numbers of personal and ambient samples collected are shown in Table 10a; valid regular samples (excluding duplicate and field blank samples) by location, season, and day of the week are shown in Table 10b. Field blank and duplicates were collected for QA–QC. No personal PM_{2.5} or PAH duplicate samples were collected, because of the burden on subjects of the sampling pump. A

field blank was collected on each sampling day; the percentages of field blanks collected for PM_{2.5}, VOCs, aldehydes, gas-phase PAHs, and particle-phase PAHs were 17%, 15%, 14%, 11%, and 8%, respectively. Duplicates were collected across the entire study period. The percentages of duplicates collected for PM_{2.5}, VOCs, aldehydes, gas-phase PAHs, and particle-phase PAHs were 20%, 12%, 13%, 17%, and 15%, respectively (Table 10a). The number of field blanks and duplicates collected exceeded the QA–QC objectives of 5% field blanks and 10% duplicates over the course of the study.

Personal and ambient samples were evenly distributed across the two sampling locations. The analyses reported here are based only on valid samples without duplicates or field blanks. In summary, the percent of valid samples for PM_{2.5}, VOCs, aldehydes, gas-phase PAHs, and particle-phase PAHs were 89%, 99%, 96%, 96%, and 92%, respectively (Tables 10a and b), thus achieving the QA–QC objectives of the study.

Method Detection Limits and Method Precision

As noted in the measurement method section, MDLs were defined as three times the SD of the field blanks (for PM_{2.5}, VOCs, and PAHs) or analytic detection limits (for aldehydes). The sampling volumes obtained from the 24-hr sampling were used for the calculation of MDLs; the MDLs are shown in Table 4.

Overall measurement precision (and hence reproducibility of results) for each compound was calculated as a coefficient of variation, given by the pooled SD (σ_{pooled}) divided by the mean value of the pairs. For paired data, $\sigma_{\text{pooled}} = [\sum d_i^2 / 2n]^{1/2}$, where d is the difference between pair i values and n is the number of pairs. The results are shown in Table 11 and discussed below.

Good reproducibility was observed for PM_{2.5}. The overall precision for PM_{2.5} was 14.6% ($N = 75$). Most of the VOCs measured had good reproducibility, too, except

Table 9. Summary of Subject Participation

	Number of Subjects		Total	
	Waterfront South	Copewood–Davis	Number of Subjects	Personal Exposure Measurements
Completed 4 sampling days	44	36	80	320
Completed 3 sampling days	2	4	6	18
Completed 2 sampling days	3	9	12	24
Completed 1 sampling days	5	4	9	9
Total	54	53	107	371

Table 10a. Total Numbers of Samples Collected for Each Type of Species

	Regular Sample	Duplicate	Field Blank	Duplicate (%)	Field Blank (%)	Valid Sample (%)
PM _{2.5}						
Valid	459	92	77	20	17	91
Invalid	46	21				
VOCs						
Valid	550	65	90	12	16	99
Invalid	3	1				
Aldehydes						
Valid	491	62	74	13	14	99
Invalid	6	1				
PAH gas phase						
Valid	461	79	52	17	11	97
Invalid	14	12				
PAH particle phase						
Valid	442	68	37	15	8	92
Invalid	41	6				

Table 10b. Total Numbers of Valid Regular Samples Collected in Waterfront South and Copewood–Davis for Each Type of Species by Location, Season, and Day of the Week^a

	Ambient				Personal			
	Summer		Winter		Summer		Winter	
	Weekday	Weekend	Weekday	Weekend	Weekday	Weekend	Weekday	Weekend
PM _{2.5}								
Waterfront South	18	18	16	15	48	48	38	37
Copewood–Davis	15	16	16	15	35	41	43	40
VOC								
Waterfront South	32	25	24	18	55	49	52	44
Copewood–Davis	27	22	21	16	44	42	40	39
Aldehydes								
Waterfront South	24	19	18	15	56	44	41	38
Copewood–Davis	24	18	18	14	42	38	41	41
PAHs (gas phase)								
Waterfront South	23	18	17	15	49	42	41	36
Copewood–Davis	17	14	20	13	43	34	40	39
PAHs (particle phase)								
Waterfront South	21	15	17	15	47	42	39	34
Copewood–Davis	16	12	19	13	39	34	40	39

^a The number of samples does not include duplicate or field blank samples.

for styrene. The overall precision was 11.2% ($N = 64$) for 1,3-butadiene, 6.4% ($N = 64$) for MTBE, 1.0% ($N = 62$) for hexane, 11.8% ($N = 63$) for chloroform, 5.6% ($N = 64$) for carbon tetrachloride, 18.4% ($N = 63$) for benzene, 5.7% ($N = 63$) for toluene, 14.4% ($N = 64$) for ethyl benzene, 11.7% ($N = 64$) for *m*- & *p*-xylenes, 11.4% ($N = 64$) for *o*-xylene, and 35.1% ($N = 64$) for styrene. It is worth noting

that 1,3-butadiene was below the MDL in 65% of samples; therefore, the “good” measurement precision was biased by the 0% difference from many duplicate samples with nondetectable concentrations of 1,3-butadiene. In the case of the aldehydes, overall precision was 24.6% ($N = 57$) for formaldehyde, 29.6% ($N = 55$) for acetaldehyde, 46.4% ($N = 56$) for acrolein, and 37.2% ($N = 58$) for propionaldehyde. Part of

Table 11. Measurement Precision^a and Percentage Above Method Detection Limit by Compound

	CV (%)	<i>N</i>	CV (%)	<i>N</i>	Detects ^b (%)	
PM _{2.5}	14.6	75			100	
Aldehydes						
Formaldehyde	24.6	57			90	
Acetaldehyde	29.6	55			80	
Acrolein	46.4	56			40	
Propionaldehyde	37.2	58			50	
VOCs						
1,3-Butadiene	11.2	64			56	
MTBE	6.4	64			99	
Hexane	1.0	62			74	
Chloroform	11.8	63			87	
Carbon tetrachloride	5.6	64			92	
Benzene	18.4	63			96	
Toluene	5.7	63			91	
Ethyl benzene	14.4	64			83	
<i>m</i> - & <i>p</i> -Xylenes	11.7	64			92	
<i>o</i> -Xylene	11.4	64			92	
Styrene	35.1	64			49	
	Gas		Particle		Gas	Particle
PAHs						
Naphthalene	33.0	63	26.0	55	90	41
Acenaphthylene	35.0	61	ND ^c		83	20
Acenaphthene	39.9	62	ND		86	24
Fluorene	36.1	63	ND		91	36
Phenanthrene	23.8	61	ND		89	48
Anthracene	27.4	63	ND		60	17
Fluoranthene	28.1	63	70.9	55	73	59
Pyrene	30.0	61	45.2	55	73	69
Benzo[<i>a</i>]anthracene	ND		37.7	53	20	48
Chrysene	ND		31.3	53	21	48
Benzo[<i>b</i>]fluoranthene	ND		37.6	53	4	52
Benzo[<i>k</i>]fluoranthene	ND		46.9	53	3	54
Benzo[<i>a</i>]pyrene	ND		28.3	54	4	45
Indeno[1,2,3- <i>cd</i>]pyrene	ND		48.8	54	3	27
Benzo[<i>ghi</i>]perylene	ND		45.1	55	3	23
Dibenzo[<i>a,h</i>]anthracene	ND		31.1	55	3	33

^a Values are the pooled coefficients of variation (CVs) of pairs of co-located duplicate field measurements expressed as percentages; *N* is the total number of pairs.

^b Percentage detects are overall samples above method detection limit.

^c ND indicates that CVs were not determined because more than half the values were below the detection limit.

the variability among the aldehydes was probably contributed by our having used liquid aldehyde standards for calibration, as explained earlier.

The overall precision for gas-phase PAHs was 33.0% ($N = 63$) for naphthalene, 35.0% ($N = 61$) for acenaphthylene, 39.9% ($N = 62$) for acenaphthene, 36.1% ($N = 63$) for fluorene, 23.8% ($N = 61$) for phenanthrene, 27.4% ($N = 63$) for anthracene, 28.1% ($N = 63$) for fluoranthene, and 30.0% ($N = 61$) for pyrene. The overall precision for particle-phase PAHs was 37.7% ($N = 53$) for benzo[*a*]anthracene, 31.3% ($N = 53$) for chrysene, 37.6% ($N = 53$) for benzo[*b*]fluoranthene, 46.9% ($N = 53$) for benzo[*k*]fluoranthene, 28.3% ($N = 54$) for benzo[*a*]pyrene, 48.8% ($N = 54$) for indeno[1,2,3-*cd*]pyrene, 45.1% ($N = 55$) for benzo[*ghi*]perylene, and 31.1% ($N = 55$) for dibenzo[*a,h*]anthracene. The greater variability observed for the particle-phase PAHs was caused by the low concentrations of these compounds. Many compounds with molecular weight > 252 were only ~20% above the MDL. Given the known sampling artifacts of the filter-PUF system for PAH measurement, the gas and particle concentrations for each individual PAH were combined in subsequent data analyses.

PERSONAL AND AMBIENT CONCENTRATIONS AND PERSONAL-AMBIENT ASSOCIATIONS

In the following sections, study results are presented by type of pollutants measured. Summaries of descriptive statistics for our personal and ambient measurements are presented and discussed first. Because interactions between location and day of the week or location and season were observed for some species, the concentrations for each species are then presented by day of the week and season for each location. Finally, the relationships between personal exposures and ambient concentrations are discussed and evaluated with respect to other studies, and the ambient contributions to personal total exposures are estimated from the collected data.

PM_{2.5}

Ambient PM_{2.5} Mass Concentration Ambient PM_{2.5} mass concentrations measured at the Waterfront South and Copewood-Davis fixed monitoring sites are summarized in Table 12a and shown by day of the week in Table 13 and Figure 12 and by season in Table 14 and Figure 13. The ambient PM_{2.5} concentration (mean \pm SD) measured in Waterfront South ($31.3 \pm 12.5 \mu\text{g}/\text{m}^3$) was significantly higher ($P = 0.003$) than that measured in Copewood-Davis ($25.3 \pm 11.9 \mu\text{g}/\text{m}^3$) (Table 12a). Our observations are consistent with findings from the NJDEP air toxics pilot study (2005), which showed that PM_{2.5} concentrations measured

with a continuous PM monitor on the roof of the Camden sewage treatment plant, located in Waterfront South, were higher than those measured at the Copewood-Davis monitoring site. Local industrial sources of PM_{2.5} identified by the NJDEP included several metal processing companies, a cement plant, and the Camden sewage treatment plant. In addition, the NJDEP estimated that 275 to 825 diesel trucks per day travel or idle in Waterfront South to serve the area's industrial and commercial operations. Thus, emissions from both types of sources probably contribute to the elevated concentrations of PM_{2.5} in the neighborhood. It is worth noting that some of the PM_{2.5} observed in Waterfront South is transported from Philadelphia, located ~32 km west of Camden. Average PM_{2.5} in Philadelphia is 11 to 14 $\mu\text{g}/\text{m}^3$ (U.S. EPA 2006c); the contribution from Philadelphia to the PM_{2.5} measured in our study would thus be expected to be less than 10 $\mu\text{g}/\text{m}^3$.

The PM_{2.5} concentrations measured in Waterfront South and Copewood-Davis were compared with those reported for other U.S. urban areas in order to evaluate their significance. In the RIOPA study (Weisel et al. 2005), PM_{2.5} was measured in Elizabeth, New Jersey; Houston, Texas; and Los Angeles, California. The mean ambient concentration of PM_{2.5} found in the RIOPA study was 15.5 $\mu\text{g}/\text{m}^3$. The mean concentrations found in Waterfront South and Copewood-Davis were 2.0 and 1.6 times higher than this, respectively. When compared with the U.S. national average PM_{2.5} concentration, which ranges from 11 to 18 $\mu\text{g}/\text{m}^3$, measured PM_{2.5} concentrations were 1.7 to 2.8 times higher in Waterfront South and 1.4 to 2.3 times higher in Copewood-Davis. Moreover, 23% of the PM_{2.5} measurements in Waterfront South and 10% in Copewood-Davis exceeded the national upper PM_{2.5} standard of 40 $\mu\text{g}/\text{m}^3$. Comparisons were also made using results from studies conducted in other locations. Lachenmyer and Hidy (2000), for example, reported that the 48-hr average outdoor PM_{2.5} mass concentration in Birmingham, Alabama, was 26.5 $\mu\text{g}/\text{m}^3$ in summer 1997 and 12.2 $\mu\text{g}/\text{m}^3$ in winter 1998. The median outdoor PM_{2.5} concentration reported in EXPOLIS for six cities in Europe was 7.3 $\mu\text{g}/\text{m}^3$ (Koistinen et al. 2001). These results indicate that both Waterfront South and Copewood-Davis are hot spots for PM_{2.5} air pollution.

The PM_{2.5} concentrations measured in the two neighborhoods were analyzed for differences by day of the week. As shown in Table 13, the mean PM_{2.5} concentration was $33.3 \pm 11.8 \mu\text{g}/\text{m}^3$ on weekdays and $29.4 \pm 13.0 \mu\text{g}/\text{m}^3$ on weekend days in Waterfront South and $26.7 \pm 10.5 \mu\text{g}/\text{m}^3$ on weekdays and $23.8 \pm 13.2 \mu\text{g}/\text{m}^3$ on weekend days in Copewood-Davis. The results show that PM_{2.5} concentrations were slightly higher on weekdays than on weekend days in both locations, but the differences were

Table 12a. Descriptive Summary of Ambient Air Concentrations ($\mu\text{g}/\text{m}^3$) for $\text{PM}_{2.5}$, VOCs, and Aldehydes in Waterfront South and Copewood–Davis

	<i>N</i>	Mean	SD	Median	Percentile				Detects (%)	<i>P</i> Value ^a	
					1st	5th	95th	99th			
VOCs—											
Waterfront South ^b											
MTBE	99	2.31	2.76	1.53	0.04	0.25	5.57	23.6	97.0	0.39	
Hexane	87	226	1442	2.16	0.13	0.19	65.1	12,578	63.6	0.54	
Chloroform	62	0.19	0.26	0.13	0.04	0.04	0.33	1.85	83.9	0.48	
Carbon tetrachloride	62	0.53	0.10	0.55	0.37	0.39	0.68	0.73	100	0.03	
Benzene	92	2.18	3.01	1.20	0.21	0.44	8.64	15.9	92.9	0.97	
Toluene	99	4.27	5.23	2.46	0.35	0.35	15.8	28.1	83.8	< 0.01	
Ethyl benzene	99	0.55	0.46	0.39	0.07	0.12	1.65	2.32	69.8	0.01	
<i>m</i> - & <i>p</i> -Xylenes	99	1.66	1.33	1.35	0.13	0.32	4.27	7.42	90.6	< 0.01	
<i>o</i> -Xylene	99	0.59	0.53	0.44	0.08	0.10	1.63	3.86	89.6	< 0.01	
Styrene	99	0.14	0.17	0.13	0.00	0.00	0.32	1.19	32.3	NA	
VOCs—											
Copewood–Davis ^b											
MTBE	86	2.35	2.89	1.63	0.04	0.38	7.54	22.4	98.8		
Hexane	75	266	1488	1.87	0.19	0.36	1110	11,869	67.4		
Chloroform	62	0.20	0.25	0.14	0.04	0.04	0.71	1.70	87.1		
Carbon tetrachloride	62	0.54	0.10	0.55	0.35	0.4	0.70	0.77	100		
Benzene	80	2.86	4.72	1.26	0.15	0.44	16.0	23.3	90.7		
Toluene	86	2.34	1.80	1.95	0.30	0.68	5.53	8.01	77.9		
Ethyl benzene	86	0.43	0.37	0.33	0.07	0.12	1.16	2.4	70.7		
<i>m</i> - & <i>p</i> -Xylenes	86	1.19	0.93	0.95	0.13	0.22	3.18	4.34	82.6		
<i>o</i> -Xylene	86	0.41	0.33	0.34	0.04	0.1	0.91	1.96	80.4		
Styrene	86	0.12	0.14	0.13	0.00	0.00	0.24	1.14	32.6		
Aldehydes—											
Waterfront South											
Formaldehyde	77	20.2	19.5	15.5	0.1	0.9	63.1	107.8	97.3	0.03	
Acetaldehyde	77	12.5	18.0	6.1	0.0	0.0	46.0	104.9	72.0	0.50	
Acrolein	77	0.8	0.9	0.2	0.1	0.1	3.0	3.4	41.3	NA	
Propionaldehyde	77	1.1	1.4	0.4	0.3	0.3	4.5	7.2	41.3	NA	
Aldehydes—											
Copewood–Davis											
Formaldehyde	75	24.8	20.8	20.4	0.1	2.0	68.7	101.8	98.7		
Acetaldehyde	75	14.6	21.9	8.3	0.0	0.0	52.7	135.4	72.0		
Acrolein	75	0.6	0.9	0.2	0.1	0.1	2.5	5.5	36.0		
Propionaldehyde	75	1.6	2.4	0.4	0.3	0.3	7.1	12.2	48.0		
PM _{2.5} —Waterfront South	114	31.3	12.5	29.7	10.4	12.6	51.8	60.1	100	< 0.01	
PM _{2.5} —Copewood–Davis	106	25.3	11.9	24.0	7.48	10.3	46.8	53.6	100		

^a Based on log-transformed data; original data were used for $\text{PM}_{2.5}$.^b The fraction of nondetected samples was > 90% for 1,3-butadiene (MDL = 1.97 $\mu\text{g}/\text{m}^3$). Its distribution was therefore not reported here.

NA indicates not available.

Table 12b. Descriptive Summary of Ambient PAH Concentrations (ng/m³) in Waterfront South and Copewood–Davis

	<i>N</i>	Mean	SD	Median	Percentile				Detects ^a (%)	<i>P</i> Value ^b
					1st	5th	95th	99th		
PAHs—Waterfront South										
Naphthalene	101	13.1	17.2	9.78	0.46	1.69	35.7	55.2	84.2	0.29
Acenaphthylene	101	3.37	5.90	1.78	0.15	0.15	10.3	20.4	93.1	NA
Acenaphthene	101	5.78	19.0	3.03	0.35	0.47	10.9	34.1	95.0	NA
Fluorene	101	11.8	21.3	8.38	1.17	1.73	23.7	84.2	93.1	NA
Phenanthrene	101	15.5	17.0	9.14	0.93	1.34	40.9	76.0	89.1	< 0.01
Anthracene	101	1.06	0.60	0.96	0.36	0.52	2.25	2.59	29.7	NA
Fluoranthene	101	2.44	2.65	1.91	0.17	0.38	7.39	12.7	43.6	NA
Pyrene	101	1.47	1.52	0.99	0.28	0.35	4.13	6.69	83.2	0.08
Benzo[<i>a</i>]anthracene	101	0.38	0.56	0.15	0.06	0.08	1.56	3.01	48.5	NA
Chrysene	101	0.61	0.56	0.54	0.07	0.27	1.32	3.16	24.8	NA
Benzo[<i>b</i>]fluoranthene	101	0.37	0.50	0.11	0.05	0.05	1.56	2.12	58.4	NA
Benzo[<i>k</i>]fluoranthene	101	0.24	0.35	0.09	0.04	0.05	1.21	1.44	58.4	NA
Benzo[<i>a</i>]pyrene	101	0.36	0.45	0.18	0.05	0.05	1.19	2.04	59.4	< 0.01
Indeno[1,2,3- <i>cd</i>]pyrene	101	0.23	0.41	0.05	0.05	0.05	1.12	2.05	35.6	NA
Benzo[<i>ghi</i>]perylene	101	0.11	0.20	0.05	0.04	0.05	0.34	0.81	23.8	NA
Dibenzo[<i>a,h</i>]anthracene	101	0.21	0.34	0.05	0.05	0.05	1.06	1.50	38.6	NA
PAHs—Copewood—Davis										
Naphthalene	91	11.7	11.2	7.62	1.09	1.78	33.6	46.4	82.4	
Acenaphthylene	91	2.40	2.82	1.20	0.08	0.13	9.32	10.4	84.6	
Acenaphthene	91	3.75	4.40	2.71	0.44	0.64	8.73	18.0	97.8	
Fluorene	91	9.25	8.12	6.12	1.02	1.89	26.5	37.8	92.3	
Phenanthrene	91	12.4	11.31	7.47	0.97	1.45	31.7	49.0	92.3	
Anthracene	91	1.03	0.79	0.96	0.51	0.53	1.90	4.07	25.3	
Fluoranthene	91	2.28	2.31	1.79	0.18	0.23	6.53	10.9	38.5	
Pyrene	91	1.23	1.17	0.91	0.16	0.22	3.99	5.70	72.5	
Benzo[<i>a</i>]anthracene	91	0.22	0.34	0.15	0.03	0.08	0.57	1.23	26.4	
Chrysene	91	0.46	0.25	0.42	0.08	0.10	0.79	1.32	14.3	
Benzo[<i>b</i>]fluoranthene	91	0.26	0.33	0.08	0.04	0.05	1.09	1.22	56.0	
Benzo[<i>k</i>]fluoranthene	91	0.15	0.19	0.07	0.03	0.05	0.43	0.92	56.0	
Benzo[<i>a</i>]pyrene	91	0.21	0.26	0.12	0.05	0.05	0.75	1.03	54.9	
Indeno[1,2,3- <i>cd</i>]pyrene	91	0.14	0.21	0.05	0.05	0.05	0.48	0.99	29.7	
Benzo[<i>ghi</i>]perylene	91	0.06	0.04	0.05	0.04	0.04	0.12	0.26	17.6	
Dibenzo[<i>a,h</i>]anthracene	91	0.12	0.15	0.05	0.05	0.05	0.39	0.75	31.9	

^a Percentage detects for PAHs were in gas phase, in particle phase, or both.

^b Based on log-transformed data.

NA indicates not available.

Table 13. Comparison of Ambient Concentrations for PM_{2.5}, VOCs, Aldehydes, and PAHs Between Waterfront South (WFS) and Copewood–Davis (CDS) by Day of the Week

	Weekday										Weekend										P Value of Comparison ^a	
	Waterfront South					Copewood–Davis					Waterfront South					Copewood–Davis						
	N	Mean	SD	Median	N	Mean	SD	Median	N	Mean	SD	Median	N	Mean	SD	Median	Week- day	Week- end	CDS	WFS		
Fine PM (µg/m ³)	55	33.3	11.8	23.5	55	26.7	10.5	24.9	59	29.4	13.0	26.1	51	23.8	13.2	19.8	< 0.01	0.02	0.41	0.87		
PM _{2.5}																						
VOCs (µg/m ³)																						
MTBE	56	2.11	1.76	1.36	48	2.26	2.16	1.54	43	2.57	3.70	1.68	38	2.47	3.64	1.70	0.53	0.57	0.35	0.34		
Hexane	48	5.91	11.41	2.17	41	3.84	6.38	1.74	39	4.96	21.38	1.83	34	5.83	21.85	1.99	0.34	0.88	0.05	0.10		
Chloroform	32	0.17	0.16	0.14	32	0.16	0.14	0.13	30	0.21	0.34	0.13	30	0.24	0.33	0.15	0.90	0.21	0.63	0.75		
Carbon tetrachloride	32	0.53	0.10	0.54	32	0.54	0.10	0.57	30	0.54	0.10	0.57	30	0.53	0.10	0.54	< 0.01	0.98	0.64	0.81		
Benzene	52	2.39	3.33	1.28	45	2.81	4.65	1.24	40	1.92	2.54	1.15	35	2.92	4.88	1.27	0.64	0.59	0.87	0.56		
Toluene	56	4.95	5.37	3.36	48	2.76	3.97	2.13	43	3.38	4.96	1.97	38	2.13	1.75	1.72	< 0.01	0.11	0.50	0.05		
Ethyl benzene	56	0.58	0.44	0.47	48	0.42	0.32	0.33	43	0.50	0.48	0.36	38	0.43	0.43	0.33	0.01	0.33	0.85	0.20		
m- & p-Xylenes	56	1.87	1.36	1.60	48	1.21	0.91	1.02	43	1.40	1.26	1.13	38	1.17	0.98	0.93	< 0.01	0.13	0.74	0.07		
o-Xylene	56	0.66	0.60	0.49	48	0.43	0.34	0.36	43	0.50	0.40	0.42	38	0.39	0.32	0.30	< 0.01	0.03	0.30	0.08		
Styrene	56	0.15	0.16	0.13	48	0.11	0.08	0.13	43	0.14	0.17	0.13	38	0.14	0.19	0.13	NA	NA	NA	NA		
Aldehydes (µg/m ³)																						
Formaldehyde	43	20.2	18.7	15.6	42	22.9	20.5	16.3	34	20.2	20.9	13.6	33	27.4	21.2	21.2	0.60	0.57	NA	NA		
Acetaldehyde	43	10.5	14.4	3.3	42	10.2	11.6	5.1	34	15.1	21.6	8.3	33	20.3	29.7	13.9	0.50	0.11	NA	NA		
Acrolein	43	0.8	1.1	0.2	42	0.7	1.0	0.2	34	0.7	0.7	0.2	33	0.6	0.8	0.2	NA	NA	NA	NA		
Propionaldehyde	43	0.9	1.0	0.4	42	1.4	2.3	0.4	34	1.4	1.7	0.4	33	1.8	2.6	0.4	NA	NA	NA	NA		
PAHs (ng/m ³)																						
Naphthalene	54	14.1	20.3	10.6	51	11.7	9.19	8.54	47	12.0	12.9	7.8	40	11.7	13.5	6.21	0.87	0.20	0.97	0.38		
Acenaphthylene	54	4.41	7.60	2.04	51	2.60	2.80	1.74	47	2.17	2.51	1.04	40	2.15	2.87	0.53	NA	NA	NA	NA		
Acenaphthene	54	7.33	25.5	3.19	51	4.14	5.36	2.71	47	3.99	5.15	2.93	40	3.24	2.71	2.67	NA	NA	NA	NA		
Fluorene	54	13.9	26.8	9.5	51	9.77	7.32	8.30	47	9.52	12.5	5.71	40	8.59	9.08	4.98	NA	NA	NA	NA		
Phenanthrene	54	17.5	20.2	9.6	51	12.9	11.7	7.95	47	13.3	12.1	7.63	40	11.6	10.8	6.88	0.33	< 0.01	0.77	0.11		
Anthracene	54	1.16	0.71	0.96	51	1.14	1.01	0.96	47	0.95	0.41	0.92	40	0.89	0.32	0.94	NA	NA	NA	NA		
Fluoranthene	54	3.06	3.35	2.01	51	2.46	2.70	1.91	47	1.73	1.15	1.41	40	2.05	1.69	1.71	NA	NA	NA	NA		
Pyrene	54	1.94	1.91	1.18	51	1.29	1.29	0.87	47	0.94	0.54	0.81	40	1.14	1.01	0.94	0.31	< 0.01	0.43	< 0.01		
Benzo[a]anthracene	54	0.46	0.65	0.18	51	0.23	0.42	0.15	47	0.29	0.42	0.15	40	0.21	0.19	0.15	NA	NA	NA	NA		
Chrysene	54	0.71	0.71	0.54	51	0.46	0.25	0.45	47	0.48	0.24	0.45	40	0.45	0.25	0.39	NA	NA	NA	NA		
Benzo[b]fluoranthene	54	0.43	0.57	0.12	51	0.29	0.36	0.16	47	0.31	0.39	0.10	40	0.21	0.27	0.05	NA	NA	NA	NA		
Benzo[k]fluoranthene	54	0.25	0.35	0.13	51	0.18	0.24	0.10	47	0.22	0.36	0.08	40	0.11	0.11	0.05	NA	NA	NA	NA		
Benzo[a]pyrene	54	0.44	0.54	0.20	51	0.26	0.31	0.15	47	0.26	0.31	0.08	40	0.14	0.13	0.05	0.02	< 0.01	0.11	< 0.01		
Indeno[1,2,3,-cd]pyrene	54	0.28	0.49	0.05	51	0.17	0.26	0.05	47	0.17	0.30	0.05	40	0.11	0.14	0.05	NA	NA	NA	NA		
Benzo[ghi]perylene	54	0.12	0.17	0.05	51	0.06	0.04	0.05	47	0.09	0.23	0.05	40	0.06	0.03	0.05	NA	NA	NA	NA		
Dibenzo[a,h]anthracene	54	0.22	0.35	0.05	51	0.12	0.14	0.05	47	0.19	0.32	0.05	40	0.11	0.16	0.05	NA	NA	NA	NA		

^a For PM_{2.5}, based on the original data; for other species, based on log-transformed data.

NA indicates not available.

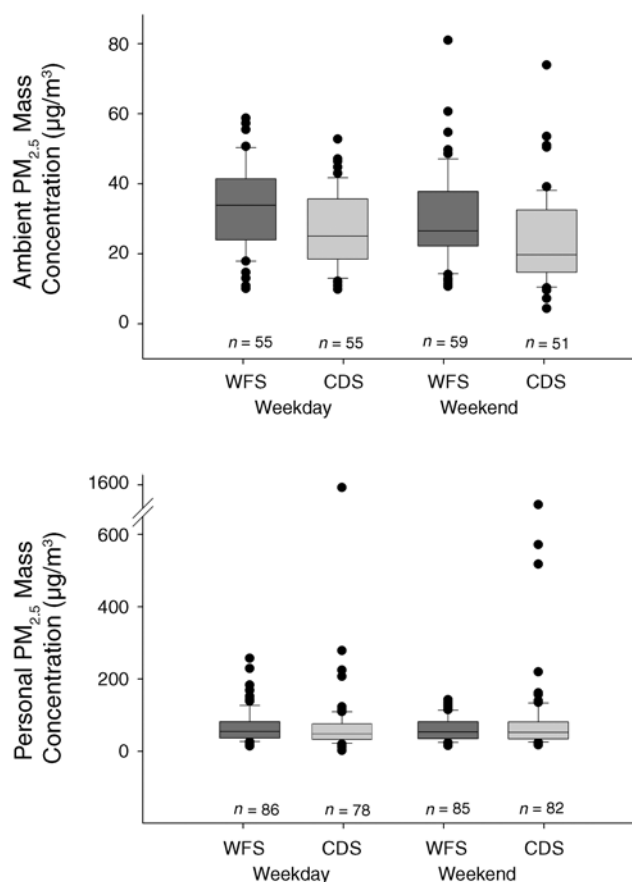


Figure 12. Ambient (top) and personal (bottom) PM_{2.5} concentrations in Waterfront South (WFS) and Copewood-Davis (CDS) by day of the week. Note that the y-axis scales differ; n is the number of samples in each category.

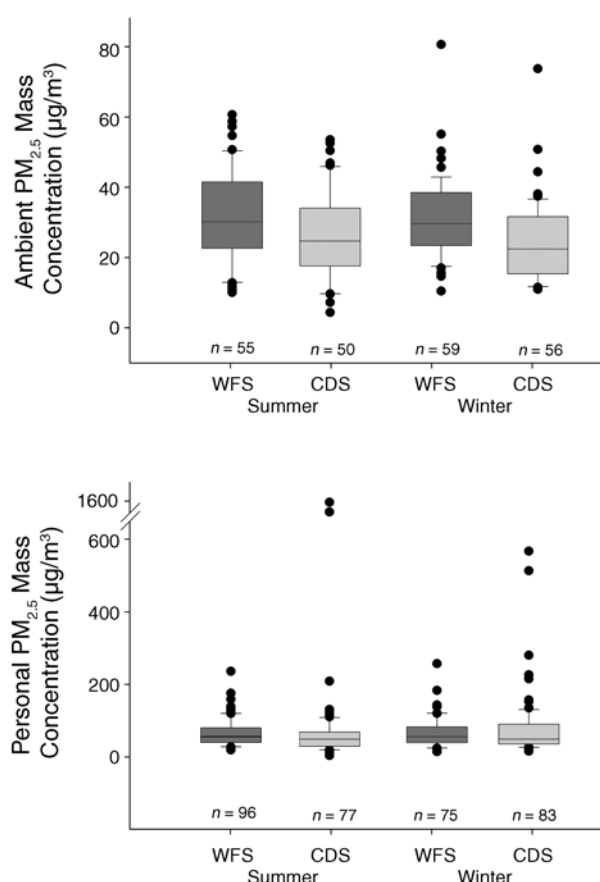


Figure 13. Ambient (top) and personal (bottom) PM_{2.5} concentrations in Waterfront South (WFS) and Copewood-Davis (CDS) by season. Note that the y-axis scales differ; n is the number of samples in each category.

not statistically significant ($P = 0.8735$ in Waterfront South and 0.4098 in Copewood-Davis). The seasonal variation in PM_{2.5} concentrations within each location was also examined. In Waterfront South the mean PM_{2.5} concentration was 31.4 ± 13.6 µg/m³ in summer and 31.2 ± 11.5 µg/m³ in winter (Table 14). In Copewood-Davis it was 26.1 ± 12.4 µg/m³ in summer and 24.6 ± 11.5 µg/m³ in winter. No significant differences in PM_{2.5} concentrations were observed between summer and winter in either location ($P = 0.23$ in Waterfront South and 0.43 in Copewood-Davis).

Personal PM_{2.5} Mass Concentration Results from personal measurements of PM_{2.5} in Waterfront South and Copewood-Davis are shown in Table 15a. The mean \pm SD and median personal PM_{2.5} concentrations were 61.9 ± 39.4 µg/m³ and 53.5 µg/m³ in Waterfront South, respectively, and 84.1 ± 179 µg/m³ and 49.1 µg/m³ in Copewood-Davis. The higher mean PM_{2.5} concentration in Copewood-Davis was caused by several extremely high

values obtained in the neighborhood and potentially by subject exposure to ETS, which was found to be significantly higher ($P = 0.01$) in Copewood-Davis (0.61 ± 0.17 µg/m³) than in Waterfront South (0.26 ± 0.16 µg/m³). Given the wide range of personal PM_{2.5} concentrations as well as the possible influence of ETS exposure, the median (instead of the mean) concentrations of PM_{2.5} measured in the two neighborhoods were used for general comparisons. The median personal PM_{2.5} concentration measured in Waterfront South was not significantly higher ($P = 0.6448$) than the concentrations measured in Copewood-Davis (Table 15a). However, the concentrations measured in both neighborhoods were higher than those reported in other U.S. urban areas (Vette et al. 2001; Weisel et al. 2005), indicating that residents in both had higher PM_{2.5} exposures.

Day of the week and season were examined for influences on personal PM_{2.5} exposures. Personal exposures were higher on weekdays than on weekend days in both locations

Table 14. Comparison of Ambient Concentrations for PM_{2.5}, VOCs, Aldehydes, and PAHs Between Waterfront South (WFS) and Copewood–Davis (CDS) by Season

	Summer										Winter										P Value of Comparison ^a			
	Waterfront South					Copewood–Davis					Waterfront South					Copewood–Davis					CDS vs. WFS		Summer vs. Winter	
	N	Mean	SD	Median	N	Mean	SD	Median	N	Mean	SD	Median	N	Mean	SD	Median	Sum-mer	Win-ter	CDS	WFS				
Fine PM (µg/m ³)																								
PM _{2.5}	55	31.4	11.6	24.7	50	26.1	12.4	24.8	59	31.2	11.5	29.7	56	24.6	11.5	22.7	0.01	< 0.01	0.43	0.23				
VOCs (µg/m ³)																								
MTBE	57	2.48	3.23	1.95	49	2.63	3.56	1.74	42	2.09	1.97	1.44	37	1.98	1.60	1.55	0.52	0.57	0.35	0.35				
Hexane	45	435	1993	3.87	38	524	2071	3.34	42	1.46	1.98	0.84	37	1.31	1.25	0.98	0.65	0.76	< 0.01	< 0.01				
Chloroform	27	0.28	0.37	0.18	27	0.30	0.33	0.17	35	0.11	0.07	0.10	35	0.13	0.13	0.11	0.34	0.89	< 0.01	< 0.01				
Carbon tetrachloride	27	0.57	0.09	0.58	27	0.58	0.09	0.59	35	0.50	0.10	0.45	35	0.51	0.10	0.50	0.29	0.06	0.01	0.01				
Benzene	50	2.77	3.82	1.25	43	3.98	6.14	1.32	42	1.49	1.30	1.17	37	1.55	1.30	1.09	0.80	0.72	0.07	0.03				
Toluene	57	4.08	4.99	2.63	49	1.79	1.25	1.73	42	4.53	5.59	2.37	37	3.40	4.51	2.12	< 0.01	0.30	0.01	0.45				
Ethyl benzene	57	0.49	0.39	0.39	49	0.34	0.34	0.28	42	0.63	0.53	0.40	37	0.55	0.37	0.33	< 0.01	0.67	< 0.01	0.11				
m- & p-Xylenes	57	1.42	0.91	1.24	49	0.96	0.70	0.92	42	2.00	1.71	1.41	37	1.51	1.11	1.24	< 0.01	0.09	0.01	0.12				
o-Xylene	57	0.50	0.33	0.43	49	0.35	0.27	0.33	42	0.71	0.70	0.47	37	0.49	0.39	0.40	< 0.01	0.06	0.03	0.27				
Styrene	57	0.14	0.18	0.13	49	0.11	0.16	0.08	42	0.15	0.15	0.14	37	0.14	0.10	0.13	NA	NA	NA	NA				
Aldehydes (µg/m ³)																								
Formaldehyde	44	20.7	20.8	15.5	42	27.7	22.1	23.3	33	19.6	18.0	14.4	33	21.3	18.7	15.3	< 0.01	0.01	NA	NA				
Acetaldehyde	44	8.7	12.7	1.2	42	11.6	14.5	4.3	33	17.7	22.4	11.2	33	18.4	28.6	10.4	0.11	0.09	NA	NA				
Acrolein	44	0.8	1.0	0.1	42	0.6	0.7	0.1	33	0.6	0.8	0.2	33	0.8	1.1	0.2	NA	NA	NA	NA				
Propionaldehyde	44	0.9	1.2	0.4	42	1.6	2.7	0.3	33	1.4	1.5	0.4	33	1.6	2.0	0.4	NA	NA	NA	NA				
PAHs (ng/m ³)																								
Naphthalene	48	5.23	3.95	3.90	42	4.81	3.49	3.95	53	20.3	21.1	16.0	49	17.6	12.2	14.4	0.84	0.21	< 0.01	< 0.01				
Acenaphthylene	48	1.98	3.80	0.46	42	1.49	2.73	0.23	53	4.63	7.10	3.02	49	3.19	2.68	2.80	NA	NA	NA	NA				
Acenaphthene	48	8.09	27.3	2.97	42	4.64	6.0	3.16	53	3.69	2.66	3.04	49	2.98	1.98	2.59	NA	NA	NA	NA				
Fluorene	48	18.1	29.5	11.3	42	13.8	9.6	13.0	53	6.18	4.38	4.75	49	5.38	3.47	4.14	NA	NA	NA	NA				
Phenanthrene	48	26.9	18.7	23.2	42	20.6	11.8	19.3	53	5.23	3.28	4.32	49	5.26	2.93	5.29	< 0.01	0.33	< 0.01	< 0.01				
Anthracene	48	1.31	0.75	0.97	42	1.28	1.10	0.96	53	0.84	0.28	0.96	49	0.82	0.18	0.89	NA	NA	NA	NA				
Fluoranthene	48	3.76	3.29	2.56	42	3.65	2.75	2.95	53	1.25	0.78	1.11	49	1.11	0.69	1.00	NA	NA	NA	NA				
Pyrene	48	1.91	1.78	1.22	42	1.60	1.21	1.28	53	1.08	1.12	0.74	49	0.91	1.05	0.51	0.08	0.45	0.03	< 0.01				
Benzo[a]anthracene	48	0.28	0.47	0.15	42	0.24	0.48	0.15	53	0.48	0.62	0.21	49	0.21	0.13	0.15	NA	NA	NA	NA				
Chrysene	48	0.47	0.33	0.43	42	0.42	0.31	0.35	53	0.73	0.68	0.54	49	0.49	0.17	0.49	NA	NA	NA	NA				
Benzo[b]fluoranthene	48	0.14	0.29	0.05	42	0.14	0.29	0.05	53	0.58	0.55	0.38	49	0.35	0.33	0.23	NA	NA	NA	NA				
Benzo[k]fluoranthene	48	0.08	0.06	0.05	42	0.07	0.05	0.05	53	0.38	0.43	0.21	49	0.21	0.24	0.12	NA	NA	NA	NA				
Benzo[a]pyrene	48	0.12	0.22	0.05	42	0.08	0.12	0.05	53	0.57	0.51	0.40	49	0.32	0.28	0.25	0.48	< 0.01	< 0.01	< 0.01				
Indeno[1,2,3-cd]pyrene	48	0.11	0.30	0.05	42	0.05	0.00	0.05	53	0.33	0.47	0.09	49	0.22	0.27	0.06	NA	NA	NA	NA				
Benzo[ghi]perylene	48	0.06	0.08	0.05	42	0.05	0.01	0.05	53	0.15	0.26	0.05	49	0.07	0.05	0.05	NA	NA	NA	NA				
Dibenzo[a,h]anthra-cene	48	0.10	0.18	0.05	42	0.05	0.02	0.05	53	0.30	0.41	0.13	49	0.17	0.19	0.08	NA	NA	NA	NA				

^a For PM_{2.5}, based on the original data; for other species, based on log-transformed data.

NA indicates not available.

Table 15a. Descriptive Summary of Personal Air Concentrations ($\mu\text{g}/\text{m}^3$) for VOCs, Aldehydes, and $\text{PM}_{2.5}$ in Waterfront South and Copewood–Davis

	<i>N</i>	Mean	SD	Median	Percentile				Detects ^a (%)	<i>P</i> value ^a
					1st	5th	95th	99th		
VOCs ^b —Waterfront South										
MTBE	200	4.15	5.47	2.59	0.12	0.40	12.9	29.6	99.0	0.22
Hexane	181	215	1116	2.81	0.19	0.63	230	7969	78.5	0.21
Chloroform	173	0.96	1.50	0.49	0.04	0.13	3.32	7.28	98.8	0.16
Carbon tetrachloride	173	0.51	0.13	0.52	0.28	0.32	0.71	0.89	100	0.02
Benzene	187	2.90	2.35	2.12	0.44	0.65	7.24	13.9	97.5	< 0.01
Toluene	200	28.0	286	5.83	0.40	1.60	17.4	107	97.0	0.34
Ethyl benzene	200	1.41	4.02	0.89	0.12	0.28	3.16	6.41	73.5	0.16
<i>m</i> - & <i>p</i> -Xylenes	200	4.06	11.7	2.45	0.41	0.67	9.37	16.0	95.4	0.29
<i>o</i> -Xylene	200	1.25	3.27	0.77	0.11	0.32	2.66	6.25	96.6	0.17
Styrene	200	0.27	0.25	0.14	0.03	0.08	0.78	1.27	54.5	0.07
VOCs ^b —Copewood–Davis										
MTBE	165	6.39	19.7	2.41	0.36	0.48	17.9	164	100	
Hexane	139	539	3711	3.16	0.53	0.81	1184	11,976	77.0	
Chloroform	130	1.96	4.99	0.52	0.04	0.07	6.95	26.5	96.2	
Carbon tetrachloride	130	0.47	0.13	0.44	0.26	0.32	0.66	0.79	99.2	
Benzene	153	4.73	8.30	2.81	0.44	0.96	11.5	57.6	98.2	
Toluene	165	11.4	40.0	6.19	0.40	1.14	24.4	87.9	96.4	
Ethyl benzene	165	1.57	1.91	0.98	0.07	0.20	5.43	11.0	87.5	
<i>m</i> - & <i>p</i> -Xylenes	165	4.50	5.70	2.82	0.14	0.41	14.9	32.2	91.8	
<i>o</i> -Xylene	165	1.42	1.80	0.93	0.08	0.11	4.92	12.1	91.8	
Styrene	165	0.37	0.70	0.18	0.01	0.05	1.03	3.88	69.1	
Aldehydes—Waterfront South										
Formaldehyde	180	16.8	15.5	14.1	0.1	0.1	46.7	85.5	90.5	0.84
Acetaldehyde	180	15.5	15.5	11.4	0.0	0.0	43.0	74.9	86.0	0.02
Acrolein	180	1.2	1.7	0.5	0.1	0.1	4.7	9.7	55.9	NA
Propionaldehyde	180	2.2	8.4	0.8	0.3	0.3	5.2	10.4	53.6	NA
Aldehydes—Copewood–Davis										
Formaldehyde	165	16.0	16.7	11.5	0.1	0.1	50.6	82.4	90.7	
Acetaldehyde	165	16.6	15.8	15.4	0.0	0.0	46.8	78.7	85.8	
Acrolein	165	1.4	2.5	0.4	0.1	0.1	4.8	14.5	53.7	
Propionaldehyde	165	1.9	2.4	0.9	0.3	0.3	6.0	10.4	56.8	
PM _{2.5} —Waterfront South	171	61.9	39.4	53.3	14.8	19.8	135	197	100	0.64
PM _{2.5} —Copewood–Davis	160	84.1	179	49.1	13.3	18.6	160	953	100	

^a Based on log-transformed data; original data were used for $\text{PM}_{2.5}$.

^b The fraction of nondetected samples was > 90% for 1,3-butadiene ($\text{MDL}=1.97\mu\text{g}/\text{m}^3$). Its distribution was therefore not reported here.

NA indicates not available.

(Table 16). In Waterfront South the median personal $\text{PM}_{2.5}$ concentration was $54.7 \mu\text{g}/\text{m}^3$ on weekdays and $50.3 \mu\text{g}/\text{m}^3$ on weekend days. In Copewood–Davis it was $50.2 \mu\text{g}/\text{m}^3$ on weekdays and $47.8 \mu\text{g}/\text{m}^3$ on weekend days. The concentration was only 9% higher on weekdays

than on weekend days in Waterfront South and 5% in Copewood–Davis, and the difference in the exposure between weekdays and weekend days was marginal in Waterfront South ($P = 0.0841$) and not of any statistical significance in Copewood–Davis ($P = 0.6652$).

Table 15b. Descriptive Summary for Personal PAH Concentrations (ng/m³) in Waterfront South and Copewood–Davis (CDS)

	<i>N</i>	Mean	SD	Median	Percentile				Detects ^a (%)	<i>P</i> value ^b
					1st	5th	95th	99th		
PAHs—Waterfront South										
Naphthalene	156	64.4	197	22.3	1.68	2.25	205	851	92.3	0.01
Acenaphthylene	156	5.34	10.5	2.49	0.11	0.15	15.4	56.4	87.2	NA
Acenaphthene	156	12.3	16.2	7.33	0.40	0.44	39.6	74.9	92.9	NA
Fluorene	156	39.8	51.7	26.1	1.72	2.03	119	230	92.3	NA
Phenanthrene	156	24.9	28.0	17.5	1.12	1.84	64.3	130	92.9	0.44
Anthracene	156	1.92	2.02	1.30	0.56	0.78	4.00	13.0	62.8	NA
Fluoranthene	156	1.97	1.51	1.75	0.13	0.29	5.02	7.43	32.1	NA
Pyrene	156	1.28	1.16	0.94	0.21	0.34	3.38	6.89	76.3	0.34
Benzo[<i>a</i>]anthracene	156	0.36	1.17	0.15	0.07	0.11	0.89	3.23	37.8	NA
Chrysene	156	0.58	0.74	0.54	0.07	0.24	1.13	2.83	13.5	NA
Benzo[<i>b</i>]fluoranthene	156	0.40	0.94	0.05	0.05	0.05	1.57	4.46	50.0	NA
Benzo[<i>k</i>]fluoranthene	156	0.18	0.32	0.05	0.05	0.05	0.82	1.29	48.7	NA
Benzo[<i>a</i>]pyrene	156	0.38	0.93	0.05	0.05	0.05	1.55	4.16	38.5	0.72
Indeno[1,2,3- <i>cd</i>]pyrene	156	0.23	0.69	0.05	0.05	0.05	1.11	3.37	21.2	NA
Benzo[<i>ghi</i>]perylene	156	0.23	1.04	0.05	0.04	0.05	0.26	5.74	12.8	NA
Dibenzo[<i>a,h</i>]anthracene	156	0.32	1.15	0.05	0.05	0.05	1.00	4.82	33.3	NA
PAHs—Copewood–Davis										
Naphthalene	150	133	770	32.9	2.05	3.86	320	868	96.7	
Acenaphthylene	150	4.71	6.25	2.63	0.15	0.15	17.6	29.4	86.7	
Acenaphthene	150	14.1	22.5	9.08	0.41	0.45	48.3	115	94.7	
Fluorene	150	56.6	75.3	28.2	1.10	2.80	172	382	94.7	
Phenanthrene	150	23.4	20.5	17.5	1.02	1.84	61.7	111	92.7	
Anthracene	150	2.38	2.47	1.64	0.56	0.80	6.61	13.1	66.7	
Fluoranthene	150	2.32 ^c	5.31 ^c	1.64	0.12	0.26	6.18	37.5	26.0	
Pyrene	150	1.56 ^c	2.60 ^c	0.88	0.22	0.27	4.38	19.0	72.0	
Benzo[<i>a</i>]anthracene	150	0.58	2.60	0.15	0.08	0.11	1.51	6.85	31.3	
Chrysene	150	0.75	1.26	0.54	0.24	0.30	2.47	6.97	11.3	
Benzo[<i>b</i>]fluoranthene	150	0.38	1.01	0.05	0.05	0.05	1.90	6.10	47.3	
Benzo[<i>k</i>]fluoranthene	150	0.29	1.07	0.05	0.05	0.05	0.67	4.92	46.7	
Benzo[<i>a</i>]pyrene	150	0.42	1.08	0.05	0.05	0.05	1.79	6.53	41.3	
Indeno[1,2,3- <i>cd</i>]pyrene	150	0.43	1.80	0.05	0.05	0.05	1.01	11.3	18.7	
Benzo[<i>ghi</i>]perylene	150	0.68	4.28	0.05	0.04	0.05	0.30	20.1	16.0	
Dibenzo[<i>a,h</i>]anthracene	150	0.47	2.24	0.05	0.05	0.05	0.66	14.8	28.0	

^a Percentage detects for PAHs were in gas phase, particle phase, or both.^b Based on log-transformed data.^c Two extreme values were not included in the analysis.

NA indicates not available.

No significant seasonal differences in personal PM_{2.5} were observed in either Waterfront South ($P = 0.1119$) or Copewood–Davis ($P = 0.0817$) (Table 17). In Waterfront South the median personal PM_{2.5} concentration was 56.4 µg/m³ in winter and 47.7 µg/m³ in summer. In Copewood–Davis it

was 48.5 µg/m³ in winter, similar to that in summer (49.5 µg/m³).

Personal exposure to PM_{2.5} is affected by both indoor and outdoor sources; as mentioned above, for example, a particular concern was the potential for subject exposure

Table 16. Comparison of Personal Concentrations for PM_{2.5}, VOCs, Aldehydes, and PAHs Between Waterfront South (WFS) and Copewood–Davis (CDS) by Day of the Week

	Weekday						Weekend						P Value of Comparison ^a							
	Waterfront South			Copewood–Davis			Waterfront South			Copewood–Davis			CDS vs. WFS		Weekday vs. Weekend					
	N	Mean	SD	Median	N	Mean	SD	Median	N	Mean	SD	Median	Week-day	Week-end						
Fine PM (µg/m ³)																				
PM _{2.5}	86	67.4	45.4	54.7	78	81.1	179	50.2	75	56.4	31.5	50.3	82	87.0	180	47.8	0.17	0.59	0.08	0.66
VOCs (µg/m ³)																				
MTBE	107	3.49	4.23	2.32	84	5.38	11.5	2.44	93	4.90	6.56	2.74	81	7.42	25.7	2.36	0.35	0.29	0.49	0.45
Hexane	91	3.54	3.88	2.53	69	4.50	6.21	3.02	90	4.28	15.58	3.12	70	10.66	51.94	3.25	0.17	0.32	0.04	0.06
Chloroform	87	1.06	1.83	0.47	64	1.85	3.69	0.55	86	0.86	1.07	0.51	66	2.07	6.01	0.51	0.28	0.20	0.60	0.75
Carbon tetrachloride	87	0.51	0.12	0.53	64	0.48	0.14	0.45	86	0.51	0.13	0.52	66	0.46	0.12	0.43	0.21	0.01	0.84	0.38
Benzene	100	3.06	2.27	2.29	79	4.67	6.88	2.70	87	2.73	2.45	1.99	74	4.80	9.64	3.04	< 0.01	< 0.01	0.41	0.74
Toluene	107	47.0	391	6.55	84	13.6	54.7	6.18	93	6.23	4.51	4.97	81	9.12	12.4	6.19	0.91	0.12	0.09	0.75
Ethyl benzene	107	1.76	5.42	1.05	84	1.56	1.71	1.00	93	1.01	0.88	0.75	81	1.59	2.11	0.95	0.67	0.06	0.03	0.37
m- & p-Xylenes	107	5.13	15.8	3.00	84	4.38	4.67	2.87	93	2.83	2.05	2.14	81	4.62	6.63	2.75	0.83	0.12	0.04	0.38
o-Xylene	107	1.55	4.42	0.95	84	1.41	1.38	0.93	93	0.90	0.67	0.72	81	1.44	2.17	0.93	0.60	0.08	0.05	0.36
Styrene	107	0.31	0.29	0.18	84	0.39	0.83	0.19	93	0.24	0.19	0.13	81	0.34	0.52	0.15	0.15	0.16	0.35	0.37
Aldehydes (µg/m ³)																				
Formaldehyde	98	16.7	16.5	13.7	85	16.3	16.3	11.8	82	16.8	14.2	14.3	80	15.7	17.1	10.2	0.77	0.03	0.93	0.97
Acetaldehyde	98	12.9	12.1	9.7	85	14.7	12.2	14.4	82	18.6	18.3	12.7	80	18.6	18.7	16.6	0.50	0.46	0.14	0.21
Acrolein	98	1.3	1.8	0.6	85	1.7	2.7	0.6	82	1.1	1.6	0.4	80	1.1	2.0	0.2	NA	NA	NA	NA
Propionaldehyde	98	2.9	11.3	0.8	85	2.0	3.0	0.7	82	1.4	1.4	0.6	80	1.8	1.8	1.5	NA	NA	NA	NA
PAHs (ng/m ³)																				
Naphthalene	80	58.8	138	19.4	78	194	1064	31.1	76	70.2	245	26.5	72	66.9	99.9	36.1	0.05	0.02	0.43	0.82
Acenaphthylene	80	5.08	10.2	2.23	78	4.76	5.94	2.71	76	5.61	10.8	3.10	72	4.66	6.61	2.57	NA	NA	NA	NA
Acenaphthene	80	12.2	17.5	6.77	78	14.8	23.1	10.1	76	12.4	14.8	8.05	72	13.4	21.9	7.49	NA	NA	NA	NA
Fluorene	80	40.1	59.2	21.6	78	54.0	57.9	29.9	76	39.4	42.8	28.9	72	59.4	90.9	27.2	NA	NA	NA	NA
Phenanthrene	80	22.7	29.6	15.2	78	22.9	20.6	17.0	76	27.2	26.3	20.4	72	24.1	20.5	18.10	0.83	0.19	0.09	0.59
Anthracene	80	1.93	2.31	1.26	78	2.20	1.78	1.65	76	1.92	1.67	1.32	72	2.56	3.06	1.40	NA	NA	NA	NA
Fluoranthene	80	2.21	1.79	1.86	78	1.97 ^b	1.77 ^b	1.66	76	1.72	1.10	1.66	72	2.70	7.43	1.44	NA	NA	NA	NA
Pyrene	80	1.43	1.44	0.99	78	1.50 ^b	1.83 ^b	1.00	76	1.11	0.73	0.93	72	1.63	3.24	0.68	0.84	0.26	0.53	0.15
Benzo[a]anthracene	80	0.47	1.61	0.15	78	0.34	0.83	0.15	76	0.25	0.26	0.15	72	0.84	3.65	0.15	NA	NA	NA	NA
Chrysene	80	0.70	1.00	0.54	78	0.81	1.51	0.54	76	0.46	0.20	0.48	72	0.68	0.93	0.54	NA	NA	NA	NA
Benzo[b]fluoranthene	80	0.49	1.19	0.09	78	0.33	0.85	0.05	76	0.29	0.58	0.05	72	0.44	1.17	0.05	NA	NA	NA	NA
Benzo[k]fluoranthene	80	0.17	0.27	0.05	78	0.35	1.37	0.05	76	0.19	0.36	0.05	72	0.22	0.59	0.05	NA	NA	NA	NA
Benzo[a]pyrene	80	0.32	0.47	0.05	78	0.50	1.24	0.05	76	0.44	1.25	0.05	72	0.33	0.87	0.05	0.42	0.79	0.66	0.19
Indeno[1,2,3-cd]pyrene	80	0.33	0.89	0.05	78	0.49	1.90	0.05	76	0.13	0.34	0.05	72	0.37	1.71	0.05	NA	NA	NA	NA
Benzo[ghi]perylene	80	0.23	1.01	0.05	78	0.42	3.11	0.05	76	0.23	1.08	0.05	72	0.96	5.28	0.05	NA	NA	NA	NA
Dibenzo[a,h]anthracene	80	0.35	0.92	0.05	78	0.34	1.83	0.05	76	0.30	1.35	0.05	72	0.60	2.62	0.05	NA	NA	NA	NA

^a Based on log-transformed data.

^b Outliers (360,000 ng/m³ for fluoranthene and 410,000 ng/m³ for pyrene) were excluded from analysis.

NA indicates not available.

Table 17. Comparison of Personal Concentrations of PM_{2.5}, VOCs, Aldehydes, and PAHs Between Waterfront South (WFS) and Copewood–Davis (CDS) by Season

	Summer										Winter										P Value of Comparison ^a	
	Waterfront South					Copewood–Davis					Waterfront South					Copewood–Davis						
	N	Mean	SD	Median	N	Mean	SD	Median	N	Mean	N	Mean	SD	Median	N	Mean	SD	Med	Sum-mer	Win-ter	WFS	CDS
Fine PM (µg/m ³)	96	58.3	37.3	47.7	77	92.2	242	48.5	75	66.6	41.7	56.4	83	76.6	87.3	49.5		0.75	0.64	0.11	0.08	
PM _{2.5}																						
VOCs (µg/m ³)																						
MTBE	104	4.74	6.07	3.10	86	9.55	27.0	2.39	96	3.51	4.68	2.25	79	2.94	2.07	2.44		0.26	0.39	0.71	0.62	
Hexane	85	454	1600	3.27	60	1245	5597	3.95	96	3.08	4.14	2.03	79	3.36	3.24	2.23		0.01	0.67	< 0.01	< 0.01	
Chloroform	85	1.08	1.26	0.55	51	2.81	7.54	0.52	88	0.85	1.70	0.46	79	1.42	1.97	0.48		0.20	0.29	0.05	0.04	
Carbon tetrachloride	85	0.56	0.13	0.55	51	0.51	0.16	0.51	88	0.46	0.11	0.46	79	0.44	0.10	0.39		0.02	0.24	< 0.01	0.02	
Benzene	91	3.02	2.85	1.79	74	6.83	11.5	3.36	96	2.79	1.77	2.47	79	2.77	1.78	2.30		< 0.01	0.26	0.64	< 0.01	
Toluene	104	8.62	18.2	5.39	86	9.15	12.0	6.41	96	49.1	413	6.45	79	13.8	56.5	5.75		0.30	0.66	0.42	0.74	
Ethyl benzene	104	1.55	5.47	0.78	86	1.40	1.44	0.99	96	1.26	1.16	0.96	79	1.76	2.31	0.97		0.29	0.22	0.26	0.22	
m- & p-Xylenes	104	4.50	16.0	2.15	86	4.15	4.95	2.87	96	3.58	3.01	2.78	79	4.88	6.43	2.82		0.48	0.28	0.45	0.30	
o-Xylene	104	1.38	4.44	0.74	86	1.38	1.51	1.00	96	1.10	0.99	0.89	79	1.47	2.08	0.86		0.60	0.08	0.05	0.36	
Styrene	104	0.18	0.18	0.13	86	0.26	0.48	0.13	96	0.37	0.27	0.32	79	0.48	0.86	0.32		0.37	0.05	< 0.01	< 0.01	
Aldehydes (µg/m ³)																						
Formaldehyde	101	20.7	18.2	16.3	82	21.4	20.8	16.6	79	11.7	8.8	12.0	83	10.6	8.3	9.3		0.92	0.75	< 0.01	< 0.01	
Acetaldehyde	101	12.9	13.7	7.8	82	12.5	11.4	8.9	79	18.9	17.0	16.2	83	20.7	18.3	18.1		< 0.01	0.22	0.03	0.05	
Acrolein	101	1.3	2.0	0.5	82	1.8	3.0	0.7	79	1.0	1.3	0.5	83	0.9	1.8	0.2		NA	NA	NA	NA	
Propionaldehyde	101	1.5	2.1	0.4	82	2.1	2.2	1.3	79	3.1	12.4	1.5	83	1.8	2.7	0.8		NA	NA	NA	NA	
PAHs (ng/m ³)																						
Naphthalene	82	80.2	245	23.3	73	206	1098	35.8	74	46.9	123	20.0	77	63.7	107.6	29.9		0.05	0.08	0.19	0.12	
Acenaphthylene	82	4.39	7.26	2.02	73	4.04	6.41	1.80	74	6.39	13.1	3.57	77	5.36	6.07	3.29		NA	NA	NA	NA	
Acenaphthene	82	17.6	20.3	10.5	73	15.6	24.2	9.26	74	6.47	5.47	5.40	77	12.7	20.8	7.17		NA	NA	NA	NA	
Fluorene	82	59.3	63.9	37.6	73	69.8	93.2	33.2	74	18.1	15.9	15.9	77	44.0	50.7	24.8		NA	NA	NA	NA	
Phenanthrene	82	36.2	28.4	31.6	73	33.5	24.5	25.1	74	12.4	21.8	10.1	77	13.9	8.0	14.5		0.69	0.14	< 0.01	< 0.01	
Anthracene	82	2.64	2.56	2.02	73	3.22	3.19	2.27	74	1.12	0.37	0.96	77	1.58	1.01	1.13		NA	NA	NA	NA	
Fluoranthene	82	2.57	1.69	1.97	73	3.31	7.37	1.91	74	1.30	0.90	1.25	77	1.38 ^b	1.31 ^b	1.13		NA	NA	NA	NA	
Pyrene	82	1.61	1.23	1.25	73	2.17	3.28	1.44	74	0.90	0.95	0.60	77	0.99 ^b	1.54 ^b	0.56		0.67	0.36	< 0.01	< 0.01	
Benzol[a]anthracene	82	0.46	1.58	0.15	73	0.95	3.67	0.15	74	0.25	0.35	0.15	77	0.23	0.45	0.15		NA	NA	NA	NA	
Chrysene	82	0.63	1.00	0.53	73	0.96	1.75	0.54	74	0.52	0.18	0.54	77	0.55	0.39	0.54		NA	NA	NA	NA	
Benzol[b]fluoranthene	82	0.48	1.27	0.05	73	0.49	1.33	0.05	74	0.30	0.28	0.25	77	0.28	0.57	0.14		NA	NA	NA	NA	
Benzol[k]fluoranthene	82	0.15	0.35	0.05	73	0.37	1.45	0.05	74	0.22	0.28	0.09	77	0.21	0.47	0.09		NA	NA	NA	NA	
Benzol[a]pyrene	82	0.15	0.37	0.05	73	0.44	1.40	0.05	74	0.64	1.25	0.33	77	0.40	0.65	0.20		0.21	0.08	< 0.01	< 0.01	
Indeno[1,2,3-cd]pyrene	82	0.18	0.78	0.05	73	0.59	2.47	0.05	74	0.29	0.57	0.05	77	0.28	0.74	0.05		NA	NA	NA	NA	
Benzol[ghi]perylene	82	0.23	1.02	0.05	73	1.01	5.87	0.05	74	0.24	1.07	0.05	77	0.37	1.77	0.05		NA	NA	NA	NA	
Dibenzo[a,h]anthracene	82	0.18	0.66	0.05	73	0.56	2.72	0.05	74	0.48	1.50	0.06	77	0.38	1.69	0.05		NA	NA	NA	NA	

^a Based on log-transformed data.^b Outliers (360,000 ng/m³ for fluoranthene and 410,000 ng/m³ for pyrene) were excluded from analysis.

NA indicates not available.

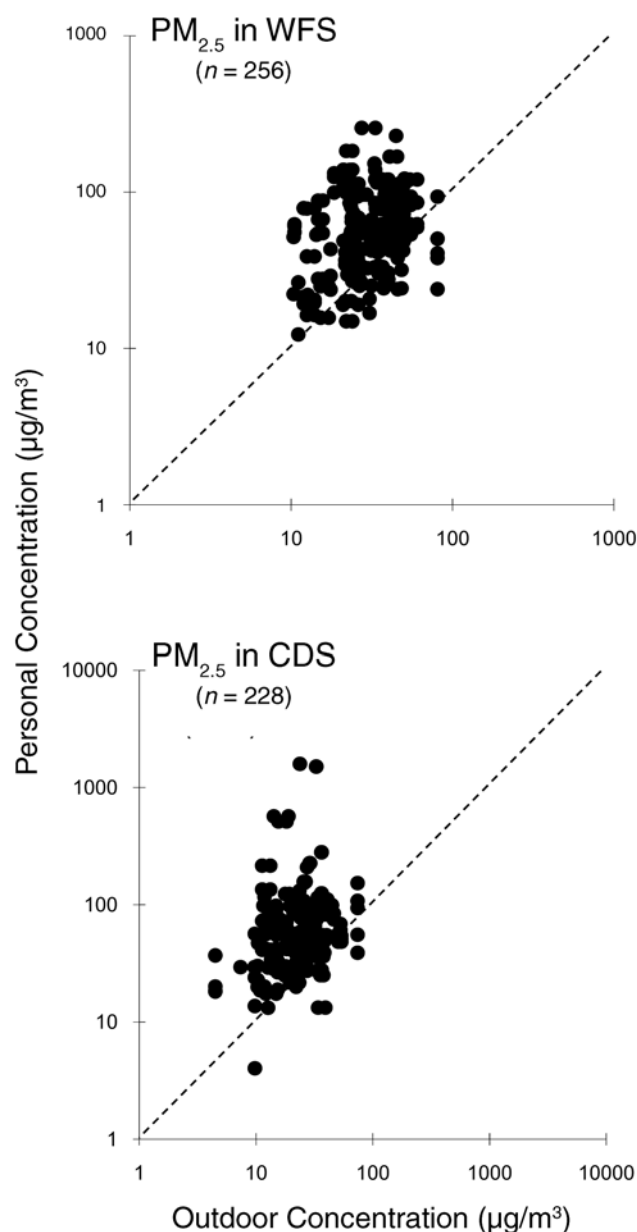


Figure 14. Scatter plots of personal versus ambient PM_{2.5} concentrations in (top) Waterfront South (WFS) and (bottom) Copewood-Davis (CDS). Note that scales are logarithmic and differ; *n* is the number of samples in each category.

to ETS. In addition, because we had measured 24-hr integrated personal exposure concentrations, occupational and other exposures occurring outside the neighborhoods might have contributed to the personal concentrations measured for PM_{2.5} and the other pollutants.

Personal and Ambient PM_{2.5} Relationships The relationships between personal exposures and ambient concentrations were examined in the following ways. First, personal

Table 18. Spearman Correlation Coefficients (*R*) Between Personal and Ambient Concentrations by Location

	Waterfront South	Copewood-Davis
VOCs		
MTBE	0.82	0.65
Hexane	0.64	0.66
Chloroform	0.20	0.40
Carbon tetrachloride	0.73	0.85
Benzene	0.47	0.45
Toluene	0.48	0.35
Ethyl benzene	0.53	0.26
<i>m</i> - & <i>p</i> -Xylenes	0.56	0.27
<i>o</i> -Xylene	0.53	0.32
Styrene	0.36	0.33
Aldehydes		
Formaldehyde	0.49	0.61
Acetaldehyde	0.68	0.68
PAHs		
Naphthalene	0.10	0.23
Phenanthrene	0.37	0.24
Pyrene	0.25	0.34
Benzo[a]pyrene	0.12	0.45
Fine PM		
PM _{2.5}	0.35	0.26

exposure concentrations were compared with ambient concentrations using scatter plots. Second, the strength of the associations between personal exposure and ambient concentrations was quantified using Spearman correlation coefficients. Third, the strength of the associations was quantified using mixed-effects models.

Scatter plots of personal versus ambient PM_{2.5} concentrations in Waterfront South and Copewood-Davis are shown in Figure 14. As can be seen, the data collected were mostly above the 1:1 line, meaning that the personal PM_{2.5} concentrations measured in both locations were higher than the ambient concentrations. Our observations at both locations were similar to those reported by Sax and colleagues (2004) and Weisel and colleagues (2005). In all cases the elevated personal PM exposures appeared to be affected not only by outdoor PM_{2.5} but also by indoor sources. Ott (1998) and Ott and Roberts (1998) reported various indoor sources, such as cooking and fireplace use, that might significantly contribute to personal total exposure.

The *R* was 0.31 for all personal and ambient PM_{2.5} measurements collected at the two locations (Table 18), indicating that the personal and ambient concentrations were

poorly correlated. When the population was separated by location, the R in Waterfront South was 0.35, slightly higher than that obtained for Copewood–Davis (0.26). The R was higher for weekdays (0.46) than for weekend days (0.23), which might be attributable, at least in part, to the fact that personal activities can be significantly different on weekdays and weekend days.

Season was found to be the most significant factor affecting the strength of the association between personal and ambient $PM_{2.5}$. For summer the R was 0.51; it was only 0.06 for winter. This is consistent with the amounts of time people spend in various microenvironments and the indoor air-exchange rate in the two different seasons. In summer people spend more time outdoors than in winter, and the air-exchange rate could be higher because people open windows more often in summer. Most subjects in the study did not have central air conditioning; many of them used fans for air circulation and cooling in summer. Each of these activities lead to higher personal exposures to ambient air and thus to a higher correlation between personal and ambient $PM_{2.5}$ concentrations in summer than in winter.

A mixed-effects model was used to compare personal and ambient $PM_{2.5}$ concentrations. When all personal $PM_{2.5}$ measurements collected from both locations were included in the analysis, the personal mean concentration ($73 \mu\text{g}/\text{m}^3$) was significantly higher ($P < 0.001$) than the corresponding ambient mean concentration ($28.2 \mu\text{g}/\text{m}^3$). After the removal of measurements for ETS-exposed subjects, the mean personal concentration dropped ($68.1 \mu\text{g}/\text{m}^3$) but was still significantly higher ($P < 0.001$) than the ambient mean concentration ($28.2 \mu\text{g}/\text{m}^3$). The finding indicated that both ambient and indoor $PM_{2.5}$ sources contributed to personal exposure for people living in the two areas and that indoor or occupational sources other than ETS dominated personal $PM_{2.5}$ exposures.

Mixed-effects models were also used to explore associations between personal and ambient $PM_{2.5}$ concentrations by location, season, and day of the week, as shown in Tables 19a, b, and c. A significant association ($P = 0.02$) was found only for summer, and it was significant ($P = 0.01$) only on weekdays, which is consistent with the strong associations described above between personal exposure and ambient concentrations in summer and on

Table 19a. Associations Between Personal and Ambient Air Toxic Concentrations for Waterfront South and Copewood–Davis^a

	Waterfront South			Copewood–Davis		
	Slope	P Value	R^2	Slope	P Value	R^2
VOCs ($\mu\text{g}/\text{m}^3$)						
MTBE	0.80	< 0.0001	0.92	0.80	< 0.0001	0.80
Hexane	0.88	< 0.0001	0.96	0.86	< 0.0001	0.96
Benzene	0.50	< 0.0001	0.86	0.41	< 0.0001	0.80
Toluene	0.42	< 0.0001	0.51	0.46	< 0.0001	0.50
<i>o</i> -Xylene	0.47	< 0.0001	0.62	0.53	< 0.0001	0.81
Aldehydes ($\mu\text{g}/\text{m}^3$)						
Formaldehyde	0.5	< 0.001	0.86	0.58	< 0.001	0.9
Acetaldehyde	0.6	< 0.001	0.81	0.51	< 0.001	0.79
PAHs ^b (ng/m^3)						
Naphthalene	−0.01	0.9231	0.00	0.25	0.0411	0.04
Phenanthrene	0.30	0.0010	0.10	0.19	0.0418	0.06
Pyrene	0.25	0.0066	0.07	0.31	0.0629	0.02
Benzo[<i>a</i>]pyrene	0.09	0.4427	0.02	0.47	0.0004	0.14
Fine PM ($\mu\text{g}/\text{m}^3$)						
$PM_{2.5}$	0.47	0.10	0.78	1.80	0.33	0.98

^a Values are based on log-transformed concentrations. P values are for the slope of each individual variable. R^2 s are the coefficients of determination of the mixed-effect model.

^b Sum of all the target compounds in both gas and particle phases.

Table 19b. Associations Between Personal and Ambient Air Toxic Concentrations by Season^a

	Summer			Winter		
	Slope	<i>P</i> Value	<i>R</i> ²	Slope	<i>P</i> Value	<i>R</i> ²
VOCs (µg/m ³)						
MTBE	0.95	< 0.0001	0.88	0.61	< 0.0001	0.89
Hexane	0.93	< 0.0001	0.97	0.69	< 0.0001	0.78
Benzene	0.41	< 0.0001	0.83	0.42	< 0.0001	0.82
Toluene	0.28	< 0.0001	0.64	0.44	< 0.0001	0.57
<i>o</i> -Xylene	0.28	0.0001	0.69	0.44	< 0.0001	0.89
Aldehydes (µg/m ³)						
Formaldehyde	0.66	< 0.001	0.92	0.13	0.002	0.78
Acetaldehyde	0.68	< 0.001	0.84	0.42	< 0.001	0.8
PAHs ^b (ng/m ³)						
Naphthalene	0.23	0.1724	0.01	0.22	0.1129	0.03
Phenanthrene	0.17	0.0911	0.06	0.05	0.6813	0.01
Pyrene	0.07	0.5215	0.03	0.35	0.0363	0.03
Benzo[<i>a</i>]pyrene	0.20	0.2281	0.03	−0.09	0.5275	0.0006
Fine PM (µg/m ³)						
PM _{2.5}	0.99	0.02	0.98	−0.33	0.45	0.88

^a Values are based on log-transformed concentrations. *P* values are for the slope of each individual variable. *R*²s are the coefficients of determination of the mixed-effect model.

^b Sum of all the target compounds in both gas and particle phases.

Table 19c. Associations Between Personal and Ambient Air Toxic Concentrations by Day of the Week^a

	Weekdays			Weekend Days		
	Slope	<i>P</i> Value	<i>R</i> ²	Slope	<i>P</i> Value	<i>R</i> ²
VOCs (µg/m ³)						
MTBE	0.77	< 0.0001	0.75	0.82	< 0.0001	0.81
Hexane	0.70	< 0.0001	0.69	0.87	< 0.0001	0.93
Benzene	0.50	< 0.0001	0.68	0.40	< 0.0001	0.84
Toluene	0.41	< 0.0001	0.39	0.23	0.0227	0.49
<i>o</i> -Xylene	0.54	< 0.0001	0.54	0.12	0.2614	0.84
Aldehydes (µg/m ³)						
Formaldehyde	0.38	< 0.001	0.89	0.31	< 0.001	0.76
Acetaldehyde	0.63	< 0.001	0.68	0.44	< 0.001	0.82
PAHs ^b (ng/m ³)						
Naphthalene	0.04	0.7825	0.0006	0.18	0.1592	0.02
Phenanthrene	0.12	0.0995	0.05	0.27	0.0300	0.12
Pyrene	0.24	0.0218	0.04	0.23	0.2253	0.02
Benzo[<i>a</i>]pyrene	0.17	0.2314	0.08	−0.01	0.9316	0.03
Fine PM (µg/m ³)						
PM _{2.5}	1.26	0.01	0.86	−0.007	0.99	0.83

^a Values are based on log-transformed concentrations. *P* values are for the slope of each individual variable. *R*²s are the coefficients of determination of the mixed-effect model.

^b Sum of all the target compounds in both gas and particle phases.

weekdays. When the mixed-effects models were applied only to subjects who were not exposed to ETS, the same conclusions still held, namely that personal exposure to $PM_{2.5}$ was significantly associated with ambient $PM_{2.5}$ concentrations on summer weekdays, indicating that personal activities and air-exchange rates played a critical role in $PM_{2.5}$ exposure.

VOCs

Ambient VOCs A summary of the descriptive statistics for our measures of ambient VOC concentrations in Waterfront South and Copewood–Davis is shown in Table 12a. VOC concentrations for the two locations are shown by day of the week and season in Tables 13 and 14, respectively. Box plots of personal and ambient concentrations of selected VOCs (toluene, benzene, and hexane) are shown by day of the week and season in Figures 15 through 18. 1,3-butadiene

was not included in the report because of large uncertainties in the measurement method (Chung et al. 1999; Gordon et al. 1999). The percentage of sample detections above the MDL was 32% for styrene and greater than 60% for the other measured VOCs (Table 12a). Because of the low percentage for styrene (which, incidentally, was similar to that reported in RIOPA [29%]), styrene was presented only in our descriptive summary and was not used for further location comparisons.

TEX and *o*-xylene were present at significantly higher concentrations in Waterfront South than in Copewood–Davis, especially in summer (Table 12a, Table 14, and Figure 15). Mean \pm SD concentrations of toluene and *m*- & *p*-xylenes, for example, were $4.27 \pm 5.23 \mu\text{g}/\text{m}^3$ and $1.66 \pm 1.33 \mu\text{g}/\text{m}^3$, respectively, in Waterfront South, significantly higher ($P < 0.0005$) than those measured in Copewood–Davis ($2.34 \pm 1.80 \mu\text{g}/\text{m}^3$ for toluene and $1.19 \pm 0.93 \mu\text{g}/\text{m}^3$ for

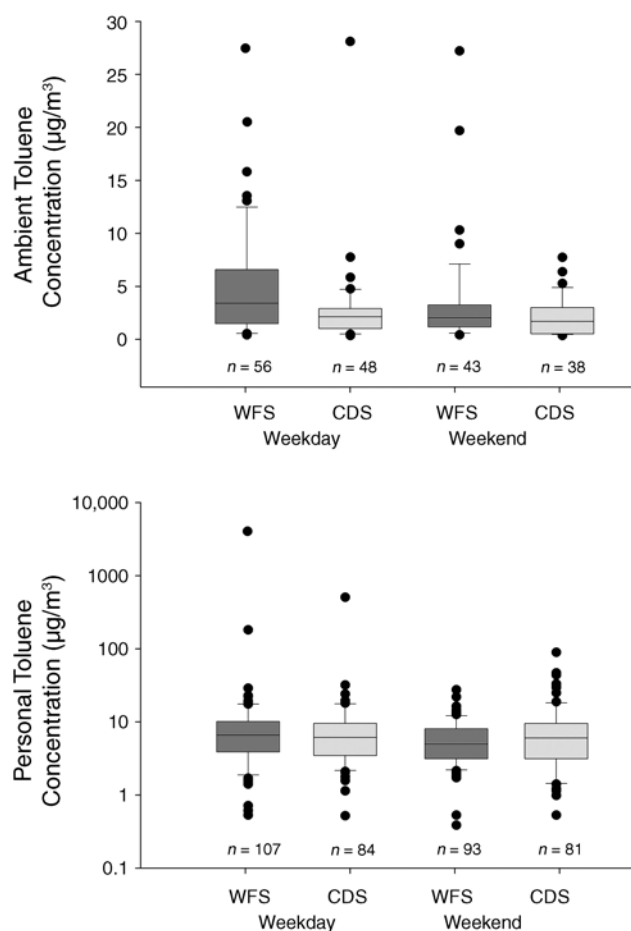


Figure 15. Ambient (top) and personal (bottom) toluene concentrations in Waterfront South (WFS) and Copewood–Davis (CDS) by day of the week. Shown on linear and logarithmic scales, respectively; n is the number of samples in each category.

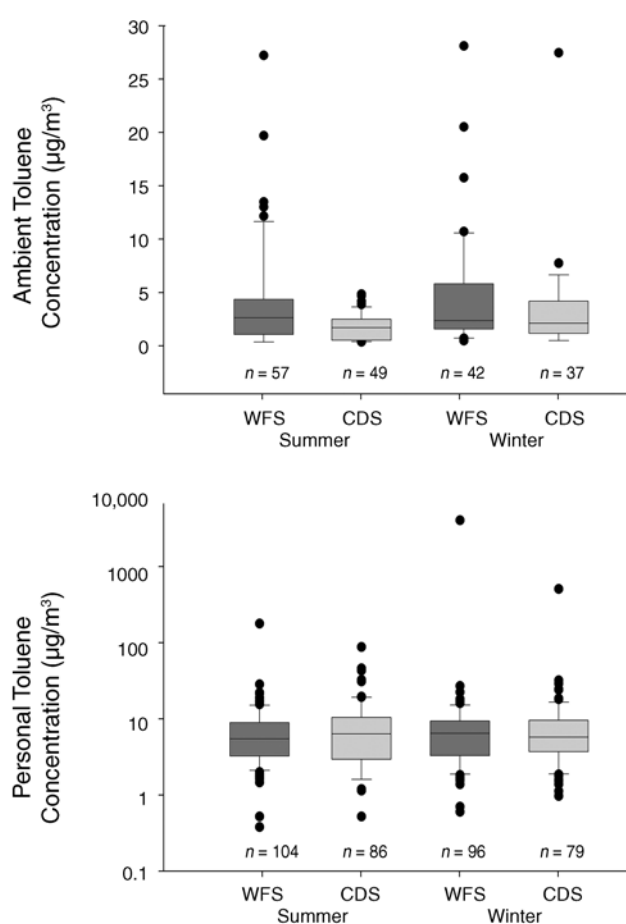


Figure 16. Ambient (top) and personal (bottom) toluene concentrations in Waterfront South (WFS) and Copewood–Davis (CDS) by season. Shown on linear and logarithmic scales, respectively; n is the number of samples in each category.

m- & *p*-xylenes) (Table 12a). The concentrations and variability of these compounds were also found to be significantly higher ($P < 0.05$) in Waterfront South than in Copewood–Davis during the spatial variation study, as described earlier. The results indicate the impact of local industrial sources on ambient TEX concentrations in Waterfront South. According to local-source emissions information collected by the NJDEP (Table 1, NJDEP 2005), more than 10 industrial facilities emit TEX in Waterfront South (Table 1), including paint applicators, metal-processing companies, and automobile repair shops. Further, the differences between the two neighborhoods were more obvious on weekdays than on weekend days, consistent with our hypothesis that larger numbers of Waterfront South industrial and commercial facilities operate on weekdays than on weekend days.

Mean \pm SD and median ambient concentrations were $2.18 \pm 3.01 \mu\text{g}/\text{m}^3$ and $1.20 \mu\text{g}/\text{m}^3$ for benzene and $2.31 \pm 2.76 \mu\text{g}/\text{m}^3$ and $1.53 \mu\text{g}/\text{m}^3$ for MTBE in Waterfront South (Table 12a, Figure 17). In Copewood–Davis they were $2.86 \pm 4.72 \mu\text{g}/\text{m}^3$ and $1.26 \mu\text{g}/\text{m}^3$ for benzene and $2.35 \pm 2.89 \mu\text{g}/\text{m}^3$ and $1.63 \mu\text{g}/\text{m}^3$ for MTBE. There were no statistically significant differences between the two neighborhoods in the concentrations calculated for the two VOCs ($P = 0.97$ for benzene and $P = 0.39$ for MTBE). Similar trends were found in our spatial variation study, although a hot spot was noted for MTBE. We conclude that, based on these observations, mobile sources are the main contributors to benzene and MTBE in these areas. In Copewood–Davis, which was downwind of local major roads on most sampling days, automobile exhaust and evaporative emissions from vehicles and gas stations are the principal

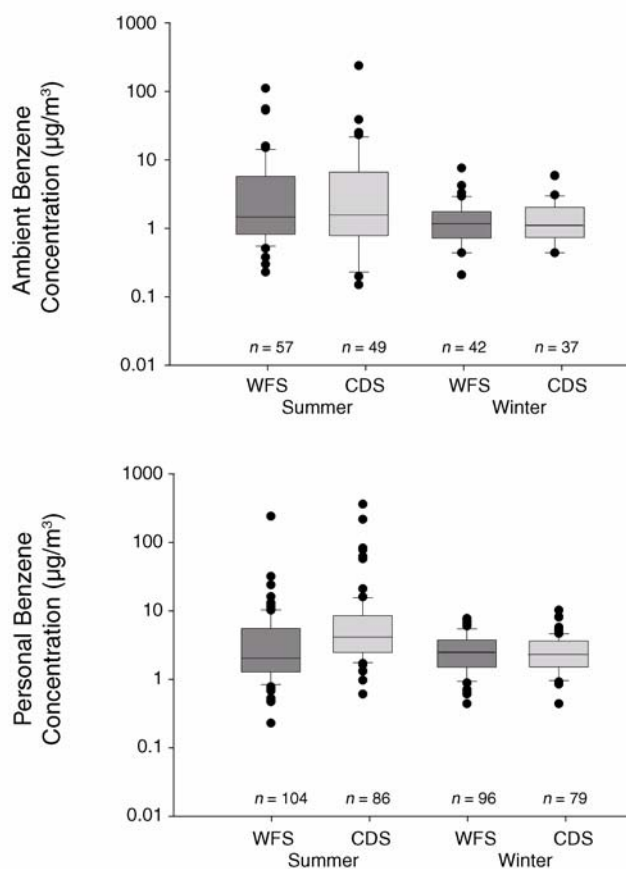


Figure 17. Ambient (top) and personal (bottom) benzene concentrations in Waterfront South (WFS) and Copewood–Davis (CDS) by season. Shown on logarithmic scales; *n* is the number of samples in each category.

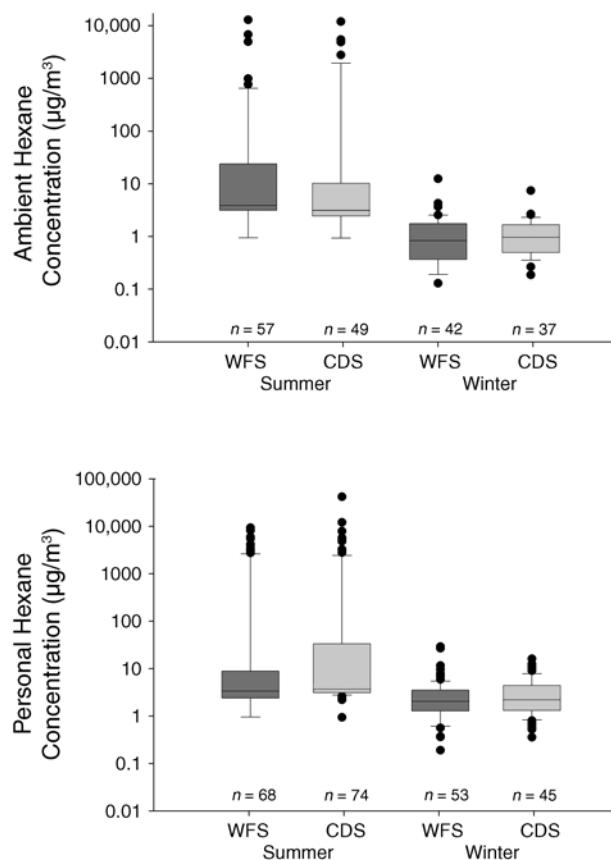


Figure 18. Ambient (top) and personal (bottom) hexane concentrations in Waterfront South (WFS) and Copewood–Davis (CDS) by season. Note that scales are logarithmic and differ between panels; *n* is the number of samples in each category.

sources of benzene and MTBE. This conclusion was confirmed by examining the BTEX ratio and correlation between BTEX and MTBE in the spatial variation study. Waterfront South, however, was upwind of these roads, meaning that the expected impact of local traffic would be small. It is worth noting that several hot spots for benzene and one for MTBE identified during the spatial variation study were not caught by the fixed monitoring sites used in the main exposure study. Results from the spatial variation study revealed that, in addition to moving automobiles, nonmobile vehicles and evaporative emissions from junked vehicles are also sources of ambient benzene and MTBE in Waterfront South.

A wide range of hexane concentrations was measured in both neighborhoods — 0.13 to 12,578 $\mu\text{g}/\text{m}^3$ in Waterfront South and 0.19 to 11,869 $\mu\text{g}/\text{m}^3$ in Copewood–Davis (Table 12a, Figure 18). Because of various suspiciously high values, the median concentrations (2.16 $\mu\text{g}/\text{m}^3$ for Waterfront South and 1.87 $\mu\text{g}/\text{m}^3$ for Copewood–Davis) were used for location comparison; they were not significantly different ($P = 0.54$). The NJDEP reported (2005) that hexane has been used as a solvent in several industrial facilities in or near Waterfront South, including metal-processing companies, automobile repair shops, and recycling plants. Extraordinarily high concentrations of hexane ($> 2000 \mu\text{g}/\text{m}^3$, Figure 18), including four measurements $> 10,000 \mu\text{g}/\text{m}^3$, were found in both neighborhoods on 7 sampling days in June and July 2005. To verify these values, the high-concentration samples were reanalyzed using new calibration standards, with no change in results. Solvent blanks, lab blanks, and field blanks obtained during the seven days were double-checked, with, again, no change in results. The use of hexane by certain industrial facilities identified by the NJDEP might have contributed to some extent to the high hexane concentrations. The concentrations found in Copewood–Davis on the sampling days were higher than those in Waterfront South, meaning that there are probably unidentified sources of hexane in or near Copewood–Davis.

Chlorinated compounds were present at essentially the same concentrations in the two neighborhoods, with mean \pm SD values of $0.19 \pm 0.26 \mu\text{g}/\text{m}^3$ for chloroform and $0.53 \pm 0.10 \mu\text{g}/\text{m}^3$ for carbon tetrachloride in Waterfront South and $0.20 \pm 0.25 \mu\text{g}/\text{m}^3$ and $0.54 \pm 0.10 \mu\text{g}/\text{m}^3$ in Copewood–Davis (Table 12a). These concentrations are consistent with U.S. nationwide ambient background concentrations (McCarthy et al. 2007; Touma et al. 2006). Therefore, although the NJDEP had originally suggested there were sources of ambient chlorinated compounds in Waterfront South, no significant industrial impacts were found on the ambient concentrations of chloroform or carbon tetrachloride.

To evaluate whether Waterfront South and Copewood–Davis were hot spots for VOCs, the ambient VOC concentrations measured in the two neighborhoods were compared with data collected by the NJDEP during the winter and summer months of 2004 to 2006 (i.e., the same sampling seasons as in our study) at air-toxic monitoring sites in Elizabeth, Camden, New Brunswick, and Chester, New Jersey. The Chester site represents the background concentrations of air pollution in the state. The concentrations from all sites are shown in Appendix Table B.1 (summer) and Table B.2 (winter) in Appendix B. For each compound, the sampling sites were ranked in descending order of the median concentrations. Box plots of ambient concentrations of benzene, MTBE, and toluene by season and location are shown in Figures 19 through 21.

Ambient concentrations of almost all the target VOCs in both seasons (except for chloroform in winter) were significantly higher ($P < 0.01$) in Waterfront South than at the NJDEP's Chester site. These results indicate the impact of VOCs from local sources in Waterfront South. The median concentrations of MTBE, benzene, toluene, and *m*- & *p*-xylenes were found to be significantly higher in Waterfront South than in New Brunswick ($P < 0.01$) in summer or winter, depending on the compound. Median concentrations of benzene in summer, for example, were 1.25 $\mu\text{g}/\text{m}^3$ in Waterfront South and 0.70 $\mu\text{g}/\text{m}^3$ in New Brunswick ($P < 0.001$), and median concentrations of toluene in winter were 2.37 $\mu\text{g}/\text{m}^3$ in Waterfront South and 1.66 $\mu\text{g}/\text{m}^3$ in New Brunswick ($P = 0.0054$). Ambient VOC concentrations in Waterfront South were found to be similar to those in Camden and Elizabeth, as shown in Figures 19 through 21 for benzene, MTBE, and toluene. Elizabeth and Camden are cities with both urban mobile and stationary sources.

Like Waterfront South, Copewood–Davis had significantly higher VOC concentrations than the Chester site ($P < 0.01$) for the majority of target compounds. Copewood–Davis, moreover, had significantly higher median concentrations of benzene than New Brunswick in summer (1.32 $\mu\text{g}/\text{m}^3$ versus 0.70 $\mu\text{g}/\text{m}^3$, $P = 0.0001$), and MTBE concentrations were also found to be significantly higher in Copewood–Davis than in New Brunswick ($P < 0.01$). These results indicate that Copewood–Davis is a hot spot for benzene and MTBE. Ambient concentrations of the majority of the VOCs were found to be significantly lower in Copewood–Davis than at the Elizabeth site.

Ambient VOC concentrations in Copewood–Davis, although measured at the same site in Camden as those of the NJDEP study, were found to be lower than the NJDEP's (Appendix Table B.2 in Appendix B). The concentrations of toluene, ethylbenzene, *o*-xylene, and styrene, for example,

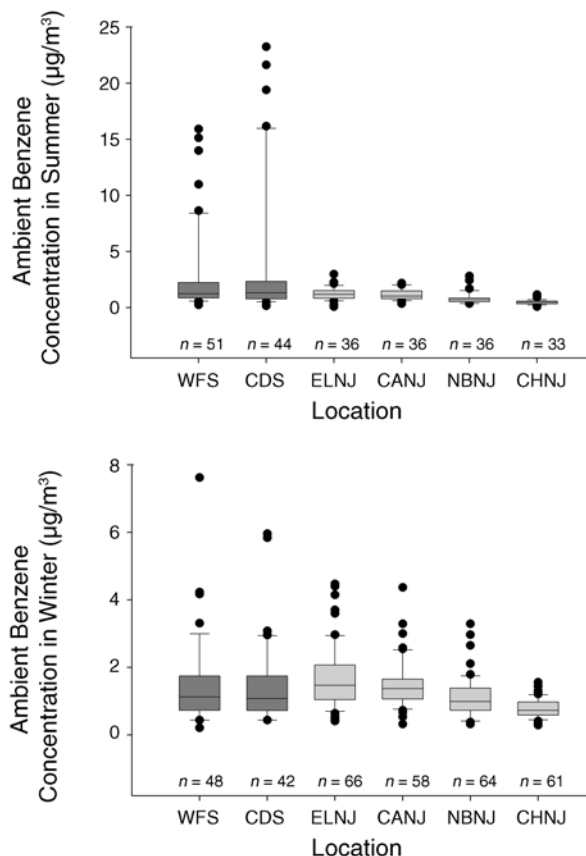


Figure 19. Ambient benzene concentrations by season in Waterfront South (WFS), Copewood–Davis (CDS), and U.S. EPA monitoring sites in Elizabeth (ELNJ), Camden (CANJ), New Brunswick (NBNJ), and Chester (CHNJ), New Jersey. Note that the y-axis scales differ; *n* is the number of samples in each category.

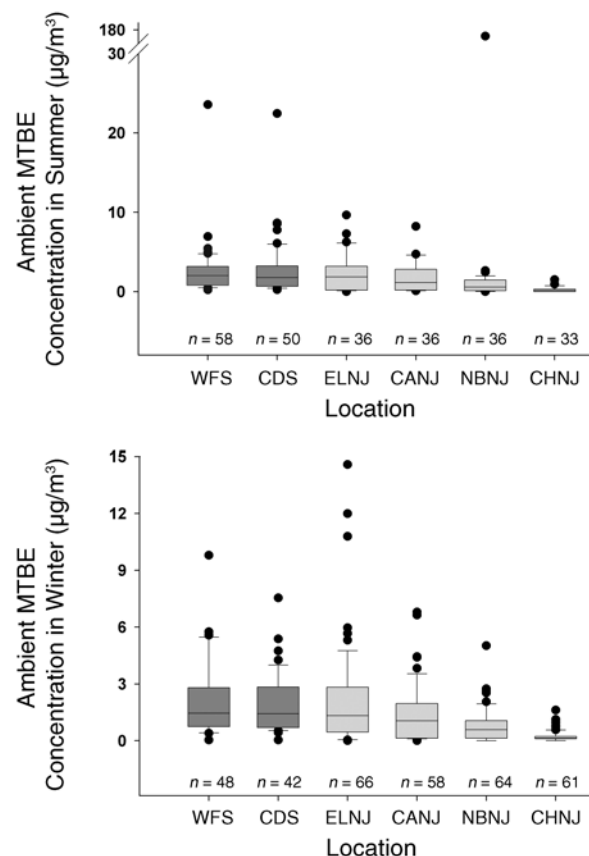


Figure 20. Ambient MTBE concentrations by season in Waterfront South (WFS), Copewood–Davis (CDS), and U.S. EPA monitoring sites in Elizabeth (ELNJ), Camden (CANJ), New Brunswick (NBNJ), and Chester (CHNJ), New Jersey. Note that the y-axis scales differ; *n* is the number of samples in each category.

were significantly lower in our Copewood–Davis measurements than those measured by the NJDEP. This was probably caused by the different study designs and measurement methods of the two studies. Between 2004 and 2006, only 13 of our sampling dates coincided with the NJDEP's, and the two 24-hr sampling periods for the measurements did not fall in the same time window: Our sampling period usually started during the early morning or late afternoon (for the subjects' convenience), and the NJDEP's sampling period ran from midnight to midnight. In addition, we collected VOCs using OVM passive sampling badges, and the NJDEP used the canister (TO-15) method. As Pratt and colleagues (2005) found, benzene is generally overestimated and chlorinated compounds and styrene are consistently underestimated by the OVM method compared with the canister method. Other aromatic compounds (e.g., TEX) tend to be underestimated by OVM badges in the low range of concentrations (Mukerjee

et al. 2004; Pratt et al. 2005). Given these concerns, the ambient concentrations of most VOC species measured in Waterfront South and Copewood–Davis might in fact have been underestimated.

The mean concentrations of most target VOCs measured in Waterfront South and Copewood–Davis were compared with data reported in the literature. The differences in measured concentrations between our study and others varied by compound. For benzene, for example, ambient mean \pm SD concentrations were $2.18 \pm 3.01 \mu\text{g}/\text{m}^3$ in Waterfront South and $2.86 \pm 4.72 \mu\text{g}/\text{m}^3$ in Copewood–Davis (Table 12a); both were similar to the means reported in RIOPA ($2.17 \pm 2.09 \mu\text{g}/\text{m}^3$) and TEACH ($1.31 \mu\text{g}/\text{m}^3$ in summer and $2.55 \mu\text{g}/\text{m}^3$ in winter) (Kinney et al. 2002). For MTBE the ambient mean \pm SD concentrations were $2.31 \pm 2.76 \mu\text{g}/\text{m}^3$ in Waterfront South and $2.35 \pm 2.89 \mu\text{g}/\text{m}^3$ in Copewood–Davis; these were lower than the mean reported in RIOPA ($7.76 \pm 8.83 \mu\text{g}/\text{m}^3$). However, the

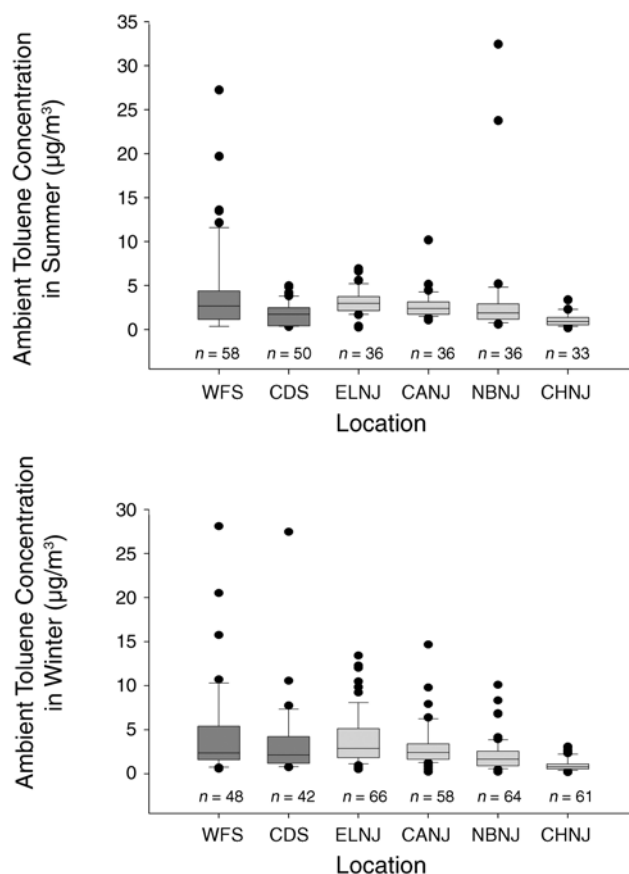


Figure 21. Ambient toluene concentrations by season in Waterfront South (WFS), Copewood–Davis (CDS), and U.S. EPA monitoring sites in Elizabeth (ELNJ), Camden (CANJ), New Brunswick (NBNJ), and Chester (CHNJ), New Jersey. *n* is the number of samples in each category.

maximum concentrations and frequencies of high concentrations occurring in our study were much higher than those reported previously. As mentioned above, hexane concentrations $> 2000 \mu\text{g}/\text{m}^3$ were measured on seven sampling days in July 2005 in both neighborhoods. In addition, benzene concentrations were $> \sim 10 \mu\text{g}/\text{m}^3$ on 7 sampling days ($\sim 8\%$ of the total sampling days) in both locations. On the highest benzene day, concentrations of toluene, ethylbenzene, and xylenes were relatively low (0.2 to $0.9 \mu\text{g}/\text{m}^3$), which suggested that the benzene peak was not caused by automotive sources but probably by stationary sources located in or near the two neighborhoods. These episodes of high benzene and hexane emissions raised the average concentrations of both significantly in both neighborhoods and exceeded the chronic minimal risk levels for inhalation exposure to the two compounds (i.e., $9.6 \mu\text{g}/\text{m}^3$ for benzene and $2110 \mu\text{g}/\text{m}^3$ for hexane [ATSDR 2010]). These high exposures might be a health concern and the subjects of future health studies.

Concentrations of chloroform and carbon tetrachloride were similar to those measured in other urban areas, such as the $0.2 \pm 0.2 \mu\text{g}/\text{m}^3$ for chloroform and $0.7 \pm 0.1 \mu\text{g}/\text{m}^3$ for carbon tetrachloride in New York City and $0.1 \pm 0.1 \mu\text{g}/\text{m}^3$ and $0.5 \pm 0.1 \mu\text{g}/\text{m}^3$ in Los Angeles reported by Sax and colleagues (2004). Similar concentrations of the two compounds were also observed at the 38 saturation-sampling sites in our spatial variation study (Table 20). These results indicate that regional emissions could be the main sources of chloroform and carbon tetrachloride in Waterfront South and Copewood–Davis.

The differences between ambient VOCs collected on weekdays and weekend days were examined to explore the impact of local industrial activities and mobile sources on VOCs in the two neighborhoods (Table 13). The concentrations of benzene, toluene, ethylbenzene, and xylenes measured on weekdays were 16% to 46% higher than those measured on weekend days in Waterfront South, but the differences were not statistically significant ($P > 0.05$). These trends would be consistent with there being higher traffic volumes and more industrial and commercial facilities in operation on weekdays than on weekend days, as hypothesized. In Copewood–Davis only toluene and xylenes were found to be higher (between 3% and 30%) on weekdays than on weekend days; these differences were not statistically significant either ($P > 0.05$).

Seasonal variability in VOCs was also examined in the two neighborhoods and was found to differ by compound and location. As shown in Table 14 and Figures 16 through 18, TEX concentrations were higher in winter than in summer in both Waterfront South and Copewood–Davis; the difference was statistically significant in Copewood–Davis ($P < 0.05$) but not in Waterfront South ($P > 0.05$). These results were consistent with those of previous studies (Weisel et al. 2005). The seasonal variability associated with the compounds probably had two main causes (Weisel et al. 2005; Touma et al. 2006). First, compounds can undergo photochemical reactions in the atmosphere in summer, which consumes a large amount of VOCs. Conversely, photochemical activity is low in winter, with the result that various VOC concentrations are higher in winter than in summer. Second, a large fraction of TEX compounds comes from automobile exhaust. Automobiles require a longer warm-up time in winter, emitting more exhaust containing these compounds and other air toxics and thus causing higher concentrations of these pollutants in winter than in summer. The seasonal difference was not significant in Waterfront South, probably because the ambient TEX concentrations were dominated by stationary sources rather than automobile sources, leading to lower seasonal variations. The opposite trend was observed for

Table 20. Descriptive Summary of VOC and Aldehyde Concentrations at 22 Sites in Waterfront South and 16 Sites in Copewood–Davis^a

	Waterfront South					Copewood–Davis				
	Mean	SD	RSD (%)	Range	Mean	SD	RSD (%)	Range	P_1	P_2
July 20–22, 2005										
MTBE	5.9	5.5	92	3.0–29	5.5	1	19	4.3–7.8	0.78	0.37
Chloroform	0.23	0.02	9	0.21–0.29	0.22	0.02	8	0.20–0.26	0.10	0.59
Carbon tetrachloride	0.57	0.02	3	0.54–0.60	0.56	0.03	5	0.51–0.60	0.25	0.42
Benzene	1.3	0.36	28	0.79–2.6	1.4	0.31	22	1.1–2.2	0.35	0.99
Toluene	7.5	6.3	83	3.7–32	4	1.4	36	3.1–8.2	0.06	0.06
Ethylbenzene	0.93	0.47	50	0.44–2.5	0.65	0.16	25	0.48–1.1	0.04	0.04
<i>m- & p</i> -Xylenes	2.8	1.7	60	1.2–8.9	1.9	0.49	26	1.4–3.1	0.07	0.02
<i>o</i> -Xylene	0.86	0.52	60	0.41–2.9	0.61	0.16	26	0.45–1.0	0.1	0.05
Formaldehyde	28	12	42	0.50–40	28	7.9	28	14–42	0.62	0.81
Acetaldehyde	17	11	65	1.7–33	13	10	83	0.07–32	0.45	0.59
Acrolein	0.26	0.24	92	0.13–1.1	1.8	1.1	61	0.16–3.9	< 0.0001	0.0005
August 17–18, 2005										
MTBE ^b	16	32	196	4.9–159	9.4	2.7	29	6.4–18	0.41	0.16
Chloroform	0.14	0.02	13	0.11–0.19	0.15	0.02	16	0.13–0.23	0.34	0.61
Carbon tetrachloride	0.45	0.02	4	0.42–0.51	0.46	0.02	5	0.41–0.49	0.23	0.69
Benzene	0.97	0.53	54	0.50–3.1	0.86	0.21	24	0.67–1.4	0.43	0.06
Toluene	7.6	12	164	1.9–60	2.5	1.2	49	1.9–6.8	0.13	0.008
Ethylbenzene	1.1	1.9	176	0.30–9.2	0.42	0.11	26	0.32–0.74	0.19	0.05
<i>m- & p</i> -Xylenes	3.1	5.4	175	0.81–25	1.2	0.33	29	0.85–2.1	0.18	0.006
<i>o</i> -Xylene	0.87	1.2	134	0.29–4.9	0.42	0.11	27	0.31–0.73	0.15	0.02
Formaldehyde	31	7.8	25	20–44	30	7	24	16–40	0.75	0.65
Acetaldehyde	28	27	98	0.13–105	24	22	93	0.12–62	0.86	0.38
Acrolein	3.9	1.6	42	1.31–8.25	3.6	1.4	39	1.4–6.4	0.57	0.79
Dec. 20–22, 2005										
MTBE	1.4	1.1	76	0.49–5.7	1.4	0.33	24	0.90–1.9	0.91	1
Chloroform	0.09	0.02	24	0.03–0.15	0.08	0	4	0.07–0.09	0.08	0.08
Carbon tetrachloride	0.51	0.08	16	0.19–0.56	0.51	0.02	3	0.49–0.55	0.95	0.12
Benzene	1.3	0.27	21	0.54–1.8	1.2	0.15	12	1.0–1.6	0.44	0.49
Toluene	5.2	4.9	94	1.9–23	2.3	0.51	22	1.7–3.5	0.04	0.007
Ethylbenzene	0.54	0.19	36	0.32–0.99	0.46	0.08	17	0.33–0.62	0.16	0.06
<i>m- & p</i> -Xylenes	1.5	0.59	40	0.87–3.2	1.3	0.27	21	0.88–1.9	0.19	0.16
<i>o</i> -Xylene	0.47	0.18	39	0.24–1.1	0.42	0.09	20	0.30–0.62	0.35	0.16
Formaldehyde	12	2.6	21	8.4–16	12	2.4	20	7.8–17	0.96	0.24
Acetaldehyde	8	8.6	107	0.07–23	4.4	4.8	108	0.07–13	0.90	0.09
Acrolein	0.48	0.57	119	0.02–2.1	0.40	0.41	102	0.02–1.5	0.32	0.36

^a Percentage CV is for duplicate samples collected at sites W8 and C13. P_1 is the P value applied to the concentration comparison between Waterfront South and Copewood–Davis by Wilcoxon rank sum test. P_2 is the variability comparison by Wilcoxon rank sum test.

^b MTBE concentrations from the August sampling campaign might be overestimated by a factor of three, because of a suspiciously lower response factor for MTBE compared with that of other calibration data.

N indicates the number of functioning sample sites.

MTBE, benzene, and hexane — i.e., higher mean and median concentrations were found in summer than in winter in both Waterfront South and Copewood–Davis. These results might suggest that evaporation from automotive and stationary sources (such as emissions of benzene and MTBE from a Waterfront South car scrapping facility [which would be expected to have higher evaporative emissions in summer]) contributed significantly to ambient concentrations of these compounds in summer. The Wintertime Oxygenated Fuels Program was repealed in New Jersey in the late 1990s, and MTBE is now added to gasoline as a fuel oxygenate to reduce emissions of carbon monoxide and ozone formation. Today, reformulated gasoline (2% oxygen by weight, equivalent to 11% MTBE by volume) is used throughout the year in New Jersey.

Personal VOCs A summary of the descriptive statistics for our measures of personal VOC concentrations is shown in Table 15a. Concentrations by day of the week and by season are presented in Table 16, Table 17, and Figures 15 through 18. More than 70% of the observations for all target compounds were above the MDL (except those for styrene, at 55% in Waterfront South and 69% in Copewood–Davis) (Table 15a). Mean \pm SD personal toluene concentrations were found to be higher in Waterfront South ($28.0 \pm 286 \mu\text{g}/\text{m}^3$) than in Copewood–Davis ($11.4 \pm 40.0 \mu\text{g}/\text{m}^3$), but the difference was caused primarily by several high values measured in Waterfront South (Figure 15). No significant difference ($P > 0.05$, Tables 15 and 16) was found when comparing the median concentrations of toluene obtained for the two locations. Mean and median personal concentrations of ethylbenzene and xylenes in Copewood–Davis were slightly higher than those in Waterfront South, but the difference was not significant (minimum P value was 0.16). Unlike the other VOCs, TEX has both indoor and outdoor sources, and personal exposure can therefore be affected not only by ambient sources, but also by nonambient sources (e.g., in indoor and in-vehicle environments) and by personal activities. Analysis was performed to examine the impact of ambient air pollution of these compounds on personal exposure; detailed results are presented below.

Like ambient concentrations of benzene and MTBE, personal concentrations of benzene and MTBE were found to be higher in Copewood–Davis than in Waterfront South (Table 15). For MTBE the difference was not significant ($P = 0.22$). For benzene the differences for both mean and median concentrations were statistically significant ($P < 0.01$). Personal benzene concentrations were also higher in Copewood–Davis than in Waterfront South by day of the week and season, except in winter (Tables 16 and 17). The higher concentrations observed in Copewood–Davis

compared with Waterfront South were probably caused by some combination of higher ETS exposures, higher automobile traffic volumes, and Copewood–Davis's downwind position relative to local major roads. It is well known that MTBE is found primarily in gasoline, and BTEX can be generated by both ETS and automobile exhaust.

We made 234 personal nicotine measurements; the mean nicotine concentration was significantly higher ($P < 0.01$) in Copewood–Davis ($0.61 \pm 0.17 \mu\text{g}/\text{m}^3$) than in Waterfront South ($0.26 \pm 0.16 \mu\text{g}/\text{m}^3$). The contribution from ETS exposure to personal VOCs was thus possibly greater in Copewood–Davis as well. Nicotine can be adsorbed onto indoor surfaces and re-emitted to the air, meaning that the measurements would reflect both fresh and historical ETS. However, controlling for ETS exposure did not change the trend; personal benzene and MTBE concentrations remained higher in Copewood–Davis than in Waterfront South. Furthermore, as mentioned earlier, Copewood–Davis is typically downwind of I-676 and NJ-168, and about 80% of subjects from Copewood–Davis lived within three blocks ($< 500 \text{ m}$) of NJ-168. Traffic emissions might thus have a greater impact on personal exposures to benzene and other traffic-related VOCs in Copewood–Davis than in Waterfront South.

As noted in the $\text{PM}_{2.5}$ section, we measured 24-hr integrated personal exposures. Occupational and out-of-neighborhood exposures might have contributed to the personal concentrations of many of the VOCs. To assess this, our time diary and activity questionnaire both had questions designed to record potential exposures from various microenvironments. According to the questionnaire information, 37 of the 107 participants had full- or part-time jobs or were self-employed. Ten had exposures to VOCs at work. Three drivers (including a bus driver and a truck driver) and a security guard might have had exposures to traffic-related air toxics. Three participants involved with warehouse packing might have been exposed to VOCs emitted from packing materials. Two had a job related to the use of solvents. The concentrations measured for these subjects on working days were not significantly higher than on nonworking days or than the mean personal exposures measured for the remaining subjects. The occupational exposures of these participants, then, did not affect the overall distribution of personal exposures obtained for the study.

Extremely high individual personal concentrations of toluene, hexane, and benzene — probably reflecting occupational exposures — were measured in some samples collected in both neighborhoods. Personal concentrations of hexane, for example, peaked in the sampling period between the end of June 2005 and the end of July 2005. In

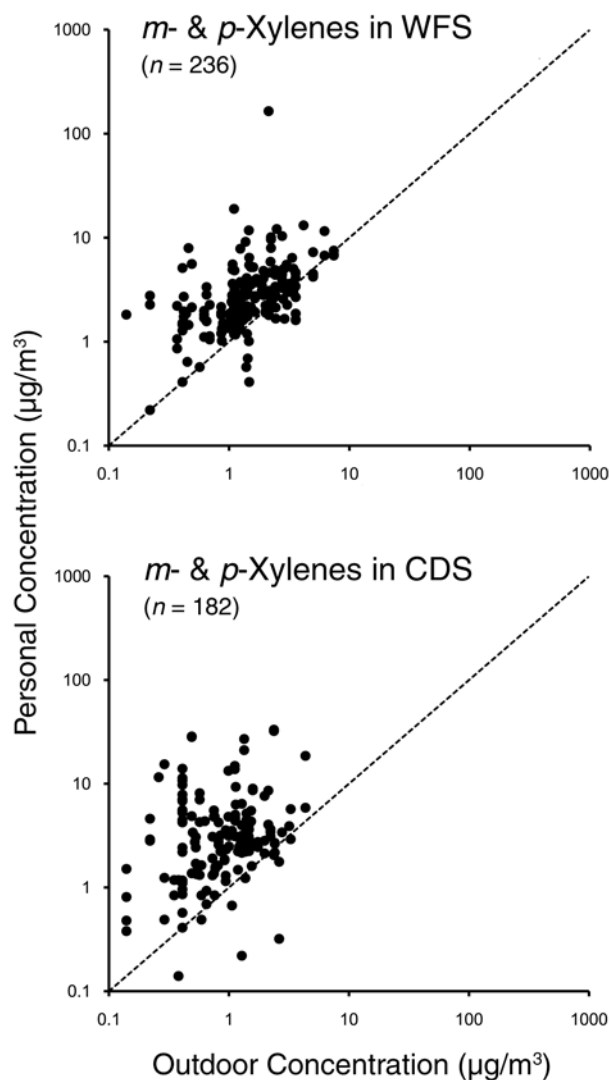


Figure 22. Scatter plots of personal versus ambient *m- & p*-xylene concentrations in (top) Waterfront South (WFS) and (bottom) Copewood-Davis (CDS). Shown on logarithmic scales; *n* is the number of samples in each category.

more than 20 personal samples, hexane concentrations of more than $2,300 \mu\text{g}/\text{m}^3$ were detected, with a high of $42,287 \mu\text{g}/\text{m}^3$ in Copewood-Davis. Similarly, high individual personal benzene concentrations ($> 200 \mu\text{g}/\text{m}^3$) were found in both locations, including two instances in Copewood-Davis and one in Waterfront South. These benzene concentrations were more than an order of magnitude higher than the personal benzene averages reported in previous studies, such as RIOPA (Weisel et al. 2005) and TEACH (Kinney et al. 2002). Our mean concentrations, however, were similar to those of RIOPA and TEACH. No ETS exposure was associated with the three outlier measures, to judge from questionnaire information. The elevated personal

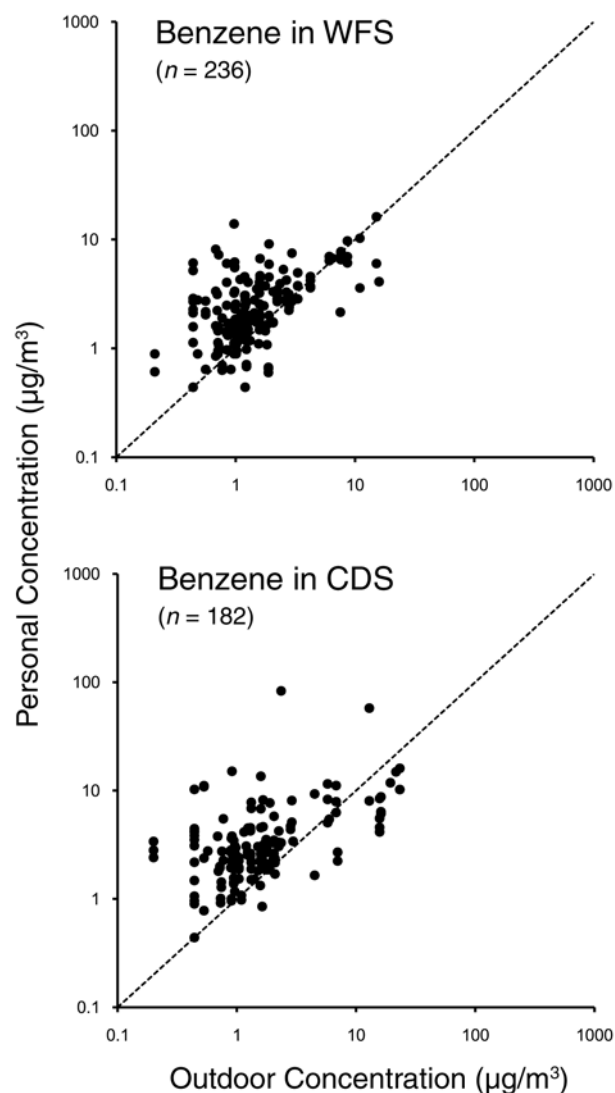


Figure 23. Scatter plots of personal versus ambient benzene concentrations in (top) Waterfront South (WFS) and (bottom) Copewood-Davis (CDS). Shown on logarithmic scales; *n* is the number of samples in each category.

concentrations of hexane and benzene during those sampling periods were probably associated with exposures to ambient and occupational sources in both neighborhoods. This supposition is supported by the fact that the high personal measurements all coincided with the highest ambient measurements. Moreover, according to information in the time-activity diaries, subjects who spent more time outdoors than other subjects measured on the same day tended to have higher personal VOC exposures. Five subjects in Waterfront South and Copewood-Davis, for example, who were monitored simultaneously on July 9, 2005 — when the highest hexane peak occurred — spent 30% to 50% of their time outdoors and had two to eight times higher

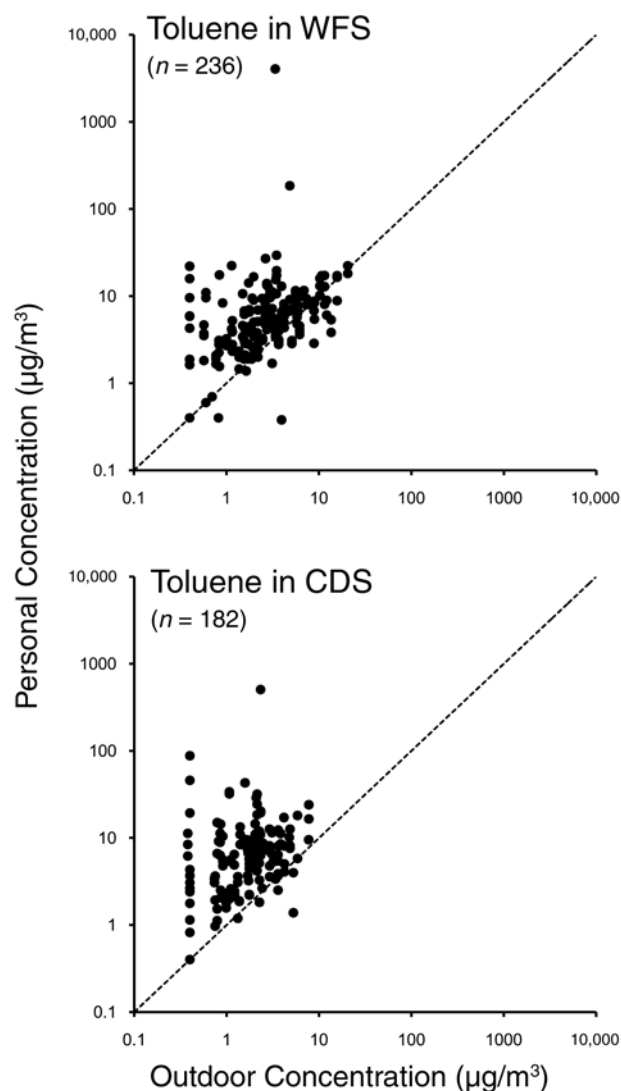


Figure 24. Scatter plots of personal versus ambient toluene concentrations in (top) Waterfront South (WFS) and (bottom) Copewood-Davis (CDS). Shown on logarithmic scales; n is the number of samples in each category.

personal hexane exposures than subjects in the same locations who only spent 2% to 3% of their time outdoors. This suggests that ambient VOC pollution (and outdoor occupational exposure) might be a significant contributor to personal VOC exposures when ambient sources are active in the area.

Personal and Ambient VOC Relationships Scatter plots of personal versus ambient concentrations of xylenes, benzene, toluene, MTBE, and hexane, which have local sources, are shown in Figures 22 through 26. Generally, the data from Waterfront South stayed close to the 1:1 line; the data from Copewood-Davis tended to scatter above it

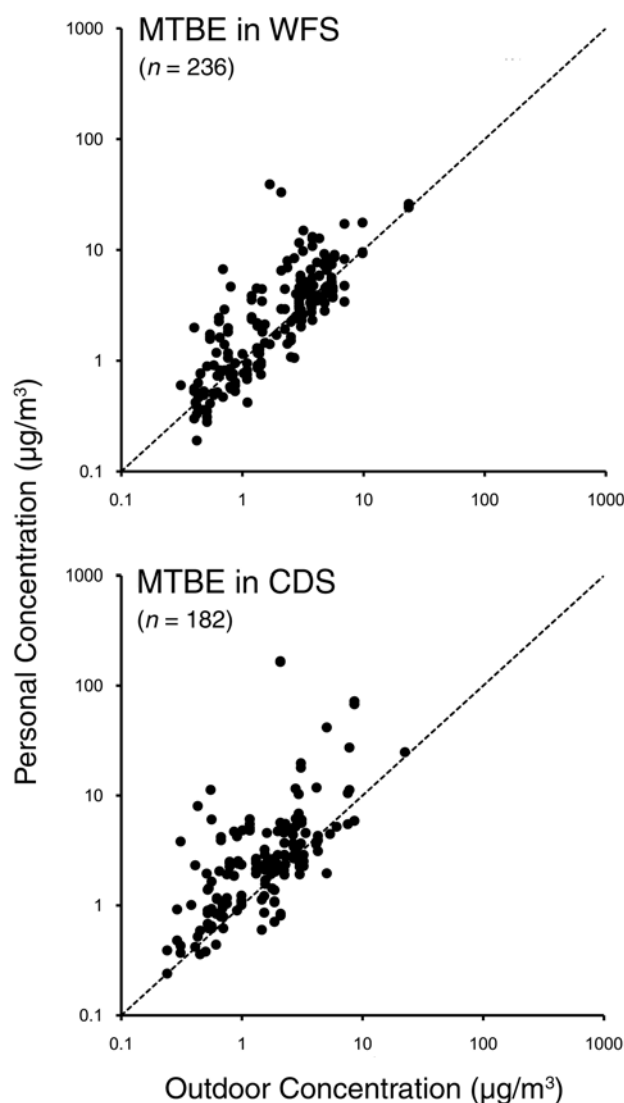


Figure 25. Scatter plots of personal versus ambient MTBE concentrations in (top) Waterfront South (WFS) and (bottom) Copewood-Davis (CDS). Shown on logarithmic scales; n is the number of samples in each category.

(see, for example, *m*- & *p*-xylenes in Figure 22). These observations suggest, again, that personal exposures to VOCs in Waterfront South were closely related to ambient pollution concentrations in the neighborhood. In Copewood-Davis personal exposures to VOCs were also influenced by indoor sources and personal activities. Benzene (Figure 23) and toluene (Figure 24) measurements were randomly distributed around the 1:1 line, but more data points were above the line, indicating that the contribution of personal exposure to the two compounds came from both ambient and indoor sources. The majority of the data points for MTBE (Figure 25) and hexane (Figure 26) fell along the 1:1 line, indicating that personal exposures to

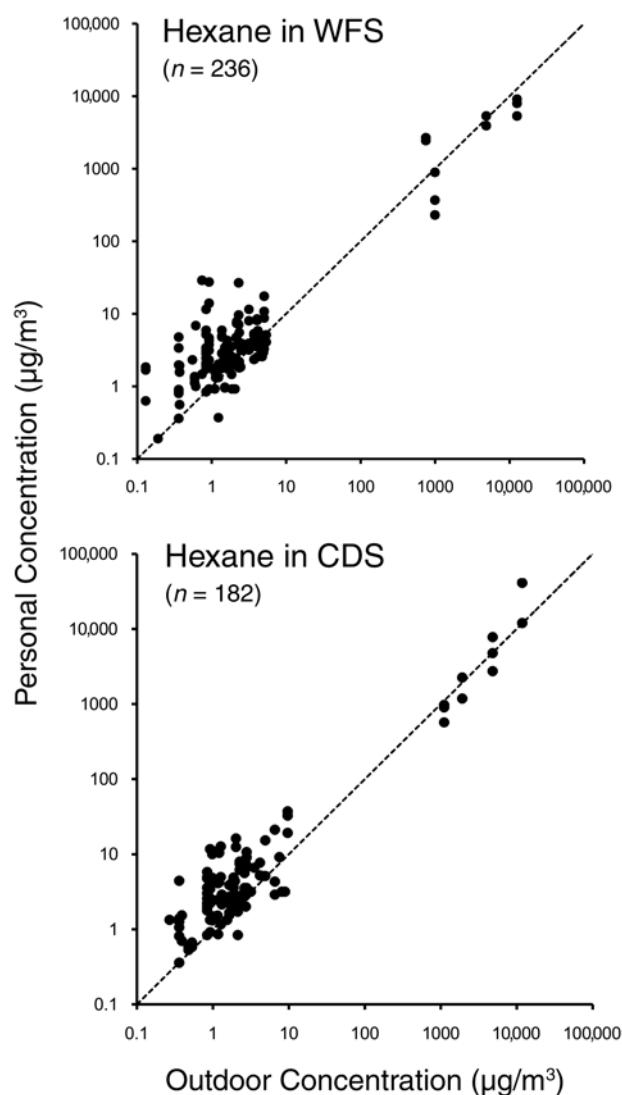


Figure 26. Scatter plots of personal versus ambient hexane concentrations in (top) Waterfront South (WFS) and (bottom) Copewood-Davis (CDS). Shown on logarithmic scales; n is the number of samples in each category.

MTBE and hexane were influenced predominantly by ambient sources in both neighborhoods.

The R 's for personal and ambient VOC concentrations are summarized in Table 18. The R 's for all the target compounds in Waterfront South were either higher than or similar to those in Copewood-Davis, indicating a higher impact of ambient pollution on personal exposures to VOCs in Waterfront South than in Copewood-Davis. Strong correlations between personal exposure and ambient concentrations were observed for MTBE ($R = 0.82$ in Waterfront South and 0.65 in Copewood-Davis), hexane (0.68 and 0.66), and carbon tetrachloride (0.73 and 0.85). These observations were consistent with the sources of

these pollutants in both areas. MTBE is generated primarily from tailpipe emissions and the evaporation of gasoline; hexane is used as a solvent in several industries in Waterfront South (NJDEP 2005), which might contribute to the personal and ambient concentrations; and carbon tetrachloride comes mainly from regional sources. The R 's for BTEX ranged from 0.47 to 0.56 in Waterfront South, suggesting moderate contributions from ambient sources to the personal exposures. The R 's for BTEX were relatively low in Copewood-Davis (0.26 to 0.45), suggesting that both indoor and ambient sources contributed to the personal exposures to these compounds. Further, the R 's for compounds with dominant ambient sources, such as MTBE, hexane, and benzene, were higher in summer than in winter for both neighborhoods. The R for MTBE, for example, was 0.83 in Waterfront South and 0.79 in Copewood-Davis in summer and decreased to 0.77 and 0.43 , respectively, in winter. This is consistent with the personal-ambient association for $PM_{2.5}$ reported above. These results again suggest that people might have had higher exposures to ambient air pollution in summer than in winter because they spent more time outdoors and the indoor air-exchange rate was higher in summer.

The associations between personal and ambient VOC concentrations were also analyzed using the mixed-effects model and slope; R^2 and P values are shown in Table 19a by location, in Table 19b by season, and in Table 19c by day of the week. Five VOCs associated with emissions from major types of sources were selected for analysis, namely MTBE (which is generated primarily from ambient automotive sources) and hexane, benzene, toluene, and *o*-xylene (which have both outdoor and indoor sources, including industrial operations, automobile exhaust, ETS, and household products). Personal exposures to these five VOCs were found to be closely associated with ambient VOCs concentrations ($P < 0.001$) for both Waterfront South and Copewood-Davis (Table 19a). The R^2 's were 0.92 , 0.86 , 0.62 , and 0.51 for MTBE, benzene, *o*-xylene, and toluene, respectively, in Waterfront South; 0.80 , 0.80 , 0.81 , and 0.50 for the same four compounds, respectively, in Copewood-Davis; and 0.96 for hexane in both locations. These results show that more than 50% of the variation in personal exposures could be explained by changes in ambient VOC concentrations.

Because BTEX can be found in ETS, the relationship between personal and ambient VOC concentrations was re-evaluated considering the possible confounding effects of ETS exposure. Nicotine measurements were included in the analysis for the subgroup of subjects that had been measured for nicotine concentrations. There was no change in the regression coefficient or P value when

nicotine was included in the analysis for Waterfront South, reaffirming the impact of ambient VOC sources on personal exposures in the neighborhood. However, when nicotine was included in the analysis for Copewood–Davis, the association became less significant; the *P* values changed from 0.0002 to 0.0458 for *o*-xylene and 0.0010 to 0.0877 for toluene. The regression coefficients for the ambient VOC concentrations decreased as well, from 0.56 to 0.44 for benzene, 0.39 to 0.25 for toluene, and 0.51 to 0.38 for *o*-xylene. These results indicate that exposure to ETS contributed significantly to personal VOCs for subjects in Copewood–Davis, causing a bias in personal exposures in Waterfront South compared with those in Copewood–Davis.

Aldehydes

Ambient Aldehydes A summary of the descriptive statistics for our measures of ambient concentrations of aldehydes in the two neighborhoods is shown in Table 12a.

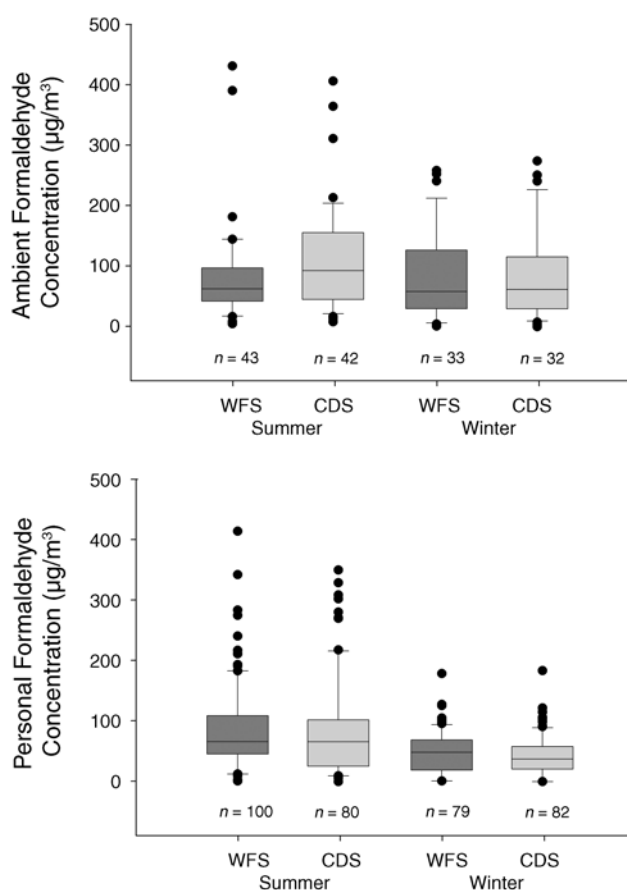


Figure 27. Ambient (top) and personal (bottom) formaldehyde concentrations in Waterfront South (WFS) and Copewood–Davis (CDS) by season. *n* is the number of samples in each category.

The distribution of ambient aldehyde concentrations by day of the week is shown in Table 13, and the seasonal distribution is shown in Table 14 and Figures 27 and 28 (only for formaldehyde and acetaldehyde). In Waterfront South formaldehyde and acetaldehyde were the most abundant ambient aldehydes measured; mean \pm SD and median concentrations were 20.2 ± 19.5 µg/m³ and 15.5 µg/m³, respectively, for formaldehyde and 12.5 ± 18.0 µg/m³ and 6.1 µg/m³, respectively, for acetaldehyde. Concentrations for acrolein and propionaldehyde were much lower; mean \pm SD and median concentrations were 0.8 ± 0.9 µg/m³ and 0.2 µg/m³, respectively, for acrolein and 1.1 ± 1.4 µg/m³ and 0.4 µg/m³, respectively, for propionaldehyde. Ambient concentrations at Copewood–Davis followed the same pattern; mean \pm SD and median concentrations were

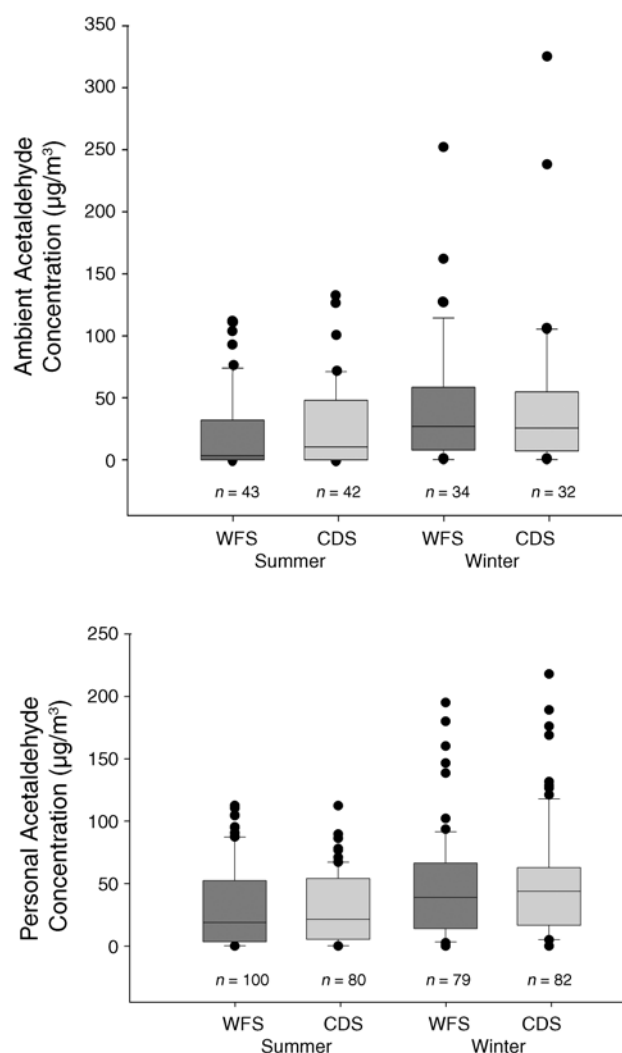


Figure 28. Ambient (top) and personal (bottom) acetaldehyde concentrations in Waterfront South (WFS) and Copewood–Davis (CDS) by season. *n* is the number of samples in each category.

$24.8 \pm 20.8 \mu\text{g}/\text{m}^3$ and $20.4 \mu\text{g}/\text{m}^3$ for formaldehyde, $14.6 \pm 21.9 \mu\text{g}/\text{m}^3$ and $8.3 \mu\text{g}/\text{m}^3$ for acetaldehyde, $0.6 \pm 0.9 \mu\text{g}/\text{m}^3$ and $0.2 \mu\text{g}/\text{m}^3$ for acrolein, and $1.6 \pm 2.4 \mu\text{g}/\text{m}^3$ and $0.4 \mu\text{g}/\text{m}^3$ for propionaldehyde.

Differences by location were examined only for formaldehyde and acetaldehyde, because the propionaldehyde and acrolein measured in ~40% of our samples were below the MDLs. Using the mixed-effects model (with log-transformed data), location was found to be a significant factor ($P = 0.03$) affecting ambient concentrations of formaldehyde ($24.8 \mu\text{g}/\text{m}^3$ in Copewood–Davis versus $20.2 \mu\text{g}/\text{m}^3$ in Waterfront South) but to be insignificant for acetaldehyde ($P = 0.5$) (Table 12a). These differences were opposite those found in the spatial variation study presented earlier (Table 20), where all aldehydes, except for acrolein in July, were found at the same or slightly higher concentrations in Waterfront South than in Copewood–Davis. This difference suggests that the measurements of aldehydes at the fixed monitoring site in Waterfront South might have underestimated the outdoor aldehyde concentrations at various locations in the neighborhood.

To examine the effects of season or day of the week, combined data from the two locations were used for analysis (because the stratified data by location did not converge). Season (summer versus winter) was found to be a significant factor affecting both formaldehyde and acetaldehyde concentrations. The summer mean ambient formaldehyde concentration ($24.2 \mu\text{g}/\text{m}^3$ for the combined data from the two locations) was marginally higher ($P = 0.09$) than the winter mean ambient concentration ($20.8 \mu\text{g}/\text{m}^3$); the opposite result was found for acetaldehyde ($10.3 \mu\text{g}/\text{m}^3$ in summer versus $18.3 \mu\text{g}/\text{m}^3$ in winter, $P = 0.02$). Seasonal variations in ambient aldehyde measurements have been reported in many previous studies (Anderson et al. 1996; Christensen et al. 2000; Sin et al. 2001; Ho et al. 2002). However, the trend varied from one study to another, reflecting the complexity of the sources, sinks, and atmospheric behaviors of aldehydes as well as local meteorologic conditions, including light intensity, temperature, and wind direction. In ambient air, aldehydes are generated from both primary sources (motor vehicles and industrial facilities) and secondary sources (photochemical reactions). Formation by way of photochemical reactions is greater in summer than in winter, which might contribute to the concentration differences for formaldehyde observed in our winter and summer sampling campaigns. We did not clearly understand the seasonal trend found for acetaldehyde in this study.

Day of the week was not found to be a significant factor affecting either formaldehyde ($P = 0.9$) or acetaldehyde ($P = 0.57$) concentrations.

In addition, an R was calculated for ambient formaldehyde measured at the Copewood–Davis and Waterfront South fixed monitoring sites. A statistically significant ($P = 0.05$) but moderate R (0.43) was observed, which implies that ambient formaldehyde in both neighborhoods was affected by similar sources, such as diesel emissions and secondary formation from photochemical reactions.

The aldehyde concentrations measured in Waterfront South and Copewood–Davis were compared with those reported in the literature. The mean and median concentrations of formaldehyde and acetaldehyde measured in the two neighborhoods were both two to three times higher than those reported previously; concentrations of acrolein and propionaldehyde were similar to those in other polluted urban areas. Weisel and colleagues (2005), for example, reported that median outdoor concentrations of formaldehyde, acetaldehyde, acrolein, and propionaldehyde were 7.09 , 7.86 , 0.39 , and $1.06 \mu\text{g}/\text{m}^3$, respectively, in Elizabeth; 6.16 , 4.70 , 0.95 , and $1.34 \mu\text{g}/\text{m}^3$, respectively, in Houston; and 6.52 , 5.27 , 0.40 , and $1.82 \mu\text{g}/\text{m}^3$, respectively, in Los Angeles. Sax and colleagues (2004) reported mean ambient concentrations for New York City in the TEACH study, including 2.5 , 2.7 , and $0.9 \mu\text{g}/\text{m}^3$ for formaldehyde, acetaldehyde, and propionaldehyde, respectively, in winter and 4.6 , 4.1 , and $0.7 \mu\text{g}/\text{m}^3$, respectively, in summer; ambient concentrations for Los Angeles were similar to those for New York City. Mean concentrations of formaldehyde, acetaldehyde, and propionaldehyde measured in the present study were also two to three times higher than those measured in other New Jersey locations (NJDEP 2004). These results indicate that both Waterfront South and Copewood–Davis are hot spots for aldehydes.

In comparing the results for Waterfront South and Copewood–Davis (Table 12a) with those reported for other cities, one would conclude that Waterfront South and Copewood–Davis have significant sources of aldehydes. (The measurement method used in our study and in RIOPA were similar, which allowed direct comparison between the two studies.) The potential sources of aldehydes in the study areas might include the four industrial facilities identified by the NJDEP (Table 1), including a sewage treatment plant and a recycling facility, as well as emissions from the diesel-powered trucks driving through and idling in the two neighborhoods. Some aldehydes might also have been contributed by the Philadelphia urban plume (NJDEP 2005).

Another factor that might explain part of the differences observed between our study and others (Jurvelin et al. 2001; Sax et al. 2004) could be the sampling technique. In previous studies, either active or passive 2,4-dinitrophenylhydrazine (DNPH) cartridges were used to measure

ambient aldehydes. In our study, PAKS cartridges — which are based on the DNSH derivation of aldehydes — were used for field sampling (Zhang et al. 2000; Herrington et al. 2005). Given its higher sensitivity, greater stability, and reduced interferences by ozone, the DNSH method has been reported to be superior to the traditional DNPH method (Zhang et al. 2000; Herrington et al. 2005). In addition, concentrations of acetaldehyde and possibly other aldehydes (except for formaldehyde) were found to be underestimated by ~60% when using the DNPH method for sampling times greater than 8 hr (Herrington et al. 2007). Details of the evaluation of the two sampling methods are beyond the scope of the current study but can be found in a previous publication (Herrington et al. 2007).

Personal Aldehydes A summary of the descriptive statistics for our measures of personal exposure to aldehydes in Waterfront South and Copewood–Davis is shown in Table 15a. The distribution of personal aldehyde concentrations by day of the week and by season is shown in Tables 16 and 17 as well as in Figures 27 and 28 (formaldehyde and acetaldehyde only). Like the ambient concentrations, personal exposures to aldehydes are dominated by formaldehyde and acetaldehyde. In Waterfront South mean \pm SD and median personal concentrations were $16.8 \pm 15.5 \mu\text{g}/\text{m}^3$ and $14.1 \mu\text{g}/\text{m}^3$, respectively, for formaldehyde and $15.5 \pm 15.5 \mu\text{g}/\text{m}^3$ and $11.4 \mu\text{g}/\text{m}^3$, respectively, for acetaldehyde. Results were similar for Copewood–Davis, where mean \pm SD and median personal concentrations were $16.0 \pm 16.7 \mu\text{g}/\text{m}^3$ and $11.5 \mu\text{g}/\text{m}^3$, respectively, for formaldehyde and $16.6 \pm 15.8 \mu\text{g}/\text{m}^3$ and $15.4 \mu\text{g}/\text{m}^3$, respectively, for acetaldehyde. Personal concentrations for acrolein and propionaldehyde in both neighborhoods were one to two orders of magnitude lower than those for formaldehyde and acetaldehyde (Table 15a).

Personal exposures to aldehydes include ambient and nonambient contributions. Spatial and temporal factors can influence personal total exposures by altering ambient air concentration, nonambient air concentrations, and personal activity pattern (U.S. EPA 2004). It is therefore important to investigate how personal total exposures covary with location, day of the week, and season. Previous studies have shown that personal exposures to many pollutants, including aldehydes, can be confounded by either active or passive smoking (Bi et al. 2005). For this reason the mixed-effects model described above was applied both to the entire study population and to subjects without ETS exposures (nicotine concentrations $< 0.5 \mu\text{g}/\text{m}^3$).

For the entire study population, location significantly ($P = 0.02$) affected personal exposure to acetaldehyde ($15.5 \mu\text{g}/\text{m}^3$ in Waterfront South versus $16.6 \mu\text{g}/\text{m}^3$ in

Copewood–Davis, Table 15a), and season significantly affected personal exposure both to formaldehyde ($P = 0.003$; $19.8 \mu\text{g}/\text{m}^3$ in summer versus $10.9 \mu\text{g}/\text{m}^3$ in winter for combined data from the two locations) and to acetaldehyde ($P = 0.03$; $12.5 \mu\text{g}/\text{m}^3$ in summer versus $19.0 \mu\text{g}/\text{m}^3$ in winter for combined data). Similar results were observed when comparing the data by season within each location (Table 16). Day of the week was not found to significantly affect either formaldehyde or acetaldehyde concentrations ($P > 0.9$ for formaldehyde and > 0.1 for acetaldehyde). For subjects without ETS exposure, location marginally ($P = 0.10$) affected personal exposure to acetaldehyde ($14.0 \mu\text{g}/\text{m}^3$ in Waterfront South versus $16.8 \mu\text{g}/\text{m}^3$ in Copewood–Davis), and season significantly affected exposure both to formaldehyde ($P = 0.02$; $19.6 \mu\text{g}/\text{m}^3$ in summer versus $10.9 \mu\text{g}/\text{m}^3$ in winter for combined data from the two locations) and to acetaldehyde ($P = 0.01$; $12.0 \mu\text{g}/\text{m}^3$ in summer versus $18.8 \mu\text{g}/\text{m}^3$ in winter for combined data). Again, day of the week was not found to significantly affect either formaldehyde or acetaldehyde concentrations.

After excluding subjects with ETS exposure, location changed from a highly significant factor to a marginal factor. This reinforces the fact that personal exposures to aldehydes in the two neighborhoods were also affected by nonambient exposures. As indicated by Bi and colleagues (2005), acetaldehyde is a dominant species in cigarette emissions (ten times higher than formaldehyde); a cigarette emits about 2000 μg acetaldehyde. Personal exposure to acetaldehyde was higher in Copewood–Davis than in Waterfront South both for the study population as a whole and for the subjects without ETS exposure. The seasonal variation in personal exposures was also consistent with the seasonal trend in ambient concentrations: formaldehyde exposure was higher in summer, and acetaldehyde exposure was higher in winter. The R for formaldehyde and acetaldehyde exposure was also similar to that of the ambient concentrations — higher in winter (0.54) and lower in summer (0.47). Seasonal effects on personal exposure to aldehydes have not been discussed in most exposure studies. Kinney and colleagues (2002) reported that personal exposures to formaldehyde and acetaldehyde in summer were higher than those in winter; Maitre and colleagues (2002) reported the opposite. In general, both ambient concentrations and personal exposures to aldehydes can be affected by spatial and temporal factors, such as location, season, and day of the week. Regardless of the seasonal variations, the ambient concentrations of aldehydes were higher than the personal concentrations, meaning that ambient sources were the principal determinants of personal total exposures in both Waterfront South and Copewood–Davis.

Personal exposures to aldehydes were comparable to the values reported elsewhere (Jurvelin et al. 2001; Kinney et al. 2002; Serrano-Trespalcacios et al. 2004; Weisel et al. 2005). Kinney and colleagues (2002) reported personal exposure concentrations for formaldehyde and acetaldehyde for TEACH subjects in New York City of 11.5 and 13.0 $\mu\text{g}/\text{m}^3$, respectively, in winter and 28.5 and 20.2 $\mu\text{g}/\text{m}^3$, respectively, in summer. Serrano-Trespalcacios and colleagues (2004) reported median personal exposures of 17.2, 11.1, and 2.2 $\mu\text{g}/\text{m}^3$ for formaldehyde, acetaldehyde, and propionaldehyde, respectively, in Mexico City. Jurvelin and colleagues (2001) reported mean formaldehyde and acetaldehyde personal exposures of 26.6 and 14.4 $\mu\text{g}/\text{m}^3$, respectively, for the EXPOLIS Helsinki population. In the RIOPA study, Weisel and colleagues (2005) reported personal exposures for formaldehyde, acetaldehyde, acrolein, and propionaldehyde of 21.9, 17.3, 0.74, and 1.69 $\mu\text{g}/\text{m}^3$, respectively, in Elizabeth; 21.6, 29.4, 0.91, and 2.19 $\mu\text{g}/\text{m}^3$ in Houston; and 21.7, 22.7, 1.12, and 2.15 $\mu\text{g}/\text{m}^3$, respectively, in Los Angeles.

Although our measures of personal exposures to aldehydes were similar to those reported in other studies, the principal sources contributing to our measures were different. Total personal exposure can be decomposed into various components resulting from exposures in various microenvironments. Several analyses were conducted to determine whether ambient aldehydes were significant contributors to personal exposures in Waterfront South and Copewood–Davis; the results are reported in the following sections.

Personal and Ambient Aldehyde Relationships Scatter plots for personal versus ambient concentrations of formaldehyde and acetaldehyde are shown in Figure 29 and Figure 30. As Figure 29 shows, the ambient formaldehyde concentrations were significantly higher than the personal concentrations (23.0 $\mu\text{g}/\text{m}^3$ for ambient, 15.6 $\mu\text{g}/\text{m}^3$ for personal, $P < 0.001$). The same relationship still held after exclusion of ETS-exposed subjects (23.0 $\mu\text{g}/\text{m}^3$ for ambient, 15.3 $\mu\text{g}/\text{m}^3$ for personal, $P < 0.001$). Personal acetaldehyde concentrations were moderately but statistically significantly higher than the ambient concentrations (13.9 $\mu\text{g}/\text{m}^3$ for ambient, 15.0 $\mu\text{g}/\text{m}^3$ for personal, $P < 0.001$). The same relationship still held after exclusion of ETS-exposed subjects (13.9 $\mu\text{g}/\text{m}^3$ for ambient, 14.6 $\mu\text{g}/\text{m}^3$ for personal, $P < 0.001$). These observations are different from those of most other studies (Jurvelin et al. 2001; Kinney et al. 2002; Weisel et al. 2005). In the TEACH study, Kinney and colleagues (2002) reported that personal formaldehyde exposures were five to six times higher than corresponding outdoor concentrations. In the EXPOLIS Helsinki study, Jurvelin and colleagues (2001) reported

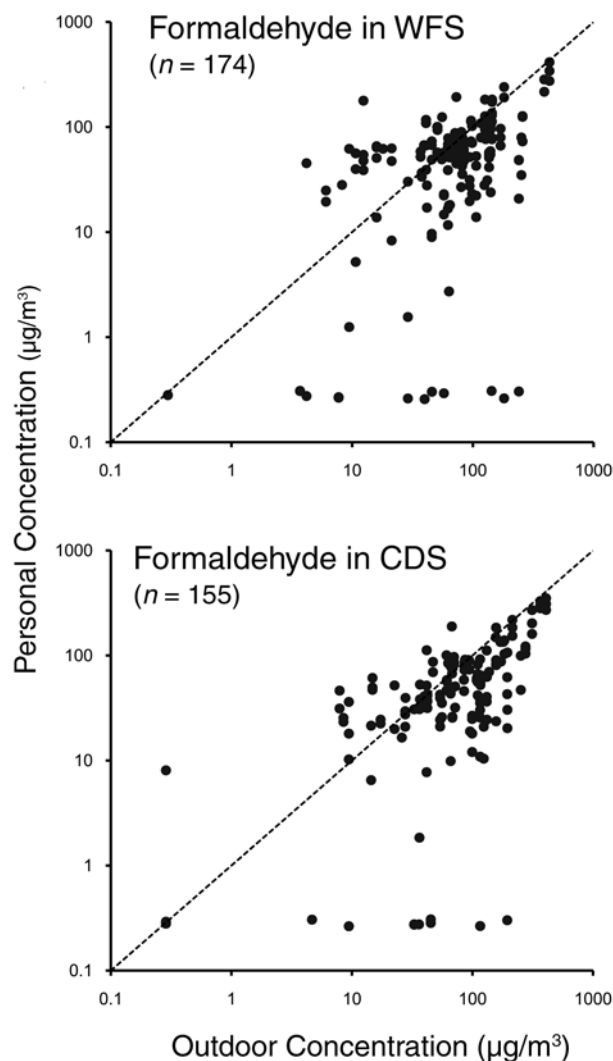


Figure 29. Scatter plots of personal versus ambient formaldehyde concentrations in (top) Waterfront South (WFS) and (bottom) Copewood–Davis (CDS). Shown on logarithmic scales; n is the number of samples in each category.

that personal formaldehyde exposures were 10 times higher than residential outdoor concentrations. In the RIOPA study, Weisel and colleagues (2005) reported that personal formaldehyde was about three times higher than residential outdoor concentrations. These studies also reported that personal acetaldehyde exposures were two to five times higher than ambient concentrations (Jurvelin et al. 2001; Kinney et al. 2002; Weisel et al. 2005).

The apparent contradiction between our study results and those of other studies is attributable to (but not limited to) the outdoor sources of aldehydes in Waterfront South and Copewood–Davis. Elevated personal exposures in the earlier studies were attributed to high indoor emissions of formaldehyde. The main indoor sources of formaldehyde

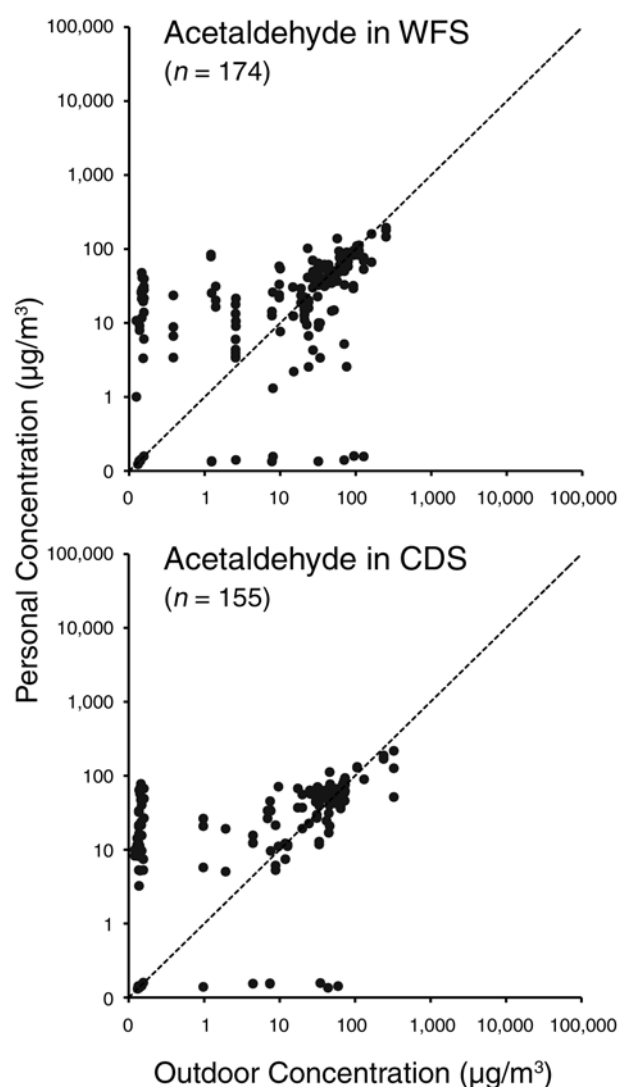


Figure 30. Scatter plots of personal versus ambient acetaldehyde concentrations in Waterfront South (WFS) and (bottom) Copewood-Davis (CDS). Shown on logarithmic scales; n is the number of samples in each category.

are off-gassing from various building materials and formaldehyde formation from indoor air chemical reactions (Hodgson et al. 2002; Morrison and Nazaroff 2002; Weschler 2004; Wang and Morrison 2006). Both mechanistic experiments and observational exposure studies have reported that old houses emit fewer aldehydes than new houses (Clarisse et al. 2003; Gilbert et al. 2005; Wang and Morrison 2006). Wang and Morrison (2006) reported that the secondary emission rate of aldehydes from living room carpets was $80 \mu\text{g}/\text{m}^2$ per hr in a 1-year-old home compared with only 8 to $20 \mu\text{g}/\text{m}^2$ per hour in two homes that were more than 10 years old. During a study of indoor air quality in Canada, Gilbert and colleagues (2005) observed elevated formaldehyde concentrations only in homes built

after 1970. Most of the subjects' homes in our study were built before 1960, according to our questionnaire survey; this could be one of the reasons why contributions from indoor sources to personal formaldehyde exposures were less significant than contributions from outdoor sources in both Waterfront South and Copewood-Davis.

Quantification of Personal and Ambient Associations

Associations between personal and ambient exposures were quantified using Spearman correlation coefficients and mixed-effects models (Table 18 and Tables 19a, b, and c). Personal exposures to formaldehyde and acetaldehyde were significantly ($P = 0.05$) correlated with ambient concentrations. For the study population as a whole, the R 's were 0.55 for formaldehyde and 0.68 for acetaldehyde (Table 18). No common associations were observed for variations in R caused (1) by ETS exposure (0.51 for non-smokers versus 0.65 for smokers for formaldehyde and 0.73 for nonsmokers versus 0.51 for smokers for acetaldehyde), (2) by location (0.61 for Copewood-Davis versus 0.48 for Waterfront South for formaldehyde and 0.68 for Copewood-Davis versus 0.68 for Waterfront South for acetaldehyde), (3) by season (0.68 for summer versus 0.42 for winter for formaldehyde and 0.71 for summer versus 0.62 for winter for acetaldehyde), or (4) by day of the week (0.56 for weekdays versus 0.54 for weekend days for formaldehyde and 0.70 for weekdays versus 0.66 for weekend days for acetaldehyde). However, in all cases the R 's were statistically significant ($P = 0.05$). Moreover, the associations between personal exposures and ambient concentration were stronger in summer than in winter. This is plausible given that people spend more time outdoors in the summer and that air-exchange rates are higher in summer in houses without central air conditioning.

The slope, associated P value, and R^2 for personal and ambient associations using the mixed-effects models are shown in Tables 19a, b, and c. The slopes were similar to the attenuation factor in Equation 1, shown earlier, which is determined by home ventilation rates, chemical properties, and the time people spend outdoors. Seasonal differences in the slopes were significant, changing from 0.66 in summer to 0.13 in winter for formaldehyde and from 0.68 in summer to 0.42 in winter for acetaldehyde (Table 19b). Weekday slopes were also higher than weekend slopes, which might reflect variations in personal activity patterns on weekdays and weekend days (Table 19c). The slopes for formaldehyde on both weekdays and weekends were lower than those for acetaldehyde, which is consistent with the chemical properties of the two compounds. Formaldehyde is more reactive than acetaldehyde; it has a higher loss rate indoors and therefore a smaller attenuation

factor (Liu et al. 2006). In other words, personal exposures are more sensitive to ambient acetaldehyde than to ambient formaldehyde. Most of the R^2 's for the aldehydes were close to or larger than 0.8, indicating that mixed models could be used to describe the associations (and that ambient sources had contributed significantly to the personal exposures).

PAHs

Summaries of the descriptive statistics for our concentrations measures (i.e., the sum of the concentrations in gas and particle phases) of the 16 PAHs are shown in Tables 12 through 14 for results from ambient samples and Tables 15 through 17 for results from personal samples. Ambient PAHs are primarily generated by various types of combustion sources and consist predominantly of compounds with two to five fused benzene rings. According to data in the literature (Naumova et al. 2002; Li et al. 2005), groups

of PAHs with the same number of fused benzene rings are often present in similar concentrations in ambient air. We therefore attempted to select one species from each such group to examine their ambient and personal exposure concentrations. Four compounds — naphthalene (a two-benzene ring), phenanthrene (a three-benzene ring), pyrene (a four-benzene ring), and benzo[*a*]pyrene (a five-benzene ring) — were selected for analysis, either because they tend to be present in higher (i.e., more easily detectable) concentrations among the 16 PAHs (e.g., naphthalene and phenanthrene) or because they are known to be of greater concern for potential major health effects (e.g., benzo[*a*]pyrene).

Ambient PAHs Given the known sampling artifacts of the filter-PUF sampling system (Coutant et al. 1988; McDow et al. 1990; Sanderson et al. 2005), the PAH

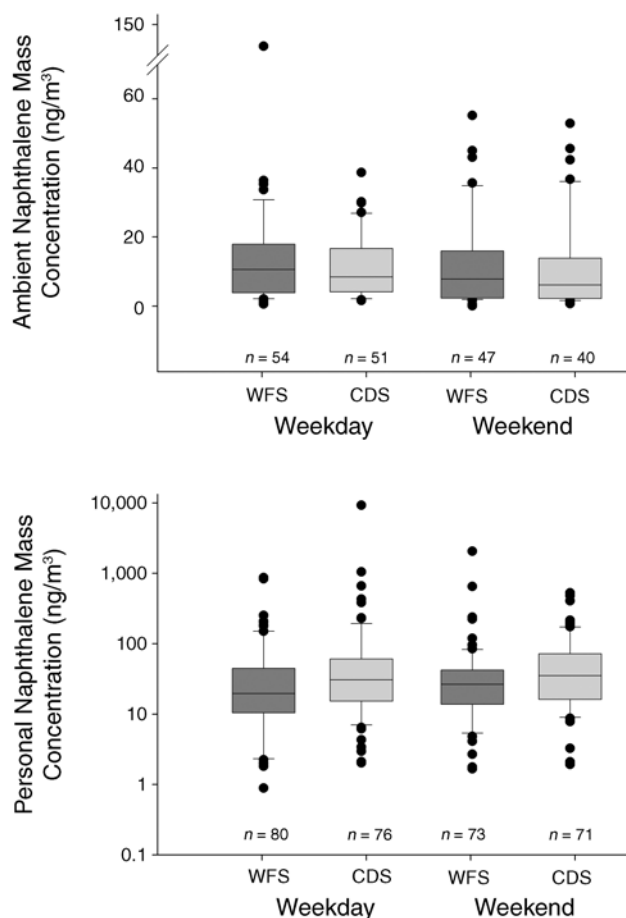


Figure 31. Ambient (top) and personal (bottom) naphthalene concentrations in Waterfront South (WFS) and Copewood-Davis (CDS) by day of the week. Scales are linear and logarithmic, respectively; *n* is the number of samples in each category.

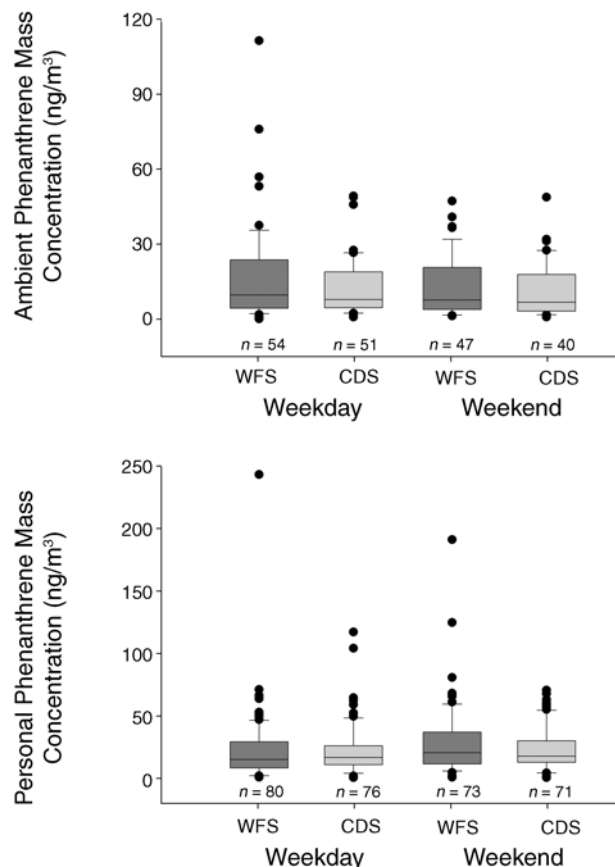


Figure 32. Ambient (top) and personal (bottom) phenanthrene concentrations in Waterfront South (WFS) and Copewood-Davis (CDS) by day of the week. Note that the y-axis scales differ; *n* is the number of samples in each category.

concentrations determined in our filter and PUF samples were summed for analysis. Mean \pm SD and median concentrations in Waterfront South were 13.1 ± 17.2 ng/m³ and 9.78 ng/m³ for naphthalene, 15.5 ± 17.0 ng/m³ and 9.14 ng/m³ for phenanthrene, 1.47 ± 1.52 ng/m³ and 0.99 ng/m³ for pyrene, and 0.36 ± 0.45 ng/m³ and 0.18 ng/m³ for benzo[a]pyrene (Table 12b). Both mean and median concentrations for all species observed in Waterfront South were significantly higher ($P < 0.0161$; see Tables 12b, 13, 14) than those in Copewood–Davis. Mean and median concentrations in Copewood–Davis were 11.7 ± 11.2 ng/m³ and 7.62 ng/m³ for naphthalene, 12.4 ± 11.3 ng/m³ and 7.47 ng/m³ for phenanthrene, 1.23 ± 1.17 ng/m³ and 0.91 ng/m³ for pyrene, 0.21 ± 0.26 ng/m³, and 0.12 ng/m³ for benzo[a]pyrene (Tables 12b, 13, and 14).

Comparisons were made to examine the effects of day of the week (Table 13 and Figures 31 through 34) and season

(Table 14 and Figures 35 through 38) on the differences in ambient PAHs observed in the two neighborhoods. A greater difference was observed for some of the PAH species on weekdays (Table 13, Figures 31 through 34). In Waterfront South mean concentrations on weekdays were 17.5 ± 20.2 ng/m³ for phenanthrene (Figure 32) and 1.94 ± 1.91 ng/m³ for pyrene (Table 13). These were 36% ($P = 0.0092$) and 50% ($P = 0.0009$) higher, respectively, than the corresponding means in Copewood–Davis (12.9 ± 11.7 ng/m³ and 1.29 ± 1.29 ng/m³; Table 13). In winter, benzo[a]pyrene concentrations in Waterfront South (0.57 ± 0.51 ng/m³) were found to be significantly (78%) higher than those in Copewood–Davis (0.32 ± 0.28 ng/m³) on both weekdays and weekend days ($P < 0.0001$ in winter, $P < 0.0001$ on weekdays, and $P = 0.0197$ on weekend days; Table 14 and Figure 38). Mean benzo[a]pyrene concentrations in Waterfront South were 0.44 ± 0.54 ng/m³ on weekdays and

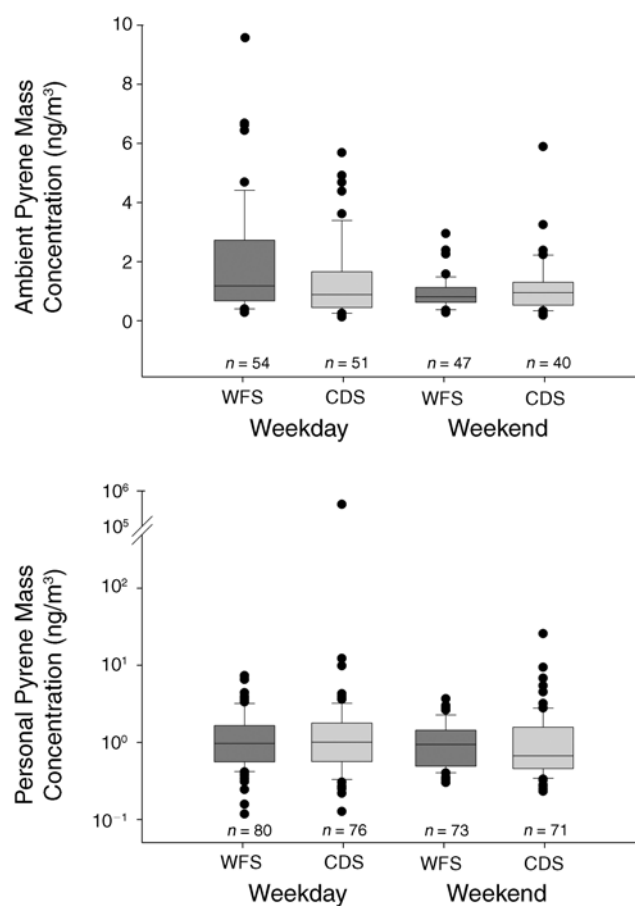


Figure 33. Ambient (top) and personal (bottom) pyrene concentrations in Waterfront South (WFS) and Copewood–Davis (CDS) by day of the week. Scales are linear and logarithmic, respectively; n is the number of samples in each category.

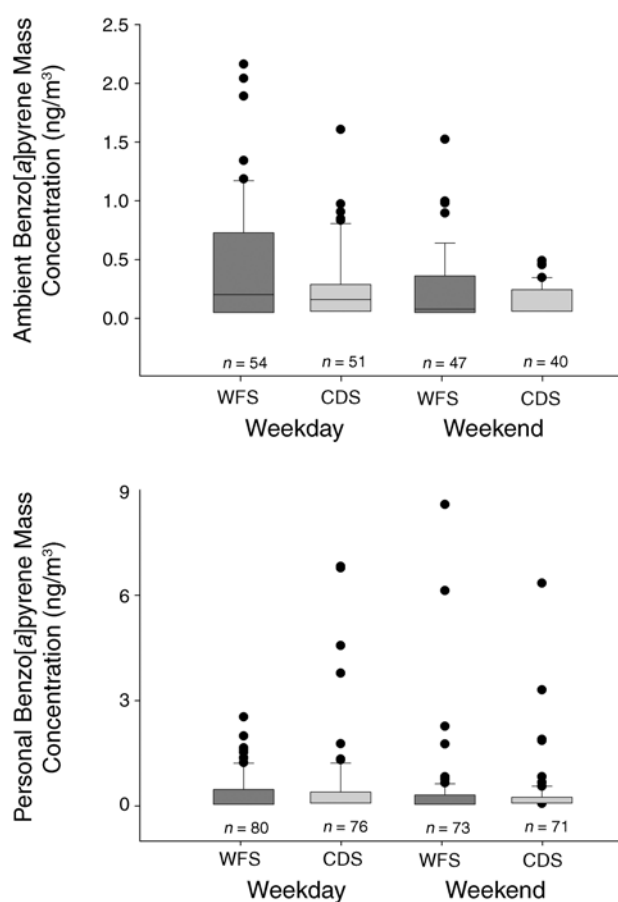


Figure 34. Ambient (top) and personal (bottom) benzo[a]pyrene concentrations in Waterfront South (WFS) and Copewood–Davis (CDS) by day of the week. Note that the y-axis scales differ; n is the number of samples in each category.

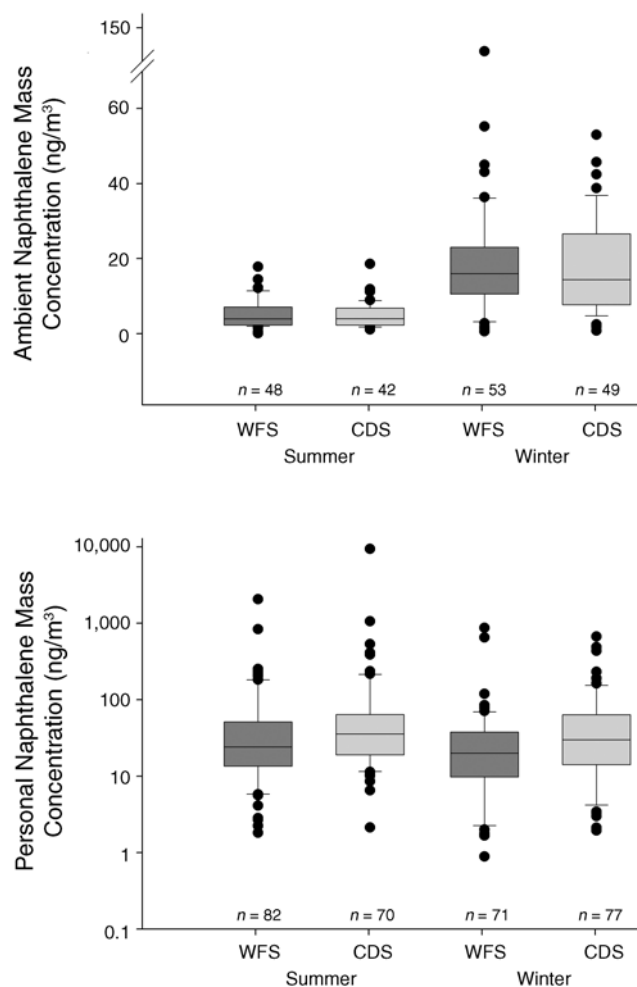


Figure 35. Ambient (top) and personal (bottom) naphthalene concentrations in Waterfront South (WFS) and Copewood–Davis (CDS) by season. Scales are linear and logarithmic, respectively; *n* is the number of samples in each category.

$0.26 \pm 0.31 \text{ ng/m}^3$ on weekend days, 69% ($P < 0.0001$) and 86% ($P = 0.0197$) higher, respectively than the corresponding means in Copewood–Davis ($0.26 \pm 0.31 \text{ ng/m}^3$ and $0.14 \pm 0.13 \text{ ng/m}^3$; Table 13 and Figure 34). The fact that all PAH concentrations in Waterfront South were significantly higher (36% to 86%) than those in Copewood–Davis suggests the presence of local PAH sources in Waterfront South. It is well known that PAHs are produced by incomplete combustion, high-temperature pyrolytic processes involving fossil fuels, and, more generally, carbonaceous materials (Baek et al. 1991; ATSDR 1995). Major outdoor sources of PAHs reported for urban areas, such as diesel exhaust and municipal and industrial waste incineration (Peltonen and Dipple 1995), can be found in and near Waterfront South (NJDEP 2005); this could result in elevated PAH concentrations in the neighborhood. In addition, the greater difference in weekday–weekend measurements

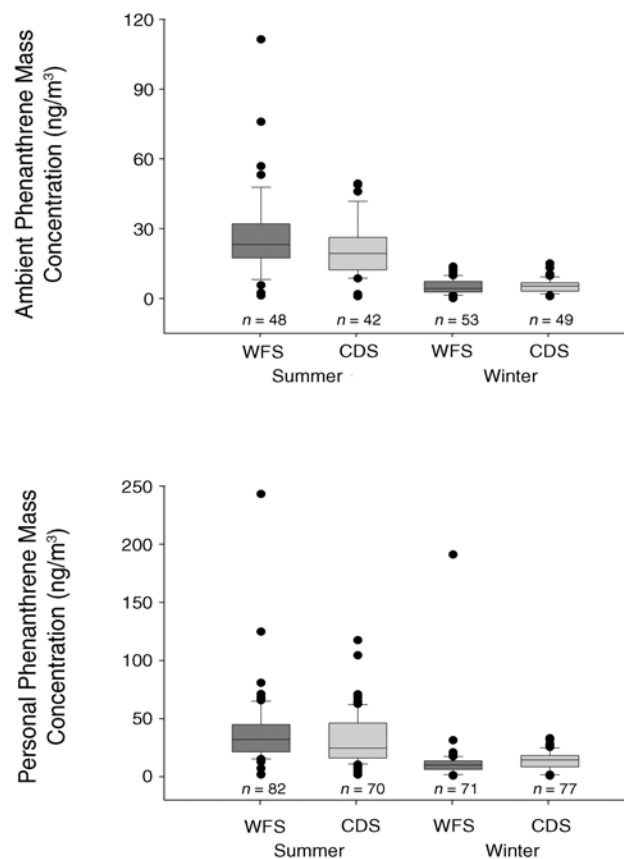


Figure 36. Ambient (top) and personal (bottom) phenanthrene concentrations in Waterfront South (WFS) and Copewood–Davis (CDS) by season. Note that the y-axis scales differ; *n* is the number of samples in each category.

supports the hypothesis that emissions from higher volumes of traffic and the operation of industrial facilities on weekdays have a significant impact on PAH pollution in Waterfront South.

Ambient PAHs measured in this study were in the same range as those reported for other urban areas (Chuang et al. 1991, 1999; Naumova et al. 2002; Li et al. 2005). Chuang and colleagues (1999) measured PAH exposures in low-income families with children in North Carolina. The mean concentrations were 433 ng/m^3 for naphthalene, 30.4 ng/m^3 for phenanthrene, 2.95 ng/m^3 for pyrene, and 0.46 ng/m^3 for benzo[*a*]pyrene. Naumova and colleagues (2002) measured indoor and outdoor concentrations of 30 PAHs in 55 nonsmoking residences in three U.S. cities. The concentrations ranged from 8.1 to 26 ng/m^3 for phenanthrene, 1.6 to 3.6 ng/m^3 for pyrene, and 0.025 to 0.14 ng/m^3 for benzo[*a*]pyrene. Our mean concentrations for phenanthrene

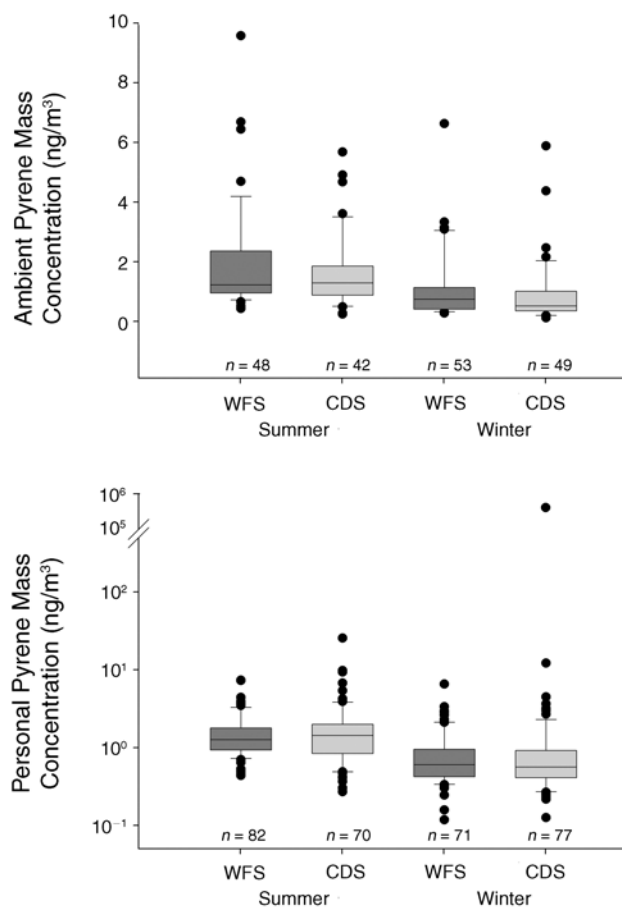


Figure 37. Ambient (top) and personal (bottom) pyrene concentrations in Waterfront South (WFS) and Copewood–Davis (CDS) by season. Scales are linear and logarithmic, respectively; *n* is the number of samples in each category.

and pyrene in Waterfront South and Copewood–Davis were similar to those of other studies. Our naphthalene was lower, probably as a result of sampling breakthrough. Previous studies (Mitra and Ray 1995; Fraser et al. 1998; Totten et al. 2005; Park et al. 2001; Naumova et al. 2002) provided some insight into nationwide ambient concentrations of benzo[*a*]pyrene; these ranged from nondetectable to 1.0 ng/m³, with a mean concentration of nondetectable to 0.14 ng/m³. Compared with these national concentrations, the ambient concentrations of benzo[*a*]pyrene in both Waterfront South and Copewood–Davis exceeded the upper 1.0 ng/m³ end of the range in 6% of samples; Waterfront South had the higher values.

Our PAH concentrations were analyzed for differences by day of the week in each location (Table 13 and Figures 31 through 34). In general, mean concentrations for all species were found to be higher on weekdays than on weekend

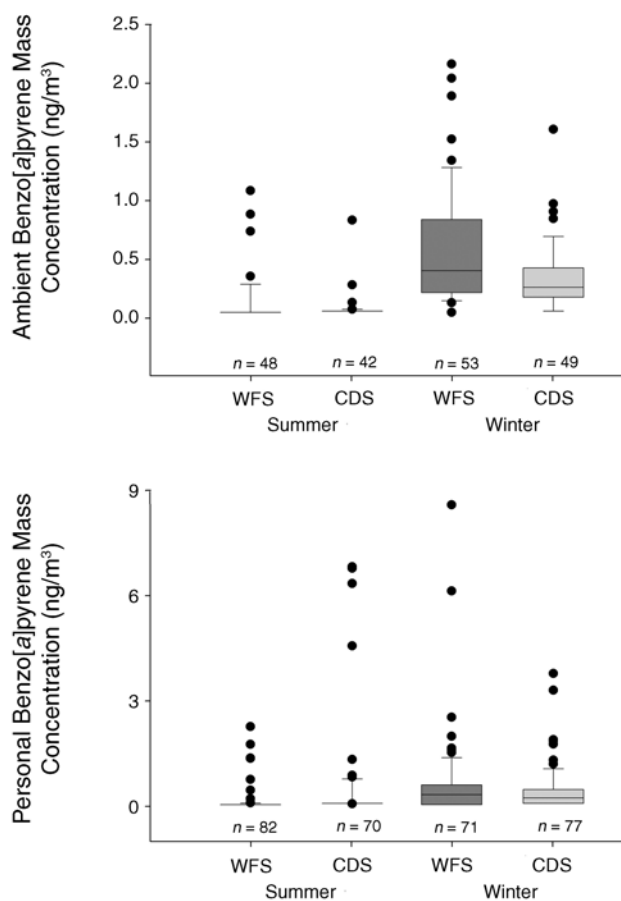


Figure 38. Ambient (top) and personal (bottom) benzo[*a*]pyrene concentrations in Waterfront South (WFS) and Copewood–Davis (CDS) by season. Note that the y-axis scales differ; *n* is the number of samples in each category.

days. In Waterfront South the differences were significant for pyrene ($P < 0.01$) and benzo[*a*]pyrene ($P < 0.01$), marginally significant for phenanthrene ($P = 0.11$), and nonsignificant for naphthalene ($P = 0.38$). In Copewood–Davis the differences for the same four species were not significant ($P = 0.11$ to 0.97). These results again demonstrate the impact of higher traffic volumes and the operation of industrial facilities on the neighborhood's PAH pollution on weekdays. In Copewood–Davis traffic is the principal source of air pollution, with the result that the day of the week is less significant.

A significant seasonal difference was observed for the four PAHs analyzed (Table 14 and Figures 35 through 38). In Waterfront South, mean ambient concentrations in winter were 20.3 ± 21.1 ng/m³ for naphthalene and 0.57 ± 0.51 ng/m³ for benzo[*a*]pyrene, about four to five times higher ($P < 0.01$ for naphthalene and $P < 0.01$ for

benzo[*a*]pyrene) than the corresponding means in summer ($5.23 \pm 3.95 \text{ ng/m}^3$ and $0.12 \pm 0.22 \text{ ng/m}^3$). In Copewood–Davis, mean ambient concentrations in winter were $17.6 \pm 12.2 \text{ ng/m}^3$ for naphthalene and $0.32 \pm 0.28 \text{ ng/m}^3$ for benzo[*a*]pyrene, about four to five times higher ($P < 0.01$ for naphthalene and $P < 0.01$ for benzo[*a*]pyrene) than the corresponding means in summer ($4.81 \pm 3.49 \text{ ng/m}^3$ and $0.08 \pm 0.12 \text{ ng/m}^3$).

These seasonal variations in ambient PAH concentrations reflect the presence of a number of PAH sources, the complexity of the source profiles for individual sources, and the reactivity of the various PAH compounds. In general, it has been found that concentrations of PAHs with lower molecular weights (< 4 fused benzene rings) are usually higher in summer and that concentrations of PAHs with higher molecular weights (≥ 5 fused benzene rings), such as benzo[*a*]pyrene, are usually higher in winter (Lioy 1990; Lioy and Greenberg 1990; Chuang et al. 1999; Dimashki et al. 2001; Naumova et al. 2002; Li et al. 2005). In urban areas traffic has been found to be the primary all-season source of ambient PAHs (Dubowsky et al. 1999). PAH emissions from home heating are highly seasonal and increase substantially in winter (Lioy and Greenberg 1990; Dimashki et al. 2001). The volatility of individual PAHs also varies greatly and is strongly associated with temperature. The vaporization of PAHs from soil, water, vegetation, etc., can thus lead to significantly different PAH emissions from season to season (Dimashki et al. 2001). The reactivity of each PAH is not the same. Benzo[*a*]pyrene, for example, is highly reactive under photochemical conditions and decomposes quickly in summer. The higher benzo[*a*]pyrene concentrations we measured in winter would therefore be attributable primarily to higher emissions or lower photochemical activity in winter. The higher phenanthrene and pyrene concentrations might have been caused by higher vaporization of PAHs from various surfaces. The higher naphthalene measured in winter might have been caused in part by measurement bias in summer (e.g., higher naphthalene breakthrough in summer than in winter, leading to lower summer naphthalene measures).

Personal PAHs Associations between personal and ambient concentrations of PAHs were analyzed to examine the contribution of ambient PAH sources to personal PAH exposures. First, scatter plots of personal versus ambient concentrations of PAH in both locations (Figures 39 through 42) were visually examined. Generally, the data from both locations followed similar patterns: data for phenanthrene (Figure 40) and pyrene (Figure 41) were close to the 1:1 line, and data for naphthalene (Figure 29)

tended to scatter above the line. Benzo[*a*]pyrene was scattered across the line (Figure 42). The data for phenanthrene and pyrene suggest that personal exposures to these two compounds were influenced primarily by ambient sources present in both locations. The data for naphthalene indicate a significant contribution from indoor sources to personal exposure to naphthalene.

Mean \pm SD and median PAH concentrations are shown in Table 15b. Concentrations by day of the week and by season are shown in Table 16, Table 17, and Figures 35 through 38. One extremely high value ($> 300,000 \text{ ng/m}^3$) observed for pyrene and fluoranthene in Copewood–Davis was excluded from the mean concentration calculation because it dominated the distribution of PAH concentrations and biased the overall analysis of measurements obtained in the study. In Waterfront South mean and median concentrations were $64.4 \pm 197 \text{ ng/m}^3$ and 22.3 ng/m^3 for naphthalene, $24.9 \pm 28.0 \text{ ng/m}^3$ and 17.5 ng/m^3 for phenanthrene, $1.28 \pm 1.16 \text{ ng/m}^3$ and 0.94 ng/m^3 for pyrene, $0.38 \pm 0.93 \text{ ng/m}^3$ and 0.05 ng/m^3 for benzo[*a*]pyrene, respectively. In Copewood–Davis mean and median concentrations were $133 \pm 770 \text{ ng/m}^3$ and 32.9 ng/m^3 for naphthalene, $23.4 \pm 20.5 \text{ ng/m}^3$ and 17.5 ng/m^3 for phenanthrene, $1.56 \pm 2.60 \text{ ng/m}^3$ and 0.88 ng/m^3 for pyrene, $0.42 \pm 1.08 \text{ ng/m}^3$ and 0.05 ng/m^3 for benzo[*a*]pyrene, respectively.

As shown in Table 15b, a wide range of personal PAH concentrations were observed during the study. Several extremely high concentrations of certain compounds were observed for certain subjects. High naphthalene concentrations, for example, were found in three personal samples, one from Waterfront South (2100 ng/m^3) and two from Copewood–Davis (9400 ng/m^3 and 1100 ng/m^3). High nicotine concentrations ($84.7 \text{ } \mu\text{g/m}^3$ and $8.69 \text{ } \mu\text{g/m}^3$) were also found in two of the same samples; a low concentration ($0.2 \text{ } \mu\text{g/m}^3$) was found in the remaining sample. In addition, concentrations of pyrene ($407,000 \text{ ng/m}^3$) and fluoranthene ($357,000 \text{ ng/m}^3$) were found to be very high in one personal sample, but the nicotine concentration of this sample was close to background level ($0.1 \text{ } \mu\text{g/m}^3$). These wide PAH concentration ranges indicate that, in addition to outdoor emission sources, other sources were contributing to personal PAH exposures.

The median personal naphthalene concentration in Waterfront South was 32% lower ($P = 0.0134$) than that in Copewood–Davis (Table 15b). The median personal benzo[*a*]pyrene concentration in Waterfront South in winter was 63% higher ($P = 0.0787$) than that in Copewood–Davis (Table 17). Median personal concentrations of phenanthrene (17.5 ng/m^3) and pyrene (0.94 ng/m^3) in Waterfront South were similar to those in Copewood–Davis

(17.5 ng/m³ and 0.88 ng/m³, respectively) (Table 15b). Differences by location for the personal PAHs (except benzo[a]pyrene) were not the same as those found for the ambient PAHs, indicating that, for most PAHs in Copewood–Davis, sources other than ambient sources were contributing to personal exposures. As noted earlier, higher ETS exposures were found for subjects who lived in

Copewood–Davis, which might have contributed in part to these personal PAH exposures. In addition, PAHs can also have other major indoor sources, such as gas cooking and heating appliances (Waldman et al. 1990; Baek et al. 1991; Chuang et al. 1991; Wagenknecht et al. 1993; ATSDR 1995), that can contribute significantly to personal PAH exposure.

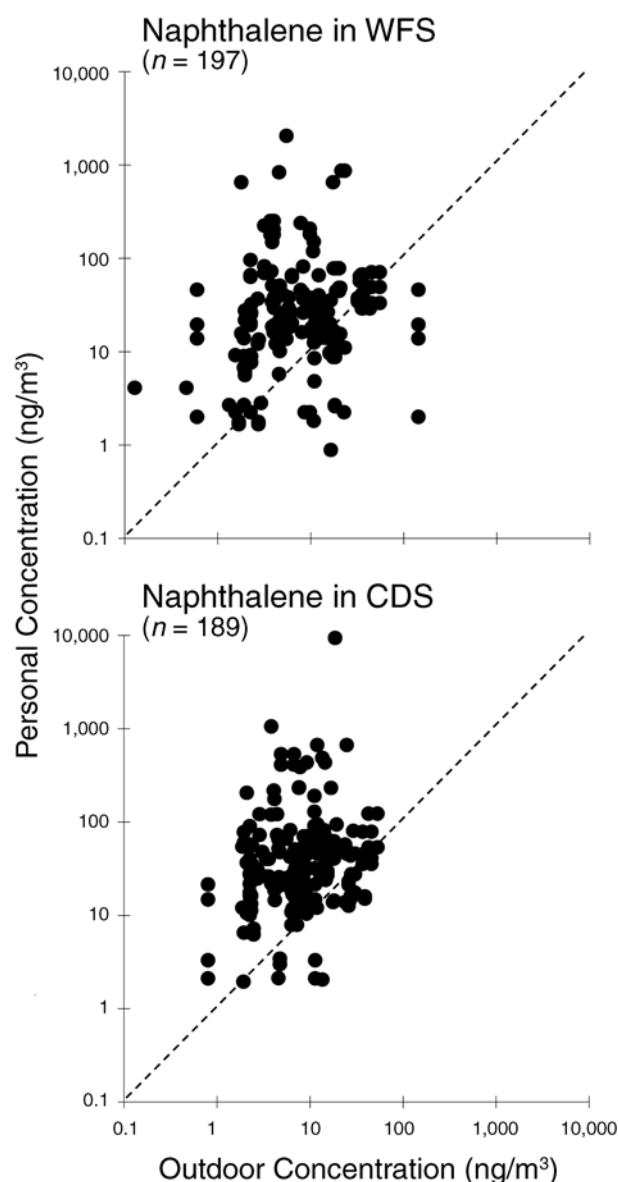


Figure 39. Scatter plots of personal versus ambient naphthalene concentrations in (top) Waterfront South (WFS) and (bottom) Copewood–Davis (CDS). Shown on logarithmic scales; n is the number of samples in each category.

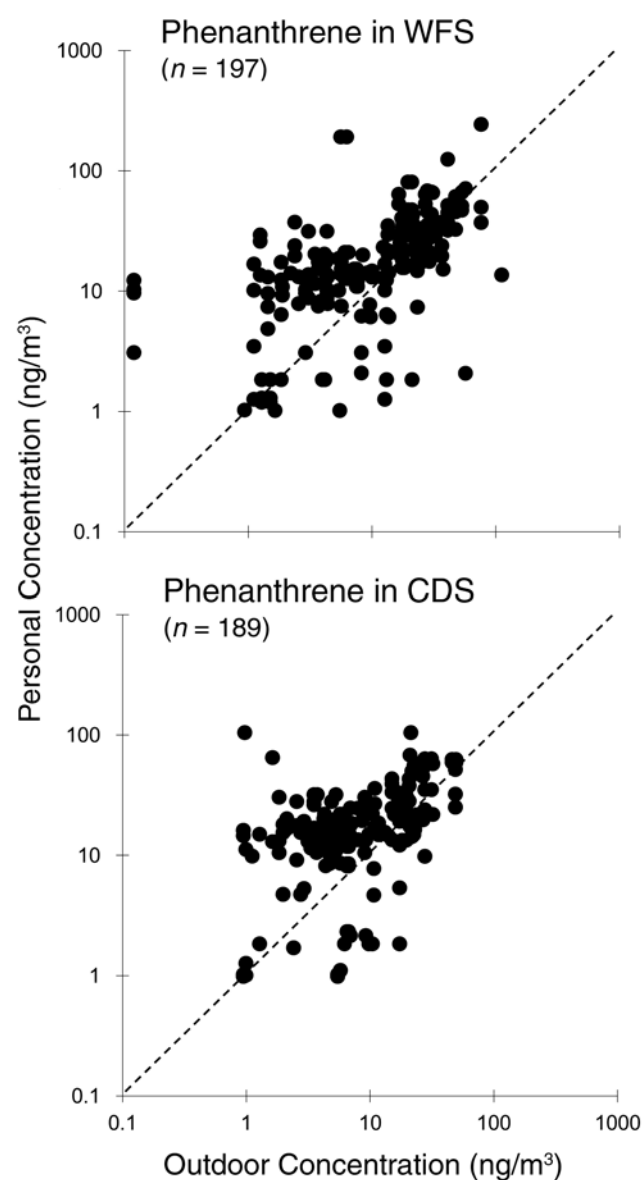


Figure 40. Scatter plots of personal versus ambient phenanthrene concentrations in (top) Waterfront South (WFS) and (bottom) Copewood–Davis (CDS). Shown on logarithmic scales; n is the number of samples in each category.

Personal PAH concentrations observed in the study were comparable to those reported for urban areas in other studies. Tonne and colleagues (2004), for example, measured personal exposures to nine PAHs with molecular weight > 228 among 348 nonsmoking African-American or Dominican pregnant women in New York City and reported mean personal concentrations of 4 ng/m³ for pyrene and 0.5 ng/m³ for benzo[a]pyrene. In personal exposure studies of other non-occupational cohorts, reported values for benzo[a]pyrene ranged from 0.07 to 4.3 ng/m³ (Liroy et al.

1988; Waldman et al. 1990; Sisovic et al. 1996; Zmirou et al. 2000). Waldman and colleagues (1990), for example, reported benzo[a]pyrene concentrations of 0.3 ng/m³ in Phillipsburg, New Jersey.

In Waterfront South the differences in personal PAH exposures between weekdays and weekend days were < 25% and were not statistically significant ($P = 0.4313$ for naphthalene, $P = 0.0878$ for phenanthrene, $P = 0.5287$ for pyrene, and $P = 0.6553$ for benzo[a]pyrene). Similar patterns were observed in Copewood-Davis, where the differences between

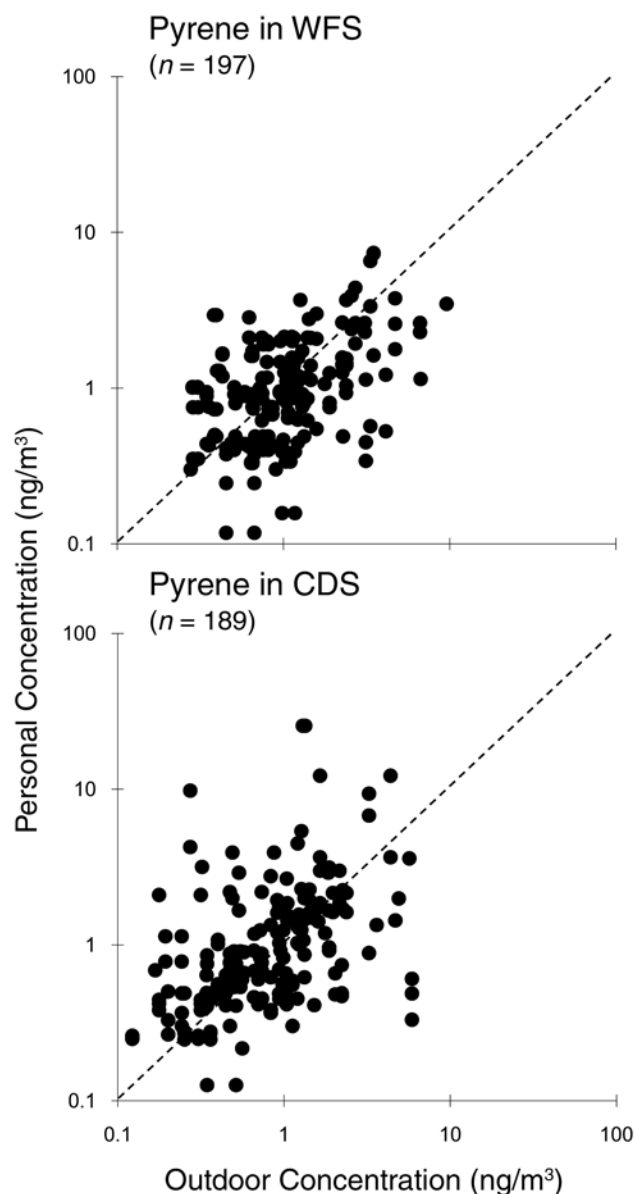


Figure 41. Scatter plots of personal versus ambient pyrene concentrations in (top) Waterfront South (WFS) and (bottom) Copewood-Davis (CDS). Shown on logarithmic scales; n is the number of samples in each category.

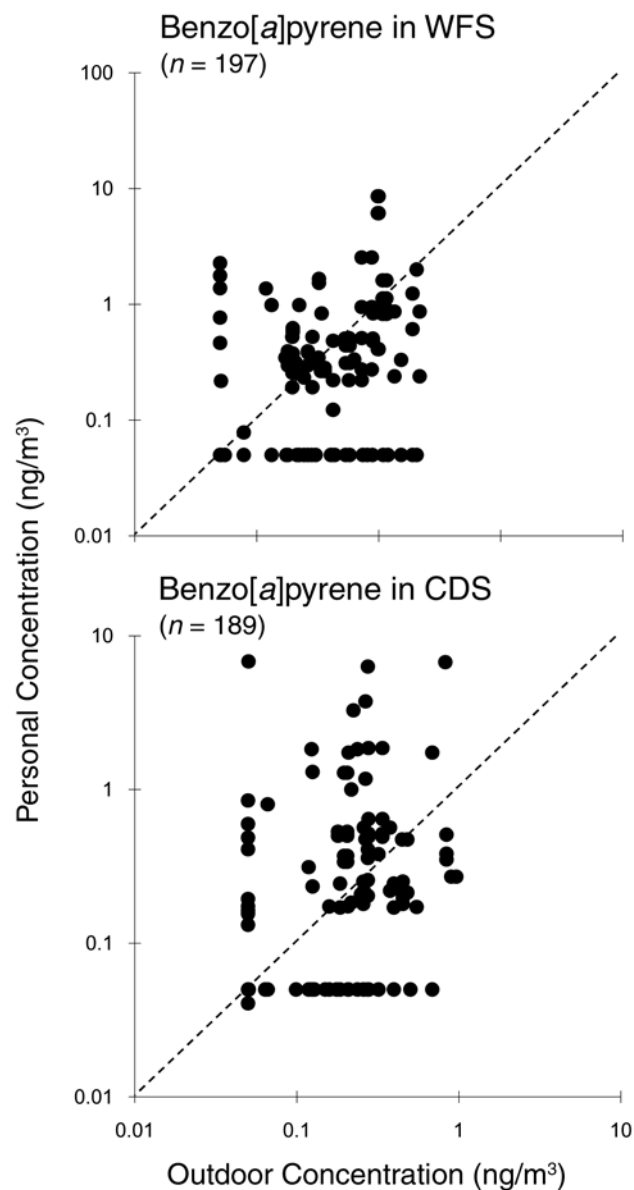


Figure 42. Scatter plots of personal versus ambient benzo[a]pyrene concentrations in (top) Waterfront South WFS and (bottom) Copewood-Davis (CDS). Note that the y-axis scales are logarithmic and differ; n is the number of samples in each category.

weekdays and weekend days were < 15% and not significant ($P = 0.8116$ for naphthalene, $P = 0.5929$ for phenanthrene, and $P = 0.1886$ for benzo[a]pyrene). A difference of 46% was observed for personal pyrene exposures, but it was not significant ($P = 0.1485$).

Except for naphthalene, the seasonal variations in personal exposures to all PAHs measured were found to be significant. In Waterfront South personal exposures to phenanthrene and pyrene were 210% and 110% higher ($P < 0.0001$), respectively, in summer than in winter. In Copewood–Davis they were 70% and 160% higher, respectively, in summer. Personal exposures to naphthalene were 20% higher in summer in both Waterfront South and Copewood–Davis, but the difference was only marginally significant ($P = 0.0686$). Personal exposures to benzo[a]pyrene were 80% higher ($P < 0.0001$) in winter than in summer in both neighborhoods. These observations are consistent with the trends noted earlier for personal $PM_{2.5}$ and suggest, again, that seasonal variations might be caused in part by seasonal changes in personal activities.

Personal and Ambient PAH Relationships The associations between personal and ambient PAH exposures were further examined by Spearman correlation analysis (Table 18). Relatively poor correlations were found for all the target compounds, with R 's ranging from 0.10 to 0.37 in Waterfront South and from 0.23 to 0.45 in Copewood–Davis. These results suggest that sources other than ambient pollution were contributing significantly to personal PAH exposures in both neighborhoods.

The associations between personal and ambient PAH exposures were also examined using the mixed-effects model; the resulting P values and R^2 's are shown in Tables 19a, b, and c. All the R^2 's were less than 0.15, which was consistent with the Spearman analyses, suggesting that sources of PAHs other than ambient sources had dominated the personal exposures in both neighborhoods. Ambient pyrene concentrations, for example, could only explain 2% and 7% of the variations in personal exposures in Waterfront South and Copewood–Davis, respectively, although each was statistically significant ($P = 0.0066$ and $P = 0.0629$, respectively; Table 19a). In addition, the slopes for all the compounds (except for phenanthrene) were greater for Copewood–Davis than for Waterfront South, though the ambient concentrations of all the compounds were higher in Waterfront South than in Copewood–Davis. These results suggest, again, that personal PAH exposures were affected by both outdoor and indoor sources as well as by personal activities. The associations between personal and ambient PAHs were greater in summer than in winter (Table 19b) for phenanthrene and benzo[a]pyrene, but these seasonal effects were not statistically significant

($P > 0.05$). There was no significant effect from the day of the week (Table 19c).

The relationship between personal and ambient concentrations was also evaluated by considering the confounding effect of ETS exposure. Nicotine measurements for 234 personal samples were included in the analysis. It was found that the association between personal and ambient concentrations in Copewood–Davis became less significant after incorporating the nicotine concentrations in the model (the P value increased from 0.0093 to 0.0263 for phenanthrene and from 0.0104 to 0.4552 for pyrene). These observations again indicated the significant impact of ETS exposure on personal exposures to PAHs and other pollutants in Copewood–Davis. There was no significant confounding effect of ETS on associations between personal and ambient PAHs in Waterfront South (the P value did not change when nicotine was included in the model). These results are consistent with our findings for $PM_{2.5}$, VOCs, and aldehydes.

SPATIAL VARIATION STUDY

Except for temperature and wind speed, meteorologic conditions were similar for all three saturation-sampling campaigns (Table 6). There was no rain or snowfall. The dominant wind directions were from the west or south-west–northwest; the August campaign also included winds from the east. Average wind speeds were 3.22 m/sec for the July campaign and 3.58 m/sec for the August campaign, both lower than the 4.41 m/sec average of the December campaign. Given the limited number of sampling campaigns in the study, no attempt was made to examine seasonal effects on spatial variations in VOCs or aldehydes.

Forty-two VOC and aldehyde samplers were used in each sampling campaign. The mean concentration, SD, and concentration range for each compound measured at the 22 Waterfront South sites and the 16 Copewood–Davis sites during each campaign are shown in Table 20. (The other 4 samplers consisted of a field blank and 3 duplicate samplers.) The P values obtained from Wilcoxon rank sum tests applied to the mean comparison (P_1) and variability comparison (P_2) of each compound are also shown in Table 20. 1,3-Butadiene and styrene were not reported, because more than half of the samples were below the MDLs. Hexane was not included, because of high lab blank levels in the July and August campaigns.

Concentrations and Spatial Distribution of BTEX and MTBE

The concentrations of VOCs and aldehydes measured in the spatial variation study are shown in Table 20. It is necessary to note that MTBE concentrations in the August

campaign might have been overestimated by a factor of ~3 because of a suspiciously lower response factor for GC–MS of MTBE compared with other calibration data. This, however, did not affect the spatial variability in MTBE observed in the August campaign. As discussed below, the variability was examined by way of the ratios and *R*'s of the concentrations measured at various locations, parameters that are not affected by calibration factors.

As shown in Table 20 and Figures 43 through 45 (for toluene, MTBE, and benzene only), spatial variations in VOC concentrations were observed in the three sampling campaigns; the degree of variability varied by species, day of the week, and location. These results thus provided the best opportunity to evaluate changes in short-term emissions from the various sources in Waterfront South, where large spatial variations in toluene, MTBE, ethylbenzene, and xylenes were observed. The greatest variability was found for toluene, with a relative standard deviation (%RSD) greater than 80% among measurements obtained from the 22 Waterfront South sampling sites during all sampling campaigns (Table 20). Multiple high values were found for toluene in Waterfront South (Figure 43), including highs of 32 $\mu\text{g}/\text{m}^3$ at site W2 in the July campaign, 60 $\mu\text{g}/\text{m}^3$ at site W19 in the August campaign, and 23 $\mu\text{g}/\text{m}^3$ at site W12 in the December campaign. (The sampling sites named here are identified in detail in Figure 1.) Toluene concentrations dropped by factors of two to five at locations farther from these sites and by additional factors of three to 12 at the remaining Waterfront South sampling sites. Concentrations were found to decay exponentially with distance from W19, the site with the highest concentration measured (60 $\mu\text{g}/\text{m}^3$). W19 was located ~50 m west of Pinto Brothers Recycling (Figure 46). Other toluene concentrations were 17.9 $\mu\text{g}/\text{m}^3$ at W20, 13.2 $\mu\text{g}/\text{m}^3$ at W13, 6.98 $\mu\text{g}/\text{m}^3$ at W14, and 4.49 $\mu\text{g}/\text{m}^3$ at W15. Large spatial variations were also observed for ethylbenzene (%RSD, 36% to 176%), *m*- & *p*-xylenes (40% to 175%), and *o*-xylene (39% to 134%); the variability of benzene (21% to 54%) was relatively small. In all three campaigns the highest concentrations of the compounds were always found at sites W3, W15, and W19. For MTBE the highest concentrations (29 $\mu\text{g}/\text{m}^3$ in the July campaign, 159 $\mu\text{g}/\text{m}^3$ in August, and 5.7 $\mu\text{g}/\text{m}^3$ in December) — three to nine times higher than the campaigns' means — were always found at site W3 (Figure 44).

Changes in concentrations and spatial variability were observed for these compounds from campaign to campaign. Mean concentrations and spatial variations of MTBE and TEX measured in the July and August campaigns were about two times higher than those in the December campaign (Table 20). The difference was caused

primarily by wind speed. As reported above, the average wind speed was higher in the winter campaign than in the two summer campaigns. Specifically, the highest wind speeds (5.7 to 8.8 m/sec) were recorded during 24.5% of the December campaign but only 3.6% of the summer campaigns. During the winter campaign, therefore, emissions from local sources were quickly diluted and well mixed by the higher winds, resulting in lower ambient concentrations and spatial variations of air toxics than those of the summer campaigns. In addition, given the lower temperatures (−0.7 to 3.3°C) of the winter campaign compared with those of the summer campaigns (21 to 33°C), evaporation of VOCs from local industrial operations would have been lower in winter, further lowering winter concentrations of these compounds.

To confirm the variability of our measured concentrations for MTBE and TEX in Waterfront South, the spatial variability reported above was compared with the sampling and analytic variability obtained from duplicate sample analyses made during the main exposure study (Table 11). The %RSDs were 5.7%, 6.4%, 11%, and 14% for toluene, MTBE, ethylbenzene, and xylenes, respectively, contributing less than 30% of the total variability for these compounds. These results indicate great spatial variability and the significance of stationary sources in contributing to MTBE and TEX in Waterfront South. The variability of benzene was not significantly different from the method variation (18%), indicating that benzene in the neighborhood is emitted primarily from mobile sources or recycling operations for junked vehicles.

In Copewood–Davis spatial variations in MTBE and BTEX were found to be small, with %RSD's ranging from 12% to 29% for MTBE, benzene, ethylbenzene, and xylenes and from 22% to 49% for toluene (Table 20). The spatial variability was not significantly different from the method variability. The highest concentrations of MTBE and BTEX in Copewood–Davis were observed at sites located along NJ-168 (Figures 43 through 45), particularly at site C6, which had a high frequency of stopped and idling traffic. However, all MTBE values (except for a value of 18 $\mu\text{g}/\text{m}^3$ measured in the August campaign) were below 10 $\mu\text{g}/\text{m}^3$. The highest concentration of individual compounds measured in Copewood–Davis was about two times the corresponding mean.

The concentrations and spatial variations obtained at the Waterfront South sites were compared with those from Copewood–Davis. During the July and August sampling campaigns, the variability of TEX was significantly greater in Waterfront South (P_2 was 0.057 for toluene in July; P_2 's for the rest of the measurements were < 0.05; Table 20). In the August campaign the variability of benzene was also

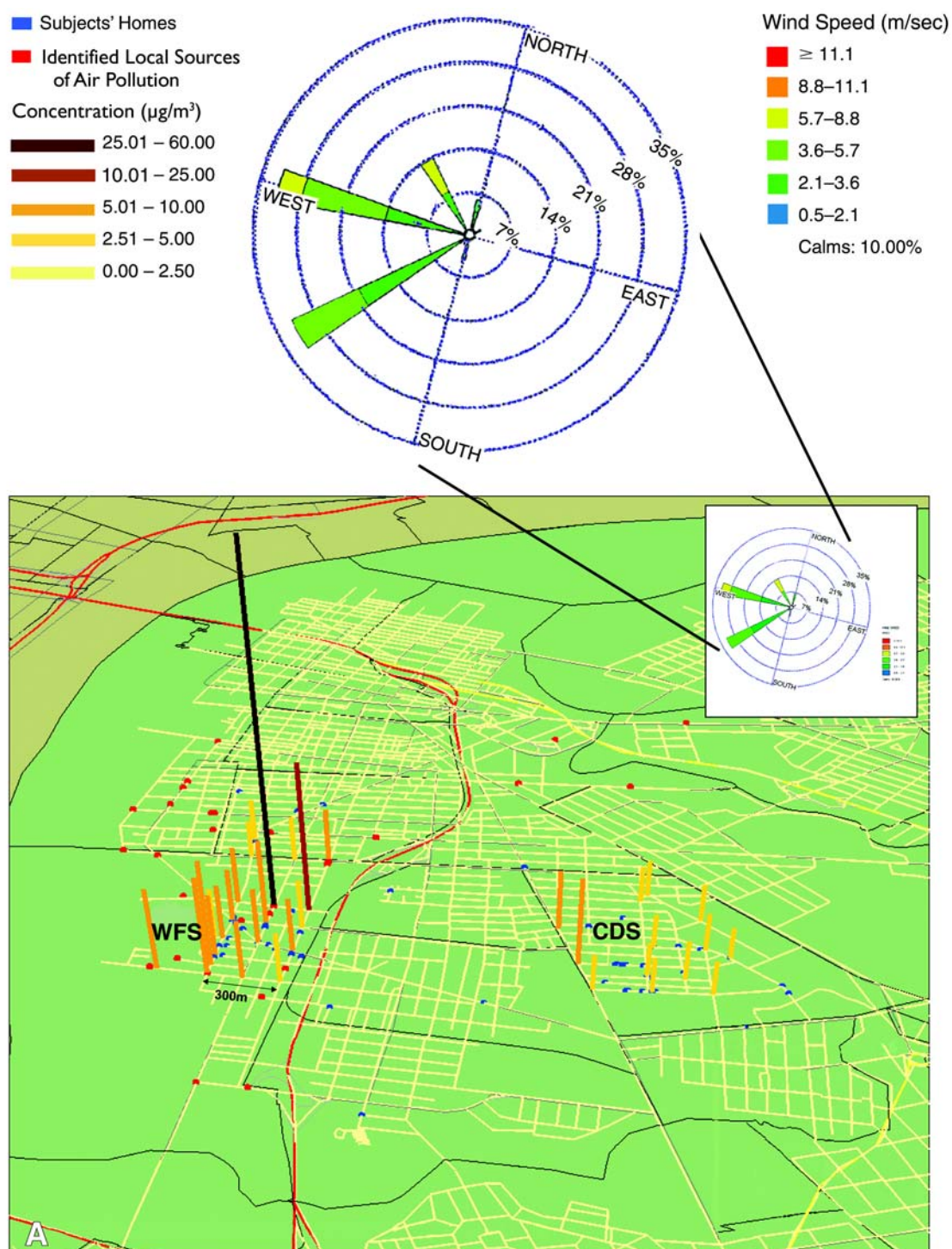


Figure 43. Spatial distribution of toluene for three saturation sampling campaigns on (A) July 20–22, (B) August 17–18, and (C) December 21–23, 2005. The height of the colored bars is proportional to the concentration they represent. In the wind rose, spoke lengths indicate the frequency of wind from the given directions. (Wind directions shown are the directions from which the wind was blowing.) (Figure continues on next two pages)

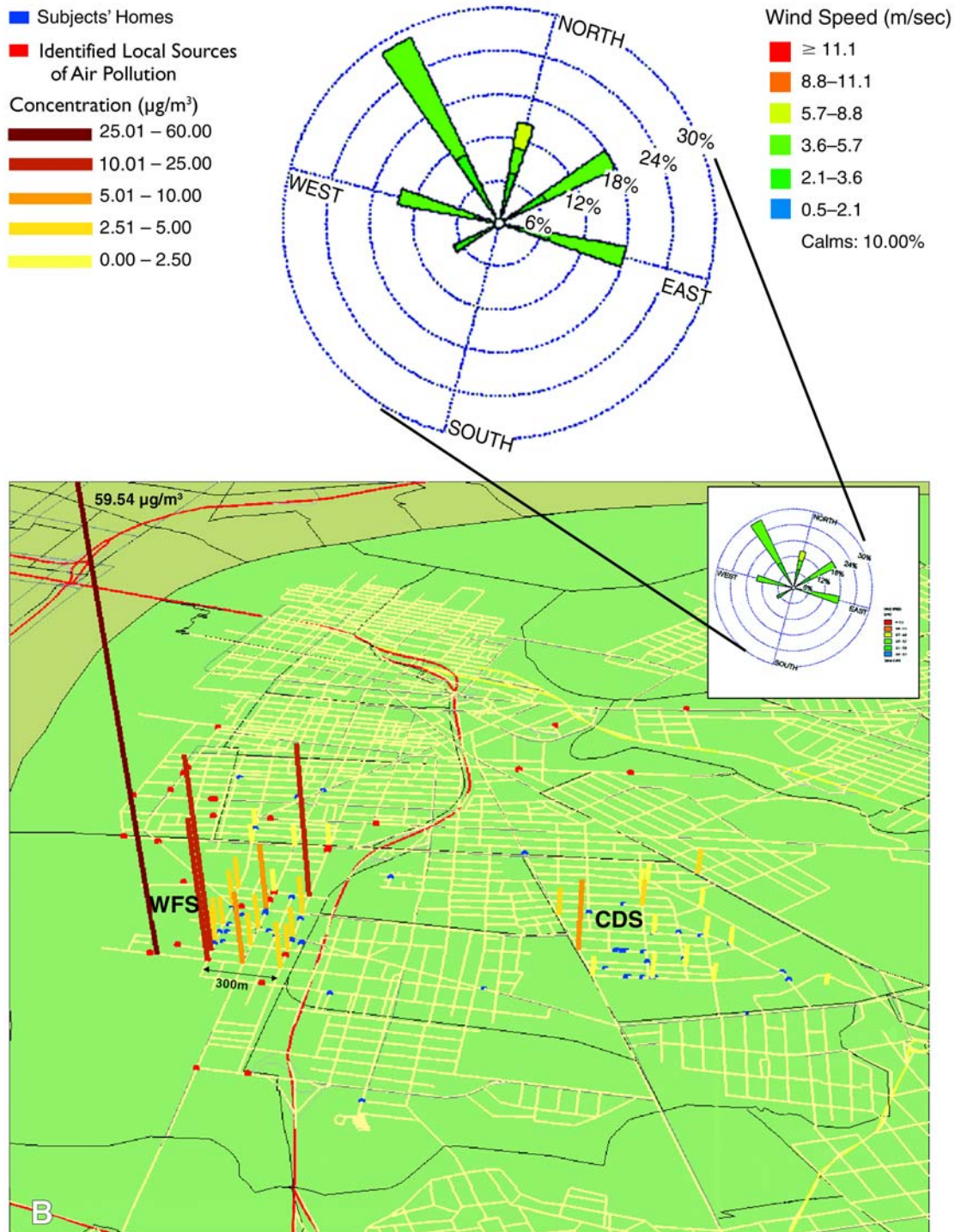
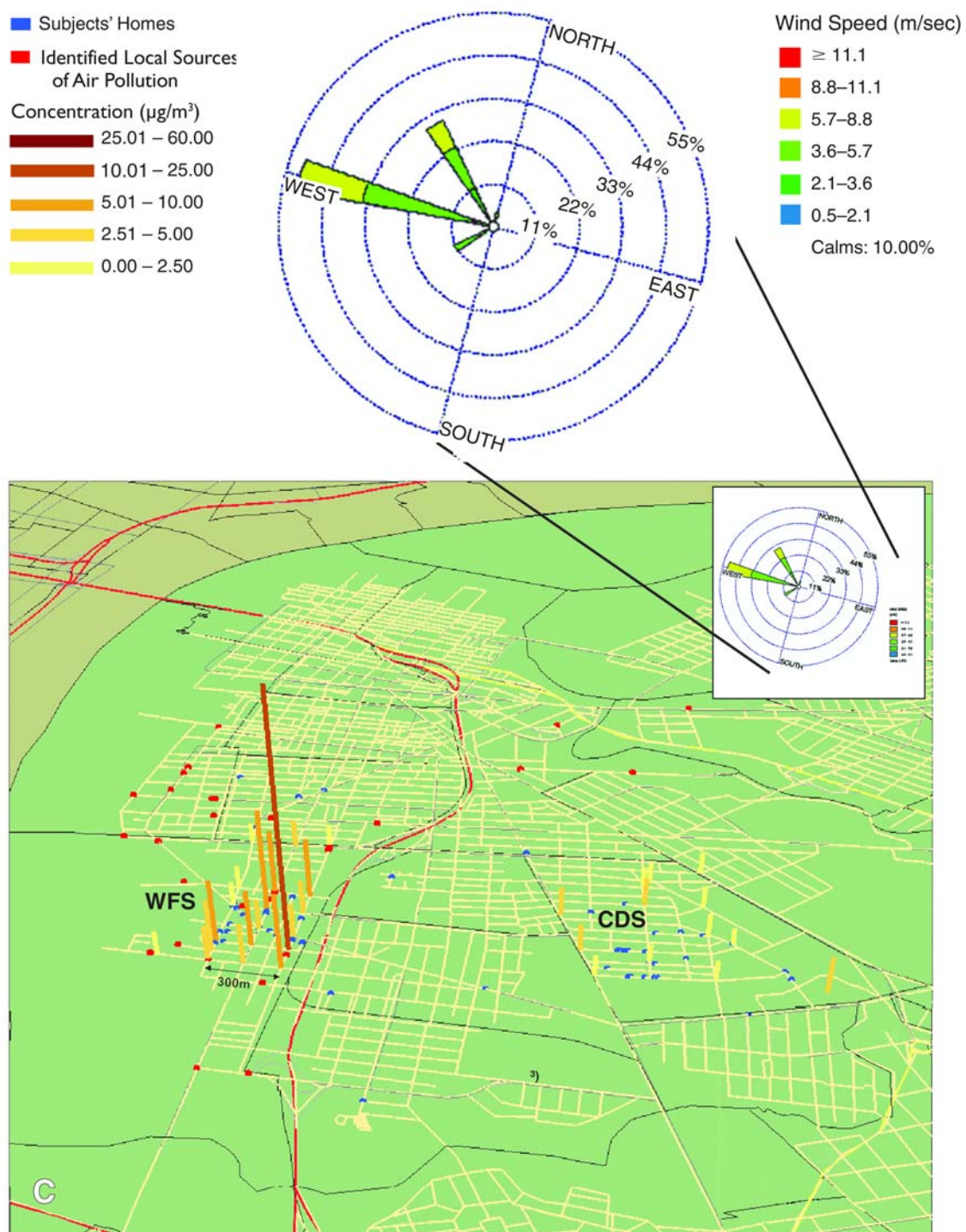


Figure 43 (Continued). (B) August 17–18, 2005. (Figure continues on next page)



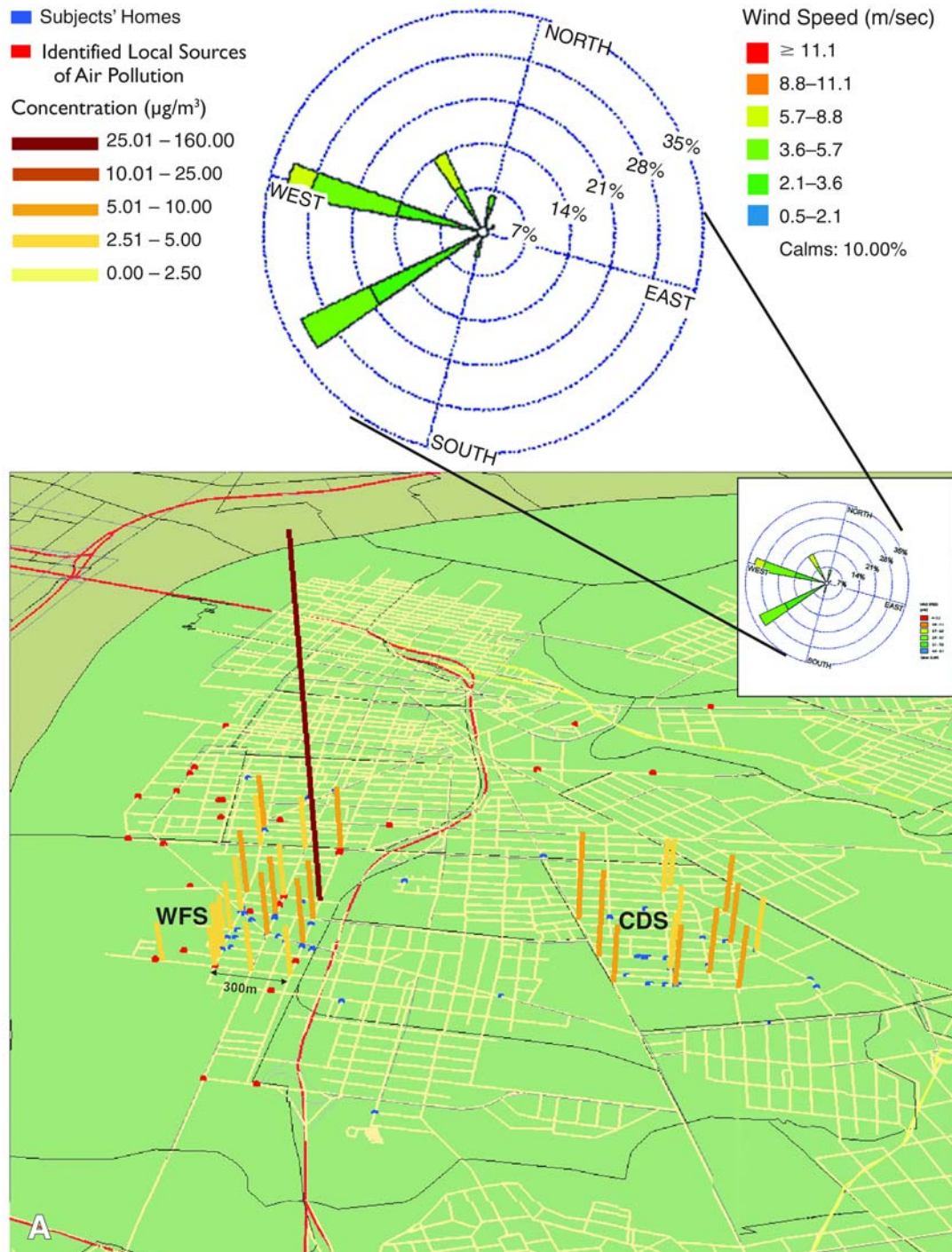


Figure 44. Spatial distribution of MTBE for three saturation sampling campaigns on (A) July 20–22, (B) August 17–18, and (C) December 21–23, 2005. The height of the colored bars is proportional to the concentration they represent. In the wind rose, spoke lengths indicate the frequency of wind from the given directions. (Wind directions shown are the directions from which the wind was blowing.) (Figure continues on next two pages)

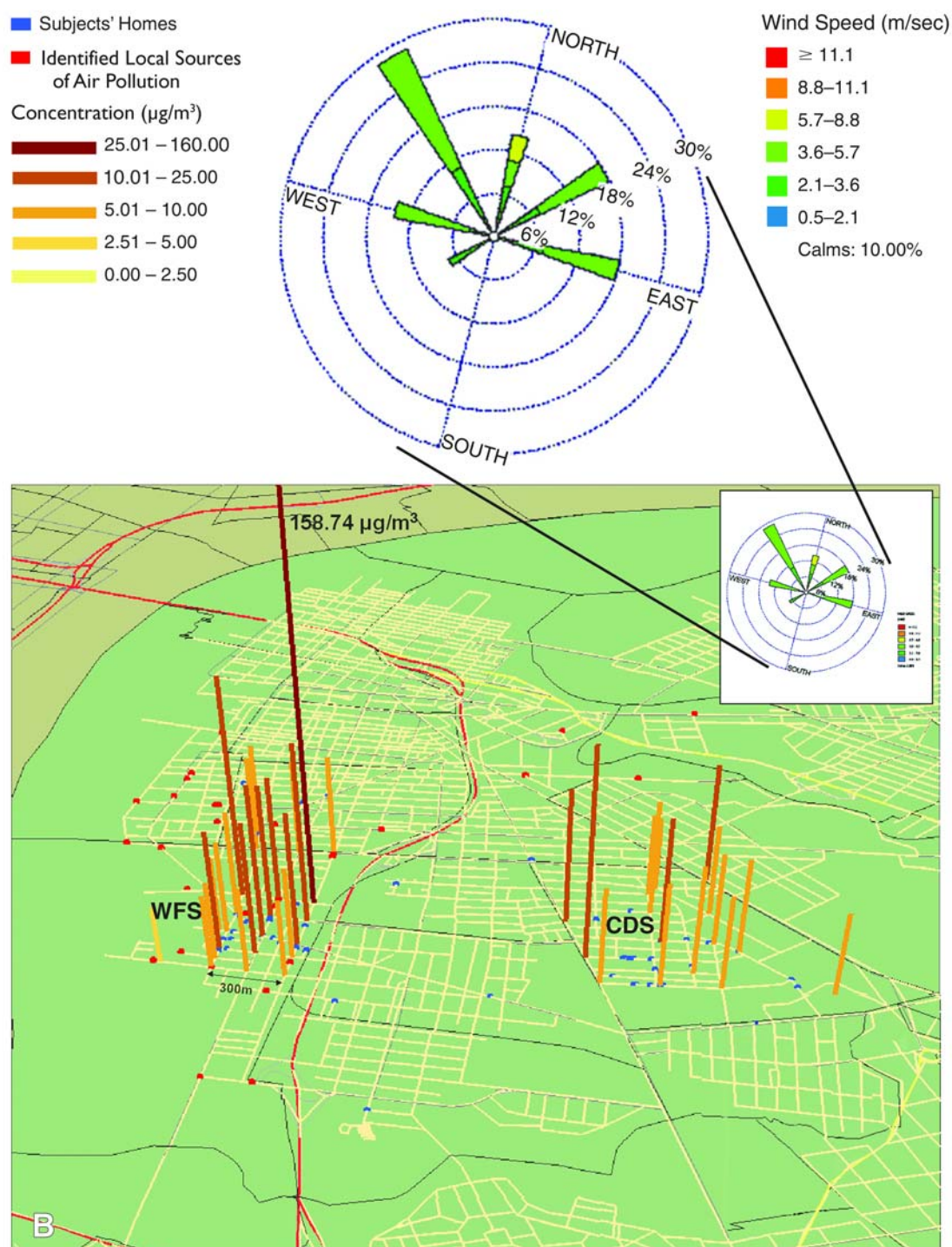


Figure 44 (Continued). (B) August 17–18, 2005. (Figure continues on next page)

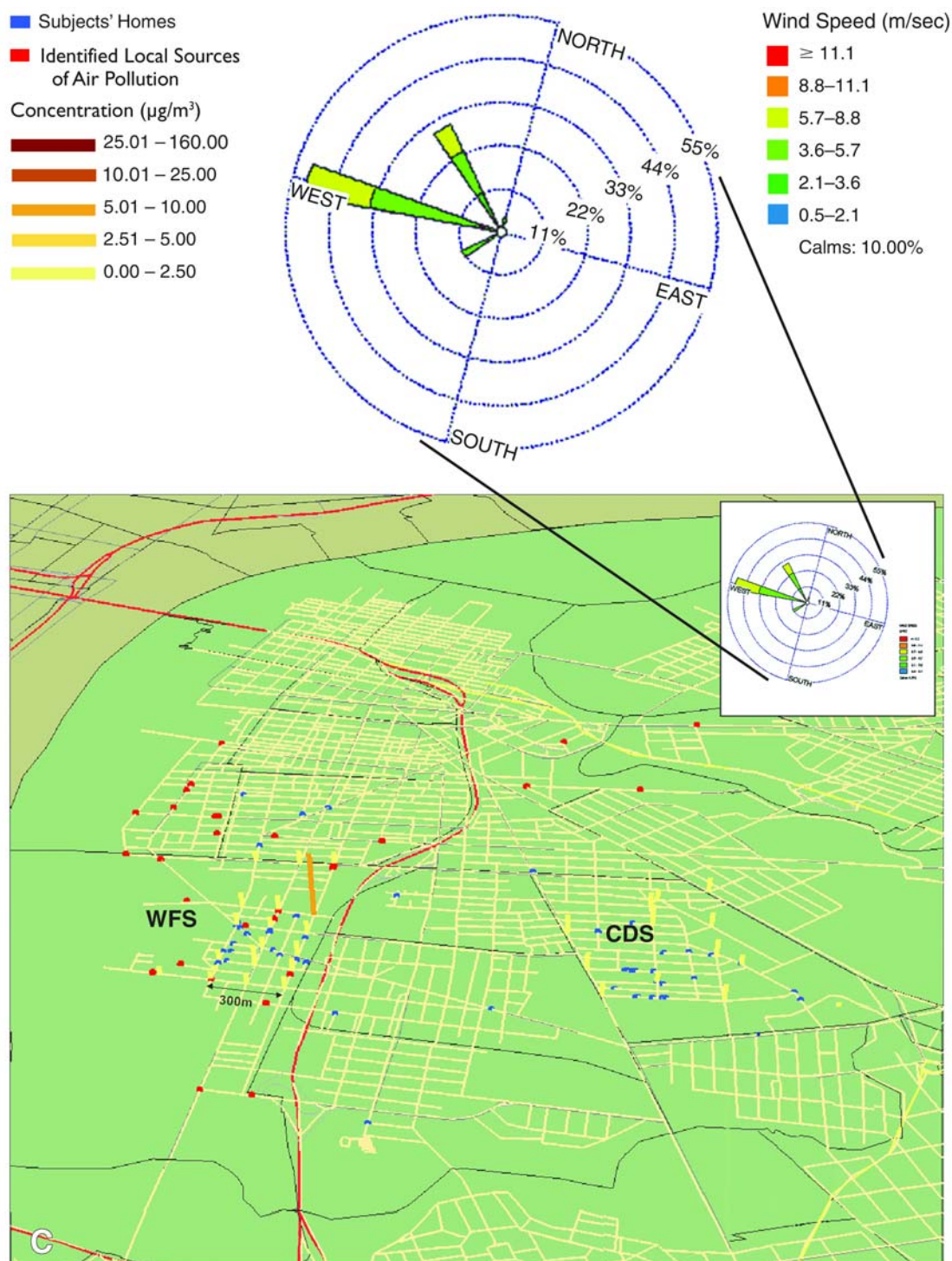


Figure 44 (Continued). (C) December 21–23, 2005.

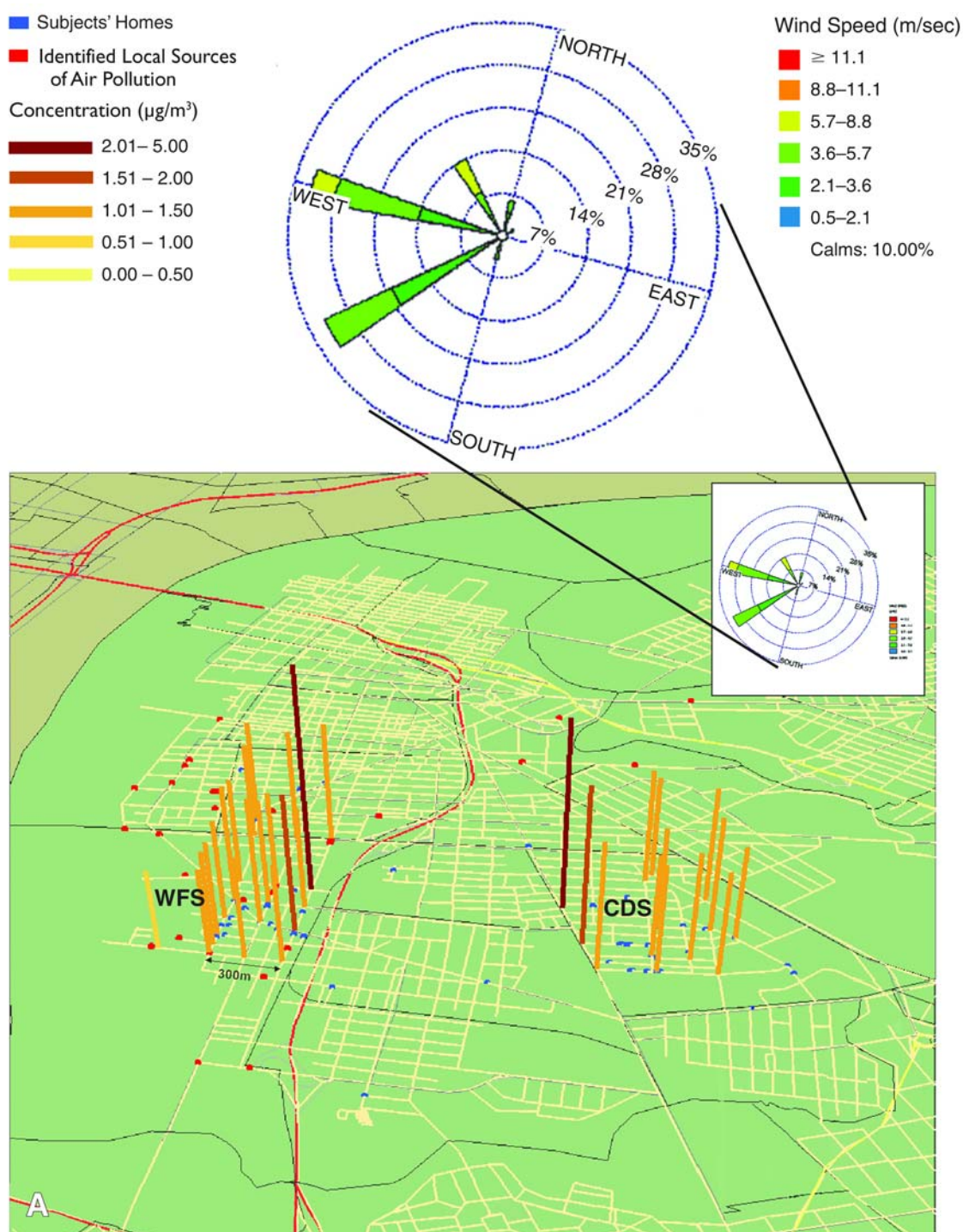


Figure 45. Spatial distribution of benzene for three saturation sampling campaigns on (A) July 20–22, (B) August 17–18, and (C) December 21–23, 2005. The height of the colored bars is proportional to the concentration they represent. In the wind rose, spoke lengths indicate the frequency of wind from the given directions. (Wind directions shown are the directions from which the wind was blowing.) (Figure continues on next two pages)

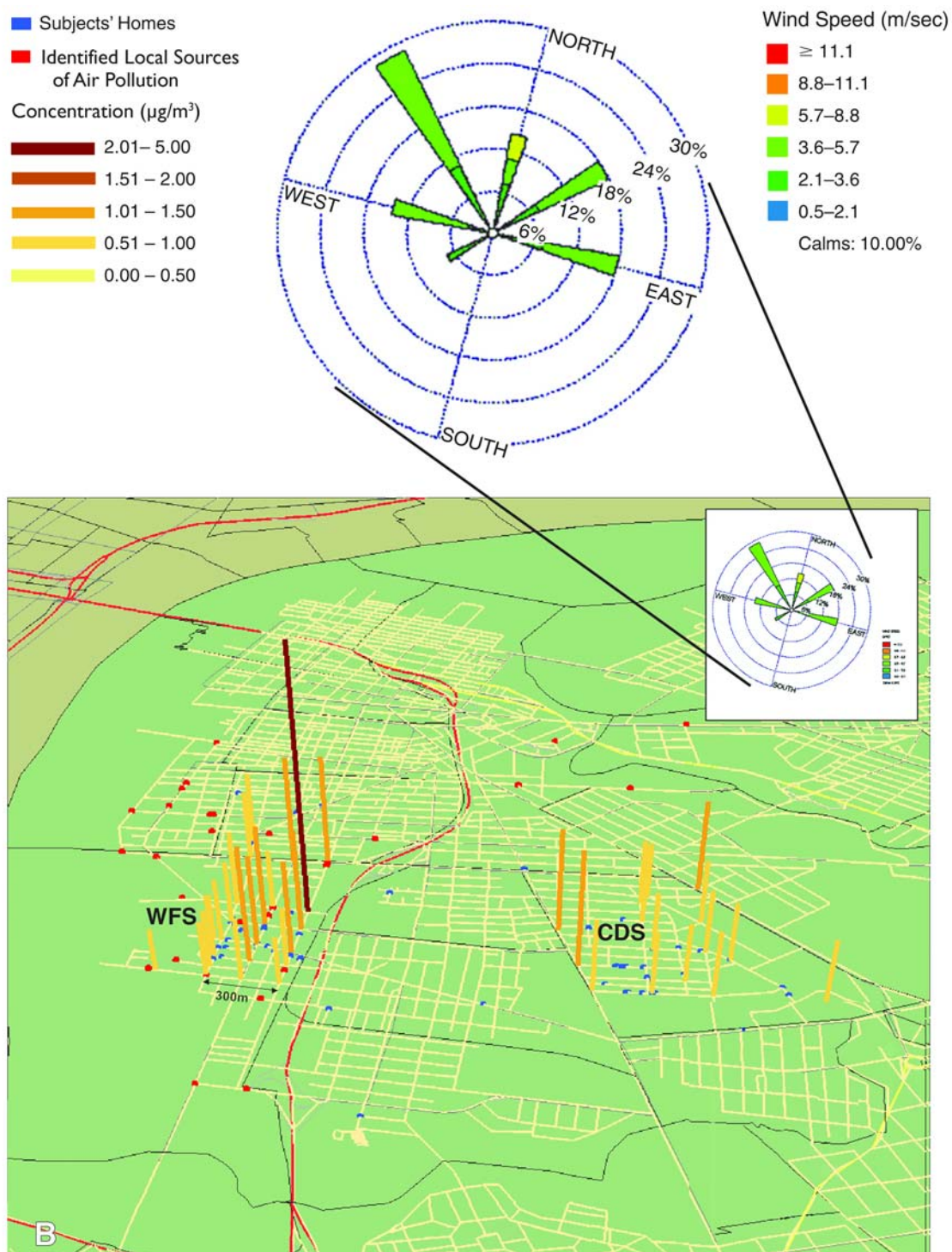


Figure 45 (Continued). (B) August 17–18, 2005. (Figure continues on next page)

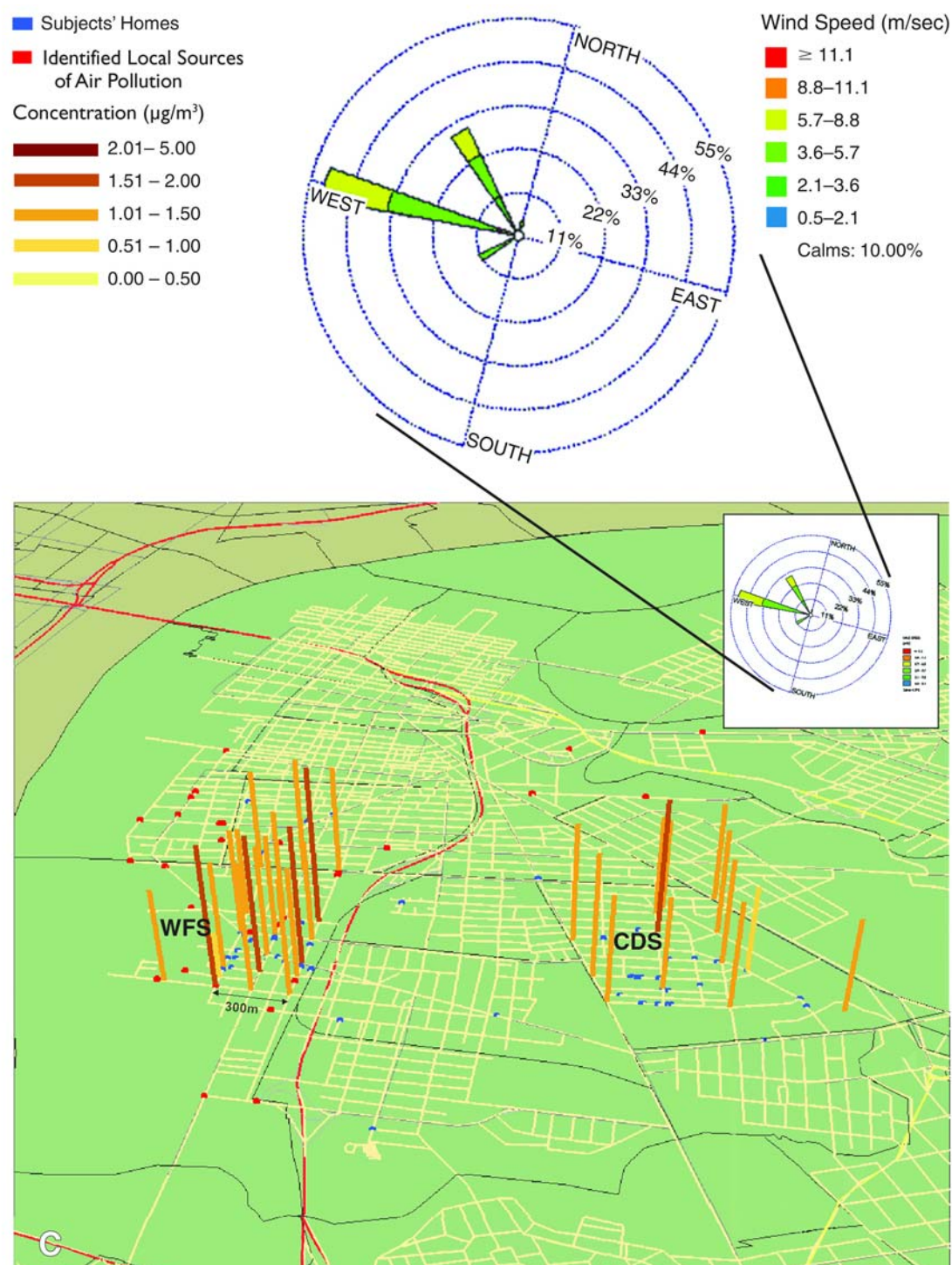


Figure 45 (Continued). (C) December 21–23, 2005.

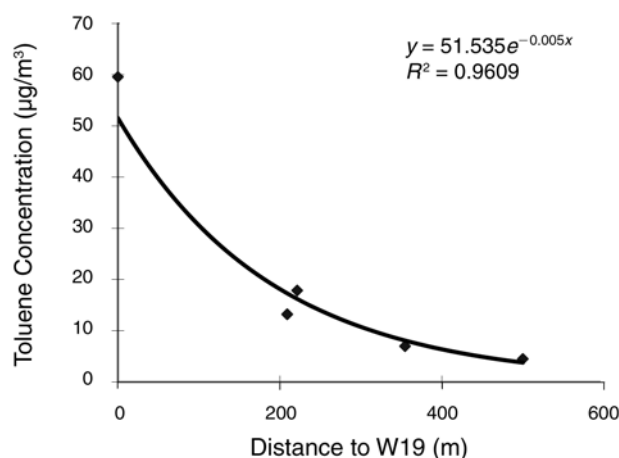


Figure 46. Toluene concentrations downwind of site W19, a recycling facility, measured during the August 17–18, 2005, saturation sampling campaign.

found to be marginally higher ($P_2 = 0.057$) in Waterfront South. Similar trends were observed in the December campaign, but only toluene was significantly higher ($P_2 = 0.007$); ethylbenzene was marginally higher ($P_2 = 0.064$) in Waterfront South. The mean concentrations of ethylbenzene, xylenes, and toluene were significantly ($P_1 < 0.05$) or marginally ($P_1 < 0.1$) higher in Waterfront South than in Copewood–Davis for the July campaign, but no differences were found for benzene. For MTBE there were no significant differences in either mean concentration ($P_1 > 0.412$) or variability ($P_2 > 0.164$), although the %RSD's were found to be large in Waterfront South (76% to 196%). The large %RSD's for MTBE were driven primarily by one high value measured at site W3. When this high value was excluded, the %RSD's dropped to a range of 17% to 35% for the three campaigns.

Sources of MTBE and BTEX in Waterfront South and Copewood–Davis

Waterfront South The above results are in good agreement with emissions inventory data reported for Waterfront South by the NJDEP (2005) (Table 1). Toluene, ethylbenzene, and xylenes have been identified as major pollutants associated with operations at several local industrial facilities, including industrial paint shops, welding and soldering equipment manufacturers with paint application, automotive body repair shops, and recycling plants. Emissions from these facilities can contribute significantly to the air pollution in Waterfront South. This assessment is supported by correlation analysis results obtained among all species in both neighborhoods. As shown in Table 21, poor correlations were found between

MTBE and TEX and between benzene and TEX in Waterfront South. R 's were less than 0.2 for the July and August sampling campaigns. Moreover, BTEX ratios for sites adjacent to industrial facilities were 1:6:1:5 at site W3 in the July campaign and 0.1:0.5:1:3 and 0.3:37:1:3 at sites W15 and W19, respectively, in the August campaign. These values were significantly different from those observed in areas dominated by onroad vehicle emissions (not vehicle recycling) (Harkov et al. 1983; Watson et al. 2001; Lee et al. 2002; Ohura et al. 2006). In Waterfront South benzene was highly correlated with MTBE in the July and August campaigns (with R 's of 0.92 and 0.79, respectively) but poorly correlated in the December campaign (R of 0.37), suggesting the impact of automobile exhaust on benzene concentrations in the neighborhood.

For MTBE the extremely high concentrations observed at site W3 suggested a significant point source nearby; an automobile recycling company is in fact located < 50 m from the site, where heavy-duty trucks carrying recyclable vehicles and gasoline leaking from scrapped vehicles were observed. MTBE, which was still being used as a fuel oxygenate in New Jersey at the time the samples were collected, evaporates, resulting in the high concentrations of MTBE measured at the site and at adjacent areas during the three campaigns. This source is also likely to have contributed to the high concentrations of benzene, toluene, ethylbenzene, and xylenes observed at the same site. However, the large %RSD's for MTBE were driven primarily by one high value from site W3. When this high value was excluded, the %RSD's dropped to a range of 17% to 35% (from 76% to 196%) for the three campaigns, only 10% higher than that of the method variability. The results thus suggest that the impact of this facility on MTBE concentrations was limited to locations < 300 m away.

Copewood–Davis There are no identified industrial facilities in or near Copewood–Davis, with the result that VOC concentrations in the neighborhood are affected primarily by area sources, such as automobile exhaust from local traffic and ambient pollutants transported from Philadelphia and other areas. This assessment is supported by correlation analyses showing good correlations among the BTEX compounds and between MTBE and BTEX. The R 's were 0.6 or greater for benzene and toluene and, except for the R between MTBE and TEX in the July campaign and between MTBE and benzene in the August campaign, were 0.8 or greater for the rest of these compounds (Table 21). BTEX ratios were also within the range of 2:5:1:4 to 2:9:1:4, with an average of 2:8:1:4, comparable with the BTEX ratios for gasoline exhaust and evaporated gasoline reported by Watson and colleagues (2001). These results

Table 21. Correlation Coefficients Among VOCs and Aldehydes for Three Sampling Campaigns in Waterfront South and Copewood–Davis

Compound	MTBE	Chloro- form	Carbon tetra- chloride	Benzene	Toluene	Ethyl- benzene	<i>m- & p-</i> Xylenes	<i>o</i> -Xylene	Formalde- hyde	Acetalde- hyde	Acrolein
July 20–22, 2005—Waterfront South											
MTBE	1.00										
Chloroform	–0.44	1.00									
Carbon tetrachloride	0.04	0.58	1.00								
Benzene	0.92	–0.37	0.06	1.00							
Toluene	0.06	–0.06	0.25	–0.06	1.00						
Ethylbenzene	0.25	0.04	0.03	0.19	0.39	1.00					
<i>m- & p</i> -Xylenes	0.26	0.03	0.04	0.20	0.40	1.00	1.00				
<i>o</i> -Xylene	0.43	0.03	0.11	0.38	0.32	0.94	0.95	1.00			
Formaldehyde	0.11	0.38	0.02	0.22	–0.31	0.05	0.07	0.12	1.00		
Acetaldehyde	–0.11	0.40	–0.08	0.13	–0.36	–0.11	–0.09	–0.03	0.86	1.00	
Acrolein	–0.08	–0.02	0.29	–0.12	0.59	0.14	0.15	0.01	–0.10	–0.27	1.00
July 20–22, 2005—Copewood–Davis											
MTBE	1.00										
Chloroform	–0.37	1.00									
Carbon tetrachloride	0.27	–0.09	1.00								
Benzene	0.64	–0.08	0.73	1.00							
Toluene	0.58	–0.14	0.69	0.74	1.00						
Ethylbenzene	0.34	0.11	0.69	0.88	0.66	1.00					
<i>m- & p</i> -Xylenes	0.43	0.07	0.64	0.91	0.62	0.98	1.00				
<i>o</i> -Xylene	0.52	–0.04	0.62	0.90	0.72	0.90	0.93	1.00			
Formaldehyde	–0.10	–0.10	–0.12	–0.34	–0.04	–0.36	–0.39	–0.40	1.00		
Acetaldehyde	0.10	–0.05	–0.36	–0.28	–0.13	–0.33	–0.32	–0.38	0.82	1.00	
Acrolein	0.15	0.35	0.29	0.54	0.45	0.59	0.62	0.74	–0.54	–0.63	1.00
August 17–18, 2005—Waterfront South											
MTBE	1.00										
Chloroform	–0.70	1.00									
Carbon tetrachloride	0.03	0.24	1.00								
Benzene	0.79	–0.61	–0.06	1.00							
Toluene	–0.09	0.11	–0.16	–0.19	1.00						
Ethylbenzene	0.02	0.17	–0.03	0.05	0.77	1.00					
<i>m- & p</i> -Xylenes	0.04	0.17	–0.01	0.02	0.76	0.99	1.00				
<i>o</i> -Xylene	0.13	0.09	–0.06	0.11	0.67	0.96	0.97	1.00			
Formaldehyde	–0.10	0.25	0.05	–0.11	0.39	0.33	0.34	0.31	1.00		
Acetaldehyde	0.17	–0.05	–0.15	0.04	–0.34	–0.45	–0.43	–0.44	–0.25	1.00	
Acrolein	–0.31	0.20	–0.22	–0.05	0.29	0.23	0.20	0.19	0.20	0.03	1.00

Table continues next page

Table 21 (Continued). Correlation Coefficients Among VOCs and Aldehydes for Three Sampling Campaigns in Waterfront South and Copewood–Davis

Compound	MTBE	Chloro- form	Carbon tetra- chloride	Benzene	Toluene	Ethyl- benzene	<i>m- & p-</i> Xylenes	<i>o</i> -Xylene	Formalde- hyde	Acetalde- hyde	Acrolein
August 17–18, 2005—Copewood–Davis											
MTBE	1.00										
Chloroform	0.33	1.00									
Carbon tetrachloride	0.04	0.01	1.00								
Benzene	0.45	0.23	0.64	1.00							
Toluene	0.69	0.36	0.31	0.59	1.00						
Ethylbenzene	0.78	0.13	0.18	0.68	0.74	1.00					
<i>m- & p</i> -Xylenes	0.82	0.18	0.15	0.67	0.80	0.97	1.00				
<i>o</i> -Xylene	0.79	0.20	0.16	0.75	0.78	0.92	0.96	1.00			
Formaldehyde	0.55	0.11	0.25	0.42	0.35	0.60	0.55	0.42	1.00		
Acetaldehyde	−0.08	−0.20	0.03	0.00	−0.27	−0.15	−0.19	−0.05	−0.14	1.00	
Acrolein	0.74	0.10	0.06	0.39	0.71	0.79	0.84	0.73	0.46	−0.34	1.00
December 20–22, 2005—Waterfront South											
MTBE	1.00										
Chloroform	−0.08	1.00									
Carbon tetrachloride	0.01	0.57	1.00								
Benzene	0.37	0.30	0.54	1.00							
Toluene	0.12	0.27	0.32	0.31	1.00						
Ethylbenzene	0.52	0.23	0.38	0.58	0.68	1.00					
<i>m- & p</i> -Xylenes	0.63	0.22	0.34	0.57	0.66	0.98	1.00				
<i>o</i> -Xylene	0.67	0.23	0.33	0.53	0.56	0.95	0.98	1.00			
Formaldehyde	−0.17	−0.15	−0.35	−0.20	−0.09	−0.09	−0.07	−0.10	1.00		
Acetaldehyde	−0.44	0.02	−0.06	0.07	0.46	0.11	0.08	−0.04	0.53	1.00	
Acrolein	0.32	0.00	0.02	0.06	−0.48	−0.24	−0.19	−0.09	−0.28	−0.51	1.00
December 20–22, 2005—Copewood–Davis											
MTBE	1.00										
Chloroform	−0.45	1.00									
Carbon tetrachloride	0.19	0.43	1.00								
Benzene	0.85	−0.04	0.48	1.00							
Toluene	0.59	0.02	0.25	0.64	1.00						
Ethylbenzene	0.91	−0.27	0.29	0.90	0.60	1.00					
<i>m- & p</i> -Xylenes	0.84	−0.11	0.26	0.86	0.63	0.96	1.00				
<i>o</i> -Xylene	0.83	−0.09	0.24	0.85	0.54	0.93	0.97	1.00			
Formaldehyde	−0.15	0.03	0.15	−0.02	0.32	0.08	0.13	0.11	1.00		
Acetaldehyde	−0.12	−0.25	−0.64	−0.32	0.17	−0.09	−0.05	−0.15	0.12	1.00	
Acrolein	−0.47	0.46	−0.08	−0.27	−0.32	−0.28	−0.13	−0.05	0.06	0.00	1.00

indicate that onroad vehicle emissions were the main sources of VOCs in Copewood–Davis.

Concentrations and Spatial Distribution of Chloroform and Carbon Tetrachloride

The distributions of chloroform and carbon tetrachloride concentrations in the two neighborhoods were comparable, with %RSD's of < 20% for chloroform and < 10% for carbon tetrachloride. The concentrations of both compounds were close to regional background concentrations in North America (McCarthy et al. 2007; Touma et al. 2006), ranging from 0.03 to 0.29 $\mu\text{g}/\text{m}^3$ for chloroform and 0.19 to 0.60 $\mu\text{g}/\text{m}^3$ for carbon tetrachloride in both neighborhoods for the three sampling campaigns. There were no significant differences in mean concentrations or variances between Waterfront South and Copewood–Davis ($P > 0.05$). Poor correlations (most R 's were less than 0.3; Table 21) were also observed between the two compounds and between MTBE and BTEX in both neighborhoods.

Concentrations and Spatial Distribution of Aldehydes

The concentrations of formaldehyde, acetaldehyde, and acrolein measured at many of the sampling sites in both Waterfront South and Copewood–Davis (Table 20 and Figures 47 through 49) were higher than ambient concentrations observed in other urban areas (Kinney et al. 2002; McCarthy et al. 2007; Weisel et al. 2005; Touma et al. 2006). Mean concentrations were > 12 $\mu\text{g}/\text{m}^3$ for formaldehyde for all three campaigns, > 12 $\mu\text{g}/\text{m}^3$ for acetaldehyde for the July campaign, and > 3.5 $\mu\text{g}/\text{m}^3$ for acrolein for August. These results indicate that there were significant sources of aldehydes in the two neighborhoods. The concentrations of all aldehydes measured in the two summer campaigns (except acrolein in July) were higher than those in the December campaign (Table 20 and Figure 47). As discussed earlier for VOCs, wind speeds explain the seasonal differences in the observed concentrations.

Spatial variations in the aldehydes were examined using measurements obtained from the three sampling campaigns. Similar spatial distributions for all three aldehydes were found for Waterfront South and Copewood–Davis. No significant spatial variations were observed for formaldehyde at any of the sampling sites in either neighborhood. The maximum formaldehyde concentration measured was twice as high as the minimum, and %RSD's were < 45% in each campaign. In contrast, large variations in concentration were found for acetaldehyde and acrolein in both Waterfront South and Copewood–Davis, where %RSD's ranged from 65% to 107% for acetaldehyde and from 39% to 119% for acrolein. It is worth noting that the large variability in acrolein in the December campaign was

caused primarily by the presence of low acrolein concentrations (many of them nondetectable). A significant portion of the apparent spatial variation in aldehydes also resulted from large variability in the sampling and analysis methods (~40%) used for aldehyde measurements in this study. No significant differences ($P > 0.05$) were found in mean aldehyde concentrations between Waterfront South and Copewood–Davis.

Sources of Aldehydes

High concentrations of formaldehyde (mean > 12 $\mu\text{g}/\text{m}^3$) with relatively low variability (%RSD < 45%) found in Waterfront South and Copewood–Davis (Figure 47) indicate significant impacts from very local outdoor area sources — i.e., motor vehicles in the two neighborhoods. Diesel truck emissions are well-known sources of formaldehyde in most urban areas (Grosjean et al. 2001; HEI 1995). As reported by the NJDEP (2005), 275 to 825 trucks per day pass through Waterfront South to deliver materials to industrial facilities in and near the neighborhood, such as the South Jersey Port Corporation, Camden Iron & Metal, and the St. Lawrence Cement plant. High concentrations of formaldehyde and other air toxics are emitted by the trucks, resulting in the neighborhood's high formaldehyde concentrations. Given the many truck routes and loading areas in and near Waterfront South and the fact that Copewood–Davis is near and downwind of Waterfront South, truck emissions of formaldehyde were ubiquitous and resulted in very limited spatial variation for formaldehyde. Although formaldehyde has been identified as one of the major pollutants from several industrial sources (NJDEP 2006), such as sewage plants, contributions from local industrial sources are less significant than the large quantities emitted by diesel trucks. These conclusions were supported by our correlation analyses. Poor correlations ($R < 0.3$; Table 21) were found between aldehydes and BTEX or MTBE. This was expected, because diesel exhaust contains higher aldehyde concentrations but lower BTEX and no MTBE concentrations.

Photochemical reactions can also contribute to the formaldehyde in ambient air. VOCs, the precursors of formaldehyde, are emitted in large quantities by local industrial sources and can react under photochemical conditions to form formaldehyde. This, however, does not result in localized formaldehyde hot spots; hence the low spatial variation in formaldehyde.

The large spatial variations in acetaldehyde and acrolein observed in the July and August campaigns indicate that, besides mobile sources, stationary sources of these two compounds were present in or near Waterfront South. However, emissions inventory data for the two compounds were

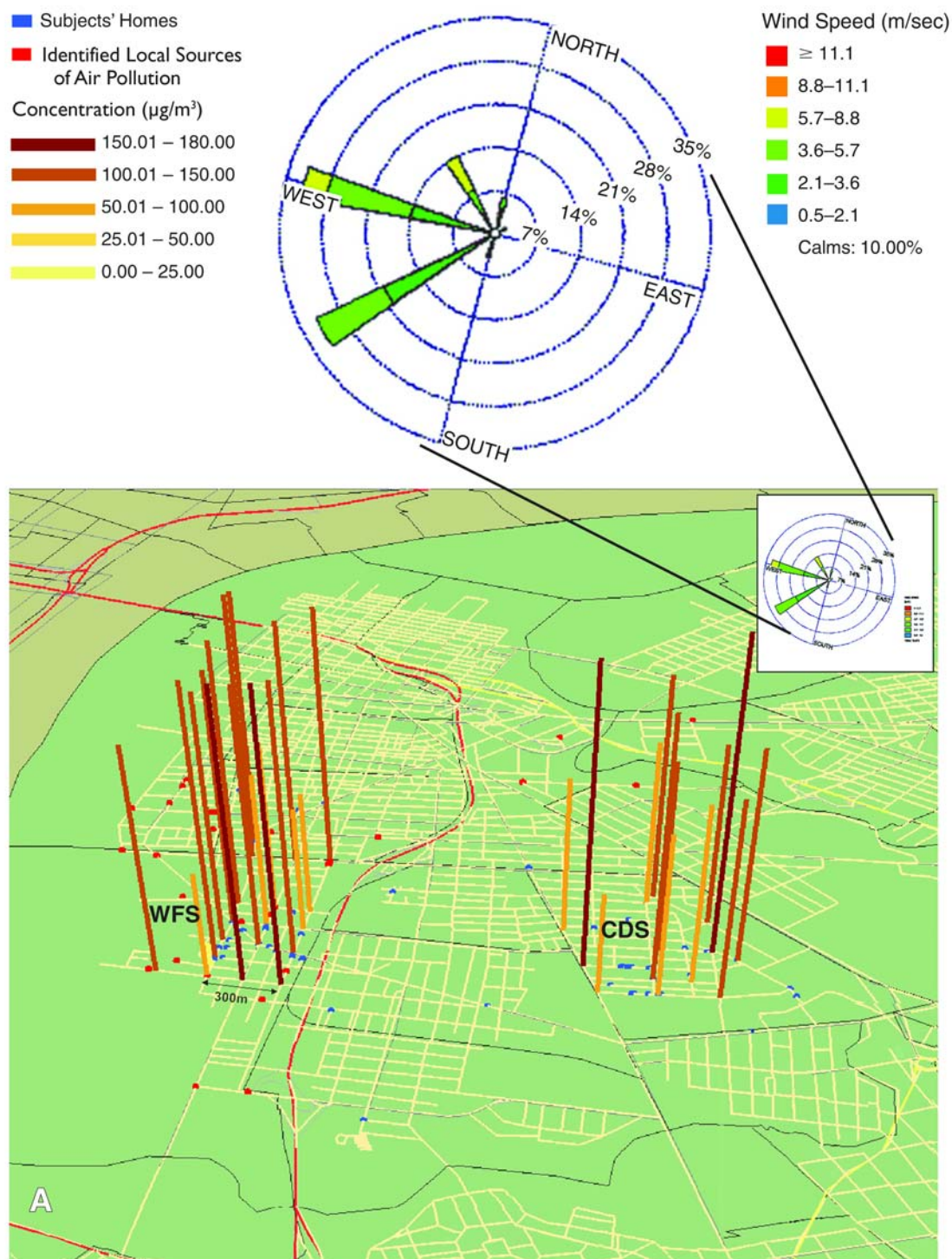


Figure 47. Spatial distribution of formaldehyde for three saturation sampling campaigns on (A) July 20–22, (B) August 17–18, and (C) December 21–23, 2005. The height of the colored bars is proportional to the concentration they represent. In the wind rose, spoke lengths indicate the frequency of wind from the given directions. (Wind directions shown are the directions from which the wind was blowing.) (Figure continues on next two pages)

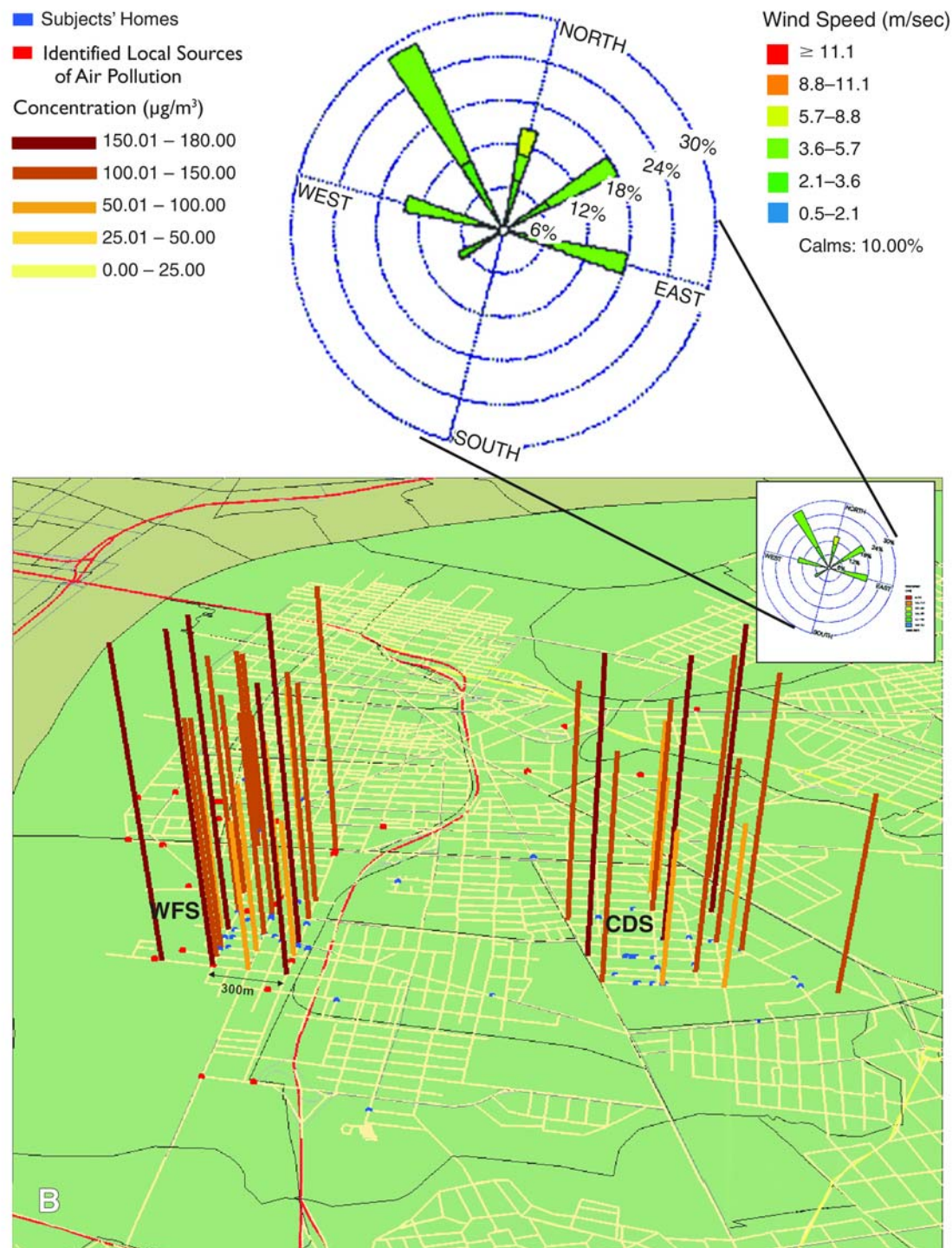


Figure 47 (Continued). (B) August 17–18, 2005. (Figure continues on next page)

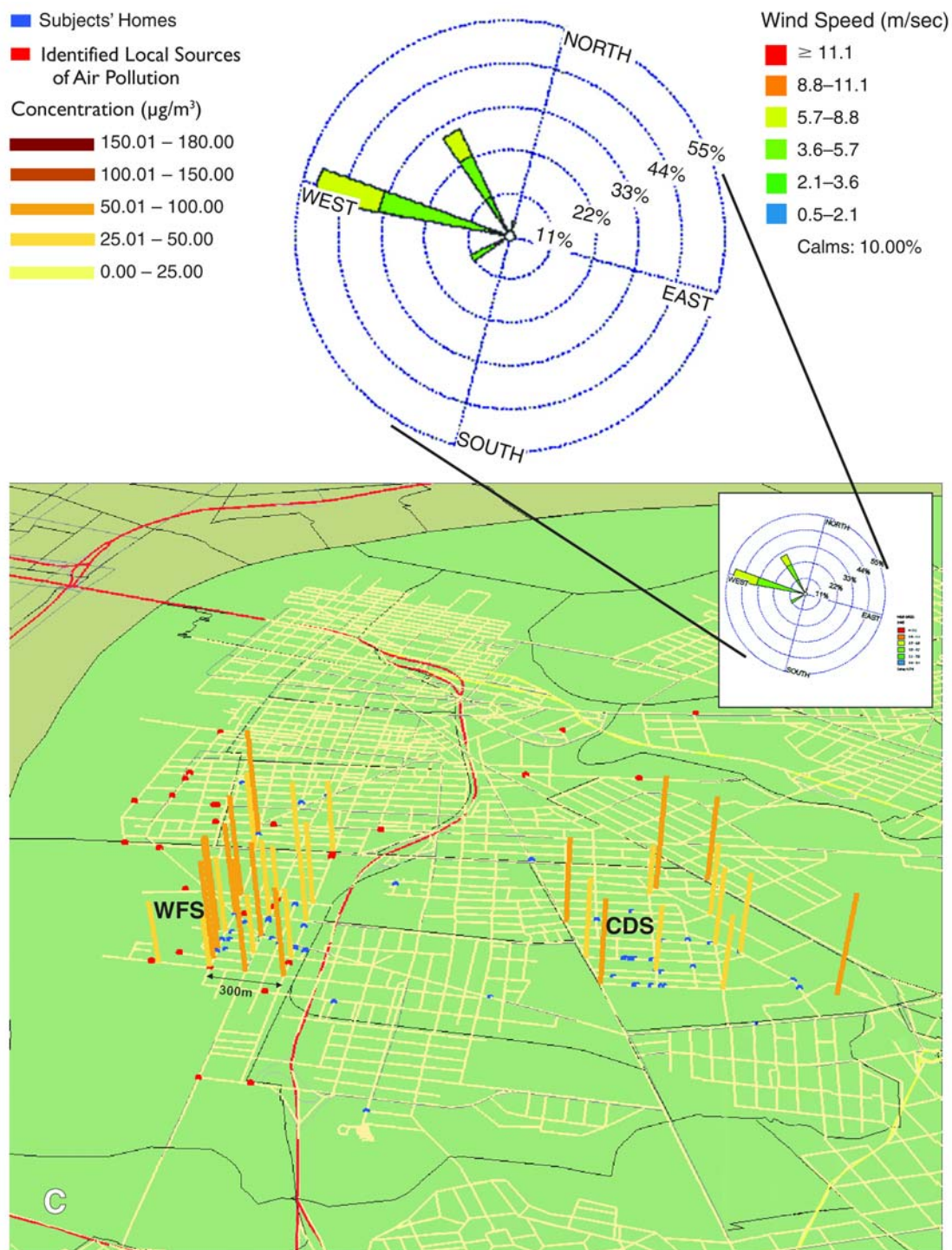


Figure 47 (Continued). (C) December 21–23, 2005.

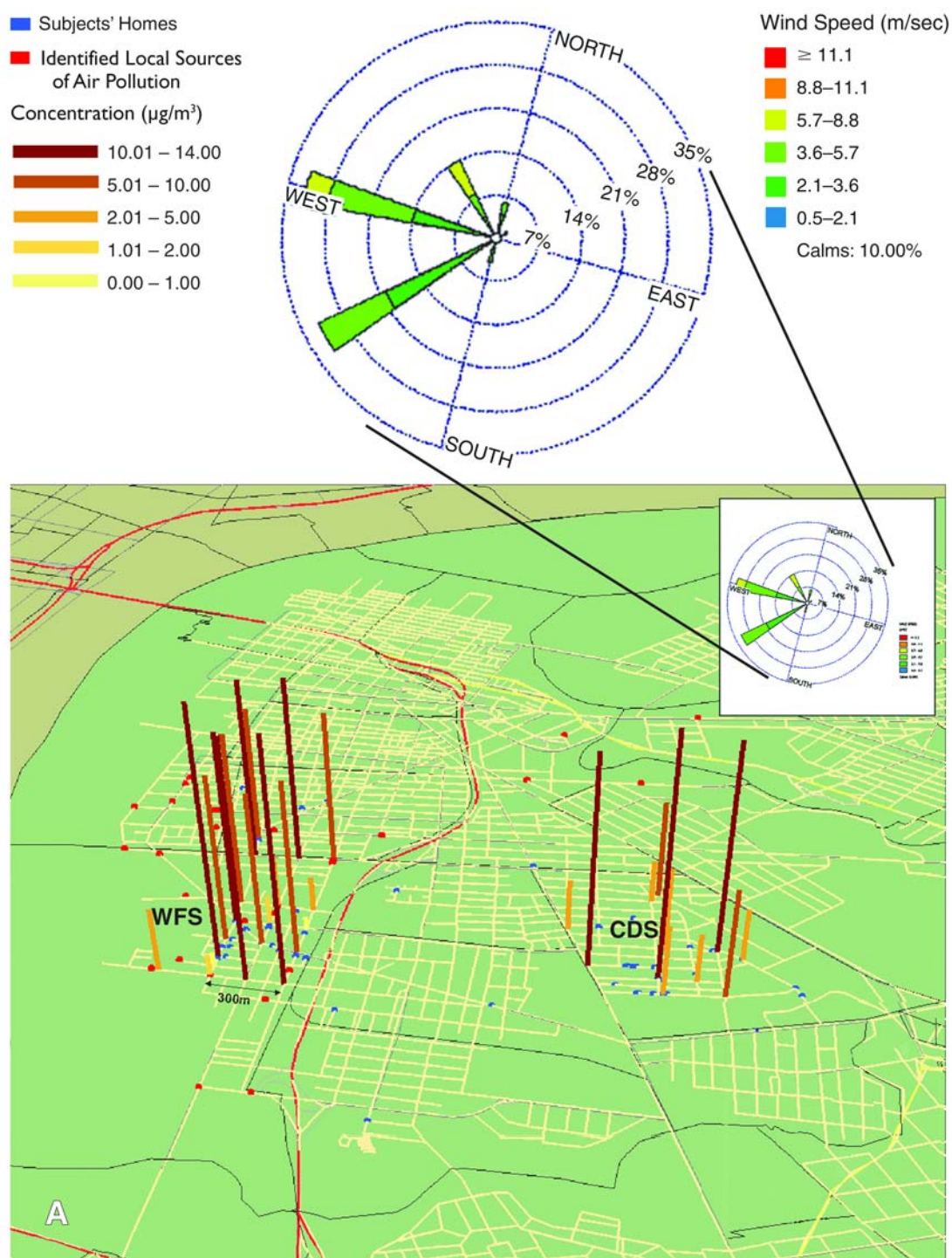


Figure 48. Spatial distribution of acetaldehyde for three saturation sampling campaigns on (A) July 20–22, (B) August 17–18, and (C) December 21–23, 2005. The height of the colored bars is proportional to the concentration they represent. In the wind rose, spoke lengths indicate the frequency of wind from the given directions. (Wind directions shown are the directions from which the wind was blowing.) (Figure continues on next two pages)

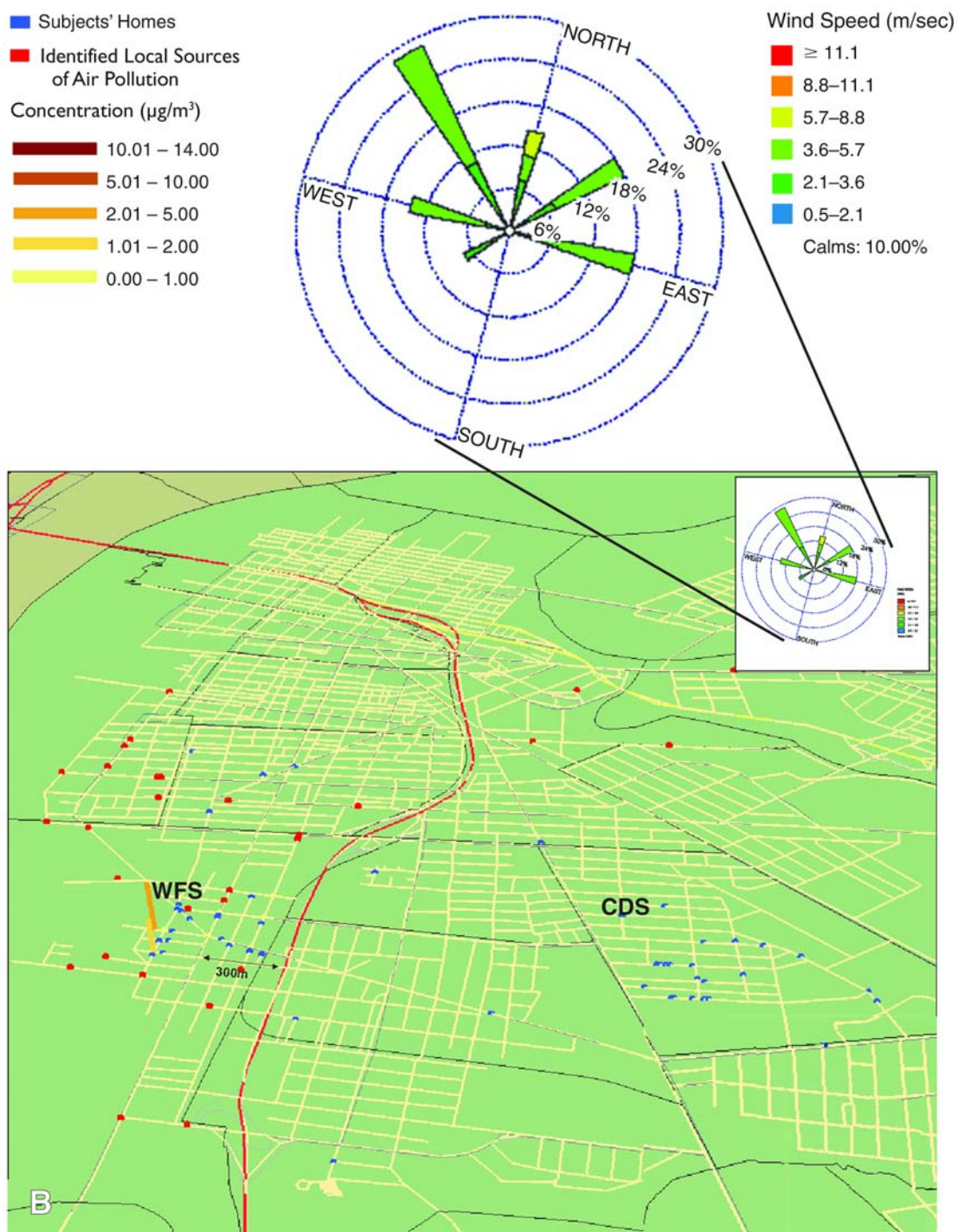


Figure 48 (Continued). (B) August 17–18, 2005. (Figure continues on next page)

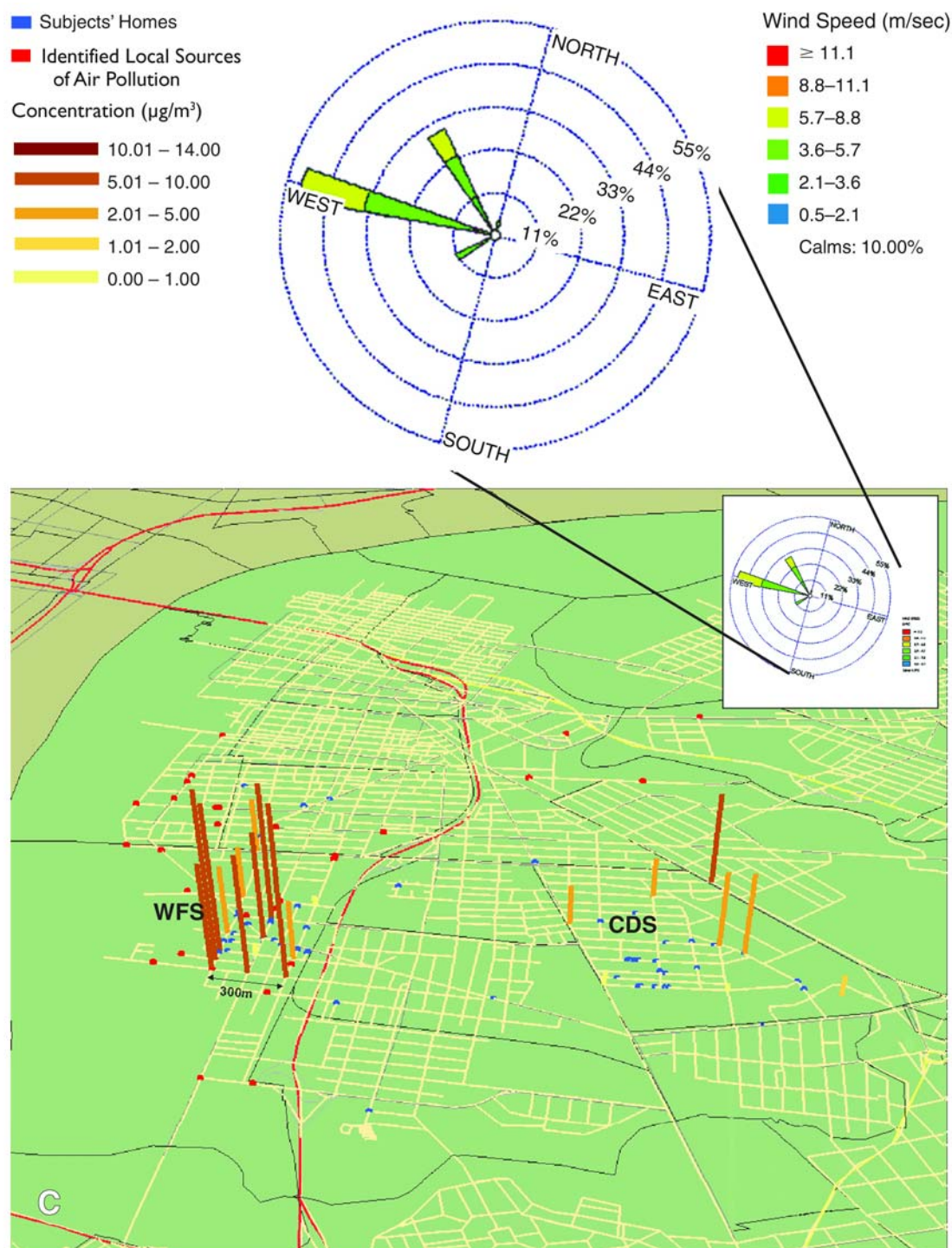


Figure 48 (Continued). (C) December 21–23, 2005.

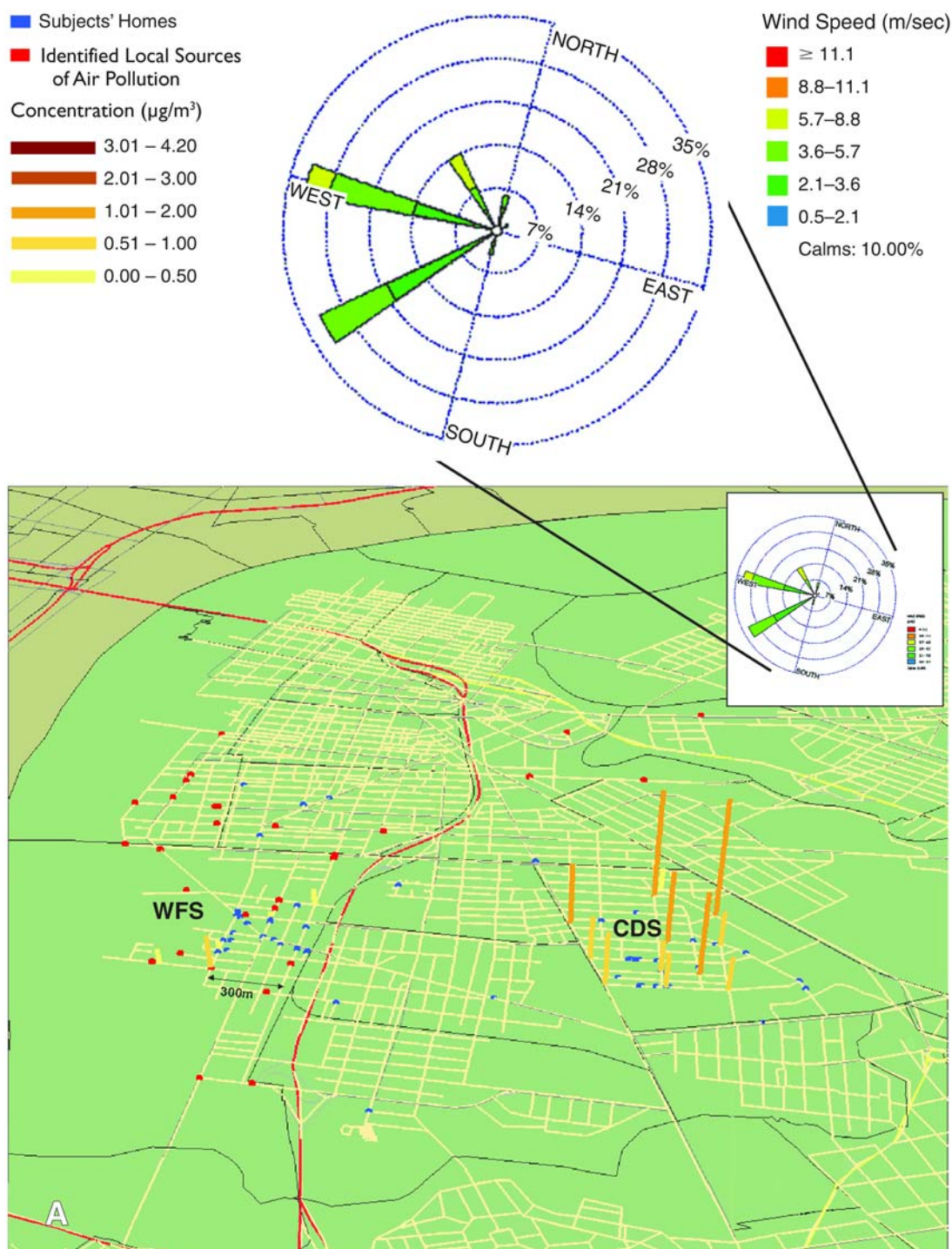


Figure 49. Spatial distribution of acrolein for three saturation sampling campaigns on (A) July 20–22, (B) August 17–18, and (C) December 21–23, 2005. The height of the colored bars is proportional to the concentration they represent. In the wind rose, spoke lengths indicate the frequency of wind from the given directions. (Wind directions shown are the directions from which the wind was blowing.) (Figure continues next two pages)

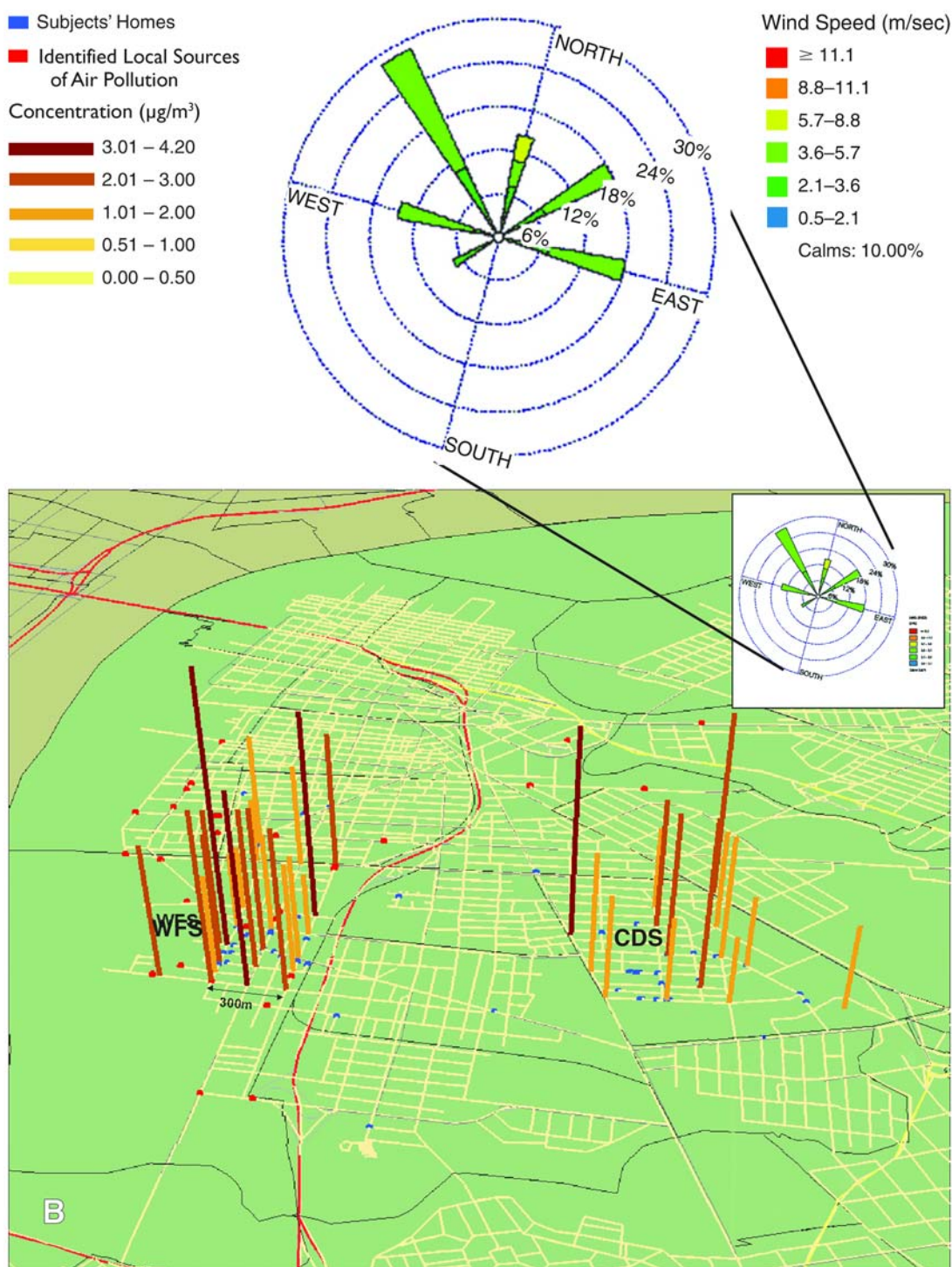


Figure 49 (Continued). (B) August 17–18, 2005. (Figure continues next page)

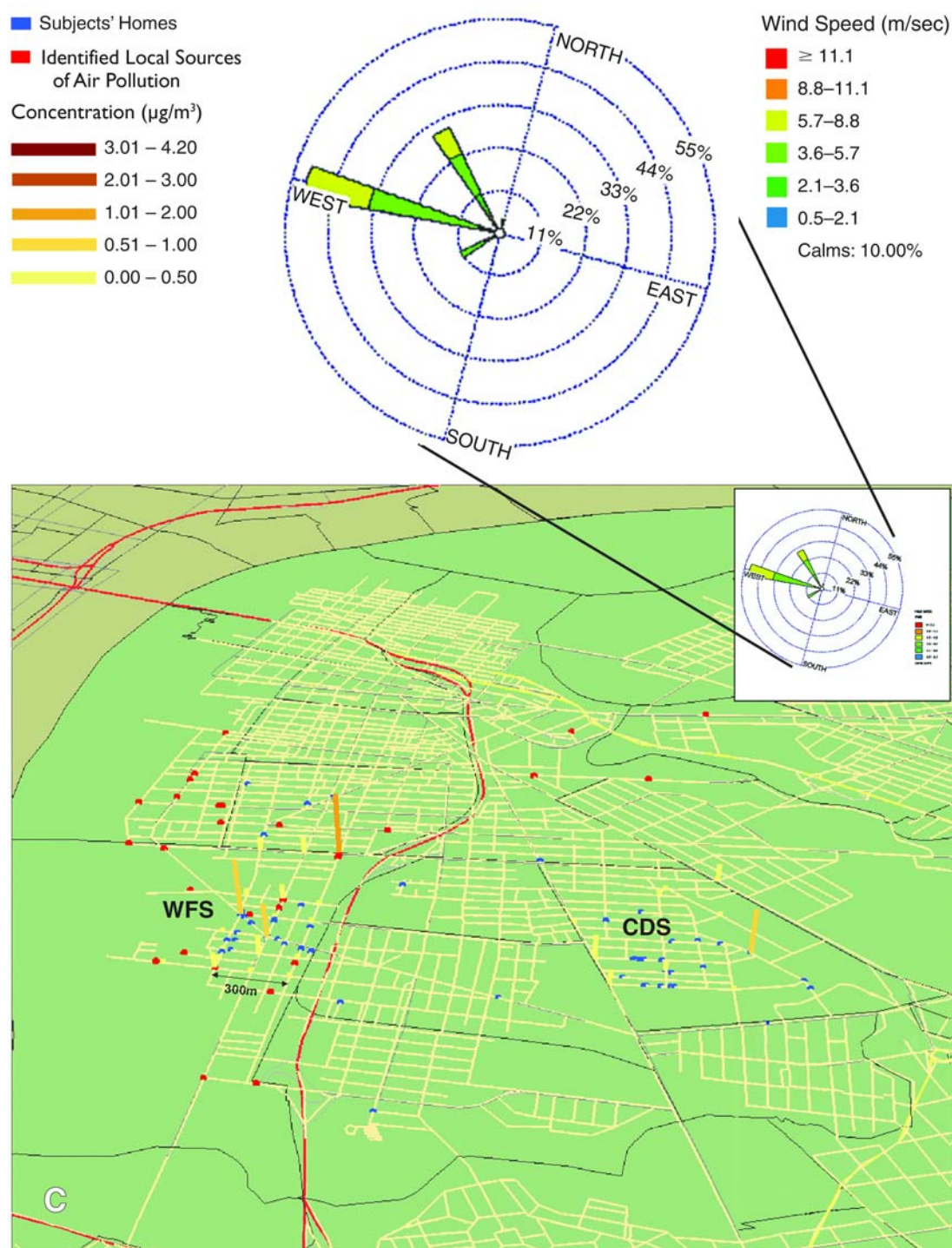


Figure 49 (Continued). (C) December 21–23, 2005.

Table 22. Ratios of Air Toxic Concentrations at the Fixed Monitoring Site to the Means for Waterfront South and Copewood–Davis in Three Sampling Campaigns

	Waterfront South			Copewood–Davis		
	July 20–22, 2005	August 17–18, 2005	December 20– 22, 2005	July 20–22, 2005	August 17–18, 2005	December 20– 22, 2005
VOCs						
MTBE	0.9	0.8	0.7	0.9	0.9	0.7
Chloroform	0.9	0.9	1.0	1.0	0.9	1.0
Carbon tetrachloride	1.0	1.0	1.0	1.0	1.0	1.0
Benzene	1.1	1.1	1.0	0.8	0.8	0.8
Toluene	0.7	0.5	0.6	0.8	0.8	0.7
Ethylbenzene	0.8	0.5	0.6	0.7	0.8	0.7
<i>m</i> - & <i>p</i> -Xylenes	0.8	0.5	0.6	0.7	0.8	0.7
<i>o</i> -Xylene	0.8	0.7	0.7	0.7	0.8	0.7
Aldehydes						
Formaldehyde	1.3	1.0	1.1	1.0	1.3	1.0
Acetaldehyde	1.2	0.3	1.8	0.6	2.4	2.6
Acrolein	0.6	1.1	2.4	0.1	0.6	3.7

not available for the neighborhood. Information on acrolein in particular was very limited because of the lack of measurement methods. We were thus not able to identify the local stationary sources of the two compounds in the neighborhood, and more research will be needed to elucidate them.

Comparison Between Spatial Variation Sampling and Fixed Site Monitoring

In the main exposure study, measurements were obtained from two fixed monitoring sites (W8 and C13), one each in Waterfront South and in Copewood–Davis, in order to represent neighborhood VOC and aldehyde concentrations. To evaluate whether these measurements were representative of the neighborhoods' air quality as a whole, the concentrations measured at the fixed sites during each of the three saturation-sampling campaigns were compared with the average concentrations measured at all the campaign sites in each neighborhood. For MTBE and TEX the concentrations from W8 and C13 were found to be lower than the mean values from the campaign sites as a whole in both neighborhoods, with differences of 10% to 30% for MTBE and 20% to 50% for TEX (Table 22). The benzene concentrations from the fixed sites were similar to the means of the campaign sites as a whole in Waterfront South but were 20% lower in Copewood–Davis (Table 22). These results indicate that our measurements at the fixed monitoring sites might have underestimated TEX and MTBE concentrations in Waterfront South.

No differences were found for chloroform or carbon tetrachloride between measurements from the fixed sites and the means of the campaign sites (Table 22). This is consistent with the view that regional sources are the only influences for these two species.

No significant difference was found for formaldehyde, because mobile sources were the main contributors to formaldehyde in each area. For acetaldehyde and acrolein the differences between concentrations at the fixed site and the means from the campaign sites varied by campaign (Table 22); the reasons for the differences in these results are not clear.

In summary, greater spatial variations in MTBE and TEX were observed in Waterfront South than in Copewood–Davis. On campaign days with lower wind speeds, mean concentrations and spatial variability of benzene, toluene, ethylbenzene, and xylenes were significantly higher ($P < 0.05$) in Waterfront South than in Copewood–Davis. Poor correlations were found between MTBE and TEX. These results indicate significant impacts from local stationary sources on ambient concentrations of these compounds in Waterfront South. Benzene concentrations ($< 3 \mu\text{g}/\text{m}^3$) and spatial variability (%RSD $< 50\%$) were found to be similar in both neighborhoods, and their spatial variability was relatively low. Good correlations ($R > 0.8$) were found between MTBE and benzene in both locations, suggesting that automobile exhaust was the main benzene source. Local stationary sources, however, also contributed to site variability, especially in Waterfront South. Background concentrations ($< 1 \mu\text{g}/\text{m}^3$) and homogenous spatial distributions of chloroform and carbon tetrachloride observed in Waterfront South and

Copewood–Davis indicate the influence of regional sources on these two compounds in both neighborhoods. Formaldehyde and acetaldehyde concentrations were found to be high at all sites in the two neighborhoods; this was consistent with results obtained in the main exposure study. The results indicate the impact of local diesel truck traffic on formaldehyde and acetaldehyde air pollution in the neighborhoods.

MENTOR MODELING APPLICATION

Results of Model Application

The results of our source-to-exposure modeling analysis of ambient and personal data for benzene, toluene, and formaldehyde are presented in two parts, the dispersion model predictions for ambient concentrations and the IBEM–MENTOR predictions for personal exposures.

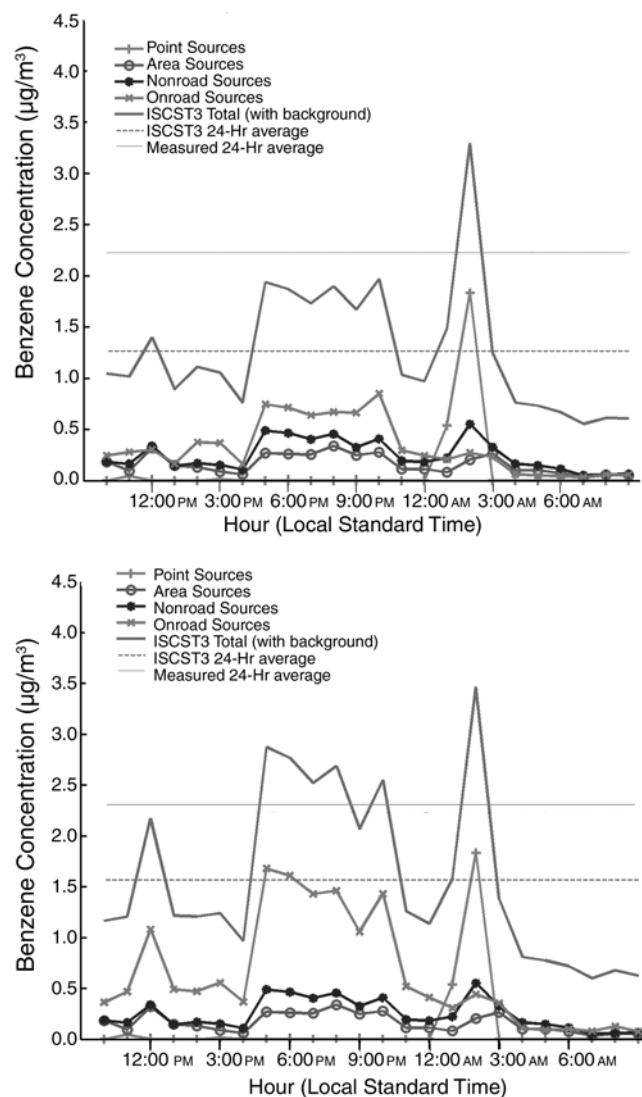


Figure 50. Comparison of time-series profiles for hourly ISCST3 predictions of ambient benzene concentrations using (A) census-tract-based spatial allocation and (B) road-link-based spatial allocation for mobile onroad emission sources. Inputs for point, area, and mobile nonroad sources were unchanged. Sampling date: July 15, 2006. The horizontal gray line indicates the measured 24-hr average. The horizontal dotted line indicates the ISCST3 24-hr average.

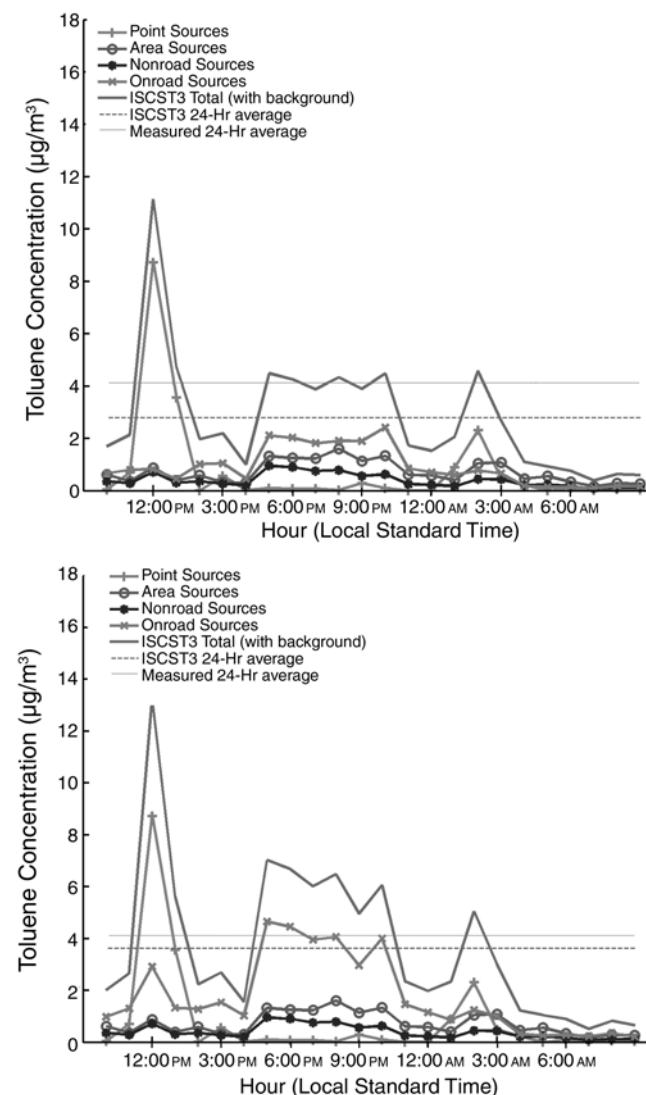


Figure 51. Comparison of time-series profiles for hourly ISCST3 predictions of toluene concentrations using (A) census-tract-based spatial allocation and (B) road-link-based spatial allocation for mobile onroad emission sources. Inputs for point, area, and mobile nonroad sources were unchanged. Sampling date: July 15, 2006.

Ambient Concentrations of Air Toxics Based on the approaches described earlier and in Appendix H, the sensitivity to using two different sets of data for spatially allocated mobile onroad emissions (census-tract-based versus road-link-based) for dispersion modeling was examined, first by comparing ISCST3 predictions for ambient benzene, toluene, and formaldehyde concentrations with the corresponding actual measurements made in the study at the two fixed monitoring sites (Figure 50 and Figure 51). The use of finer spatial allocations of mobile onroad sources to road links increased the predicted hourly ambient concen-

trations of benzene and toluene from onroad sources. The increase was a factor of two during the evening rush hours in Waterfront South compared with the census-tract-based results. Further, the 24-hr averages of the road-link-based results were closer to the actual ambient measurements from Waterfront South than were the census-tract-based results. This was reasonable, because Waterfront South is immediately adjacent to a busy truck road. Comparisons for all the sampling dates at the two fixed monitoring sites in the two neighborhoods are summarized in Figure 52. Because of the proximity of Waterfront South to major roadways, the use of road-link-based spatial allocations for mobile onroad emissions improved the predictions of the ISCST3 model for the neighborhood. However, for Copewood–Davis, which is not immediately adjacent to major roadways, both the census-tract-based and road-link-based approaches led to similar ISCST3 predictions. Because of the improvements the road-link-based approach made to model performance, it was selected for use in our spatial allocation of county-level mobile onroad emissions for all ISCST3 and AERMOD modeling presented in the study. For area and mobile nonroad sources the census-tract-based approach was still used for the fine-scale allocation of county-level emissions.

The ISCST3 predictions of ambient benzene and toluene concentrations were in general agreement with PAMS measurements (mostly within a factor of two) for our 17 matched sampling periods (see examples in Figure 53 and Figure 54). However, a number of unusually high benzene measurements from Copewood–Davis were identified for the period between July 19 and September 11, 2004 (hereinafter called the 2004 summer hot days). These were very different from the corresponding PAMS measurements and had unusually high benzene-to-toluene ratios (Figure 53). Because concurrent ambient benzene measurements had also been collected in Waterfront South over the same time period, we examined these measurements and their benzene-to-toluene ratios and found that they were unusual in similar ways as well. Although these data deserve further investigation, their use in evaluating model performance was not deemed reasonable. For personal and ambient concentrations of benzene, then, the model performance evaluations for both dispersion modeling and MENTOR exposure modeling are presented without the benzene measurements from the 2004 summer hot days.

For toluene the ISCST3 predictions, PAMS measurements, and Copewood–Davis ambient measurements were in general agreement for all 17 matched periods. Exclusion of the 2004 summer hot days was therefore not needed for toluene.

Linear regressions were conducted to assess the agreement between the model predictions and ambient measurements. All ambient measurements collected at the two fixed monitoring sites in the two neighborhoods over the three

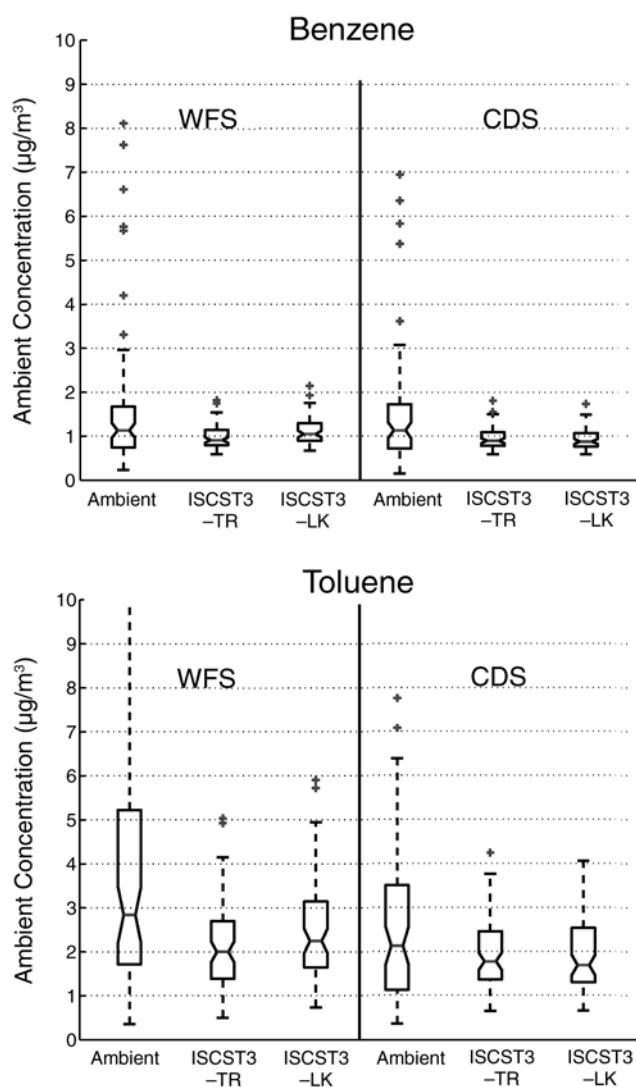


Figure 52. Box plots for ambient concentrations of (top) benzene and (bottom) toluene measured at the two fixed monitoring sites in Waterfront South (WFS) and Copewood–Davis (CDS) and corresponding ISCST3 predictions using data from all sampling dates and census-tract-based (ISCST3-TR) or road-link-based (ISCST3-LK) methods for spatial allocation of mobile onroad sources.

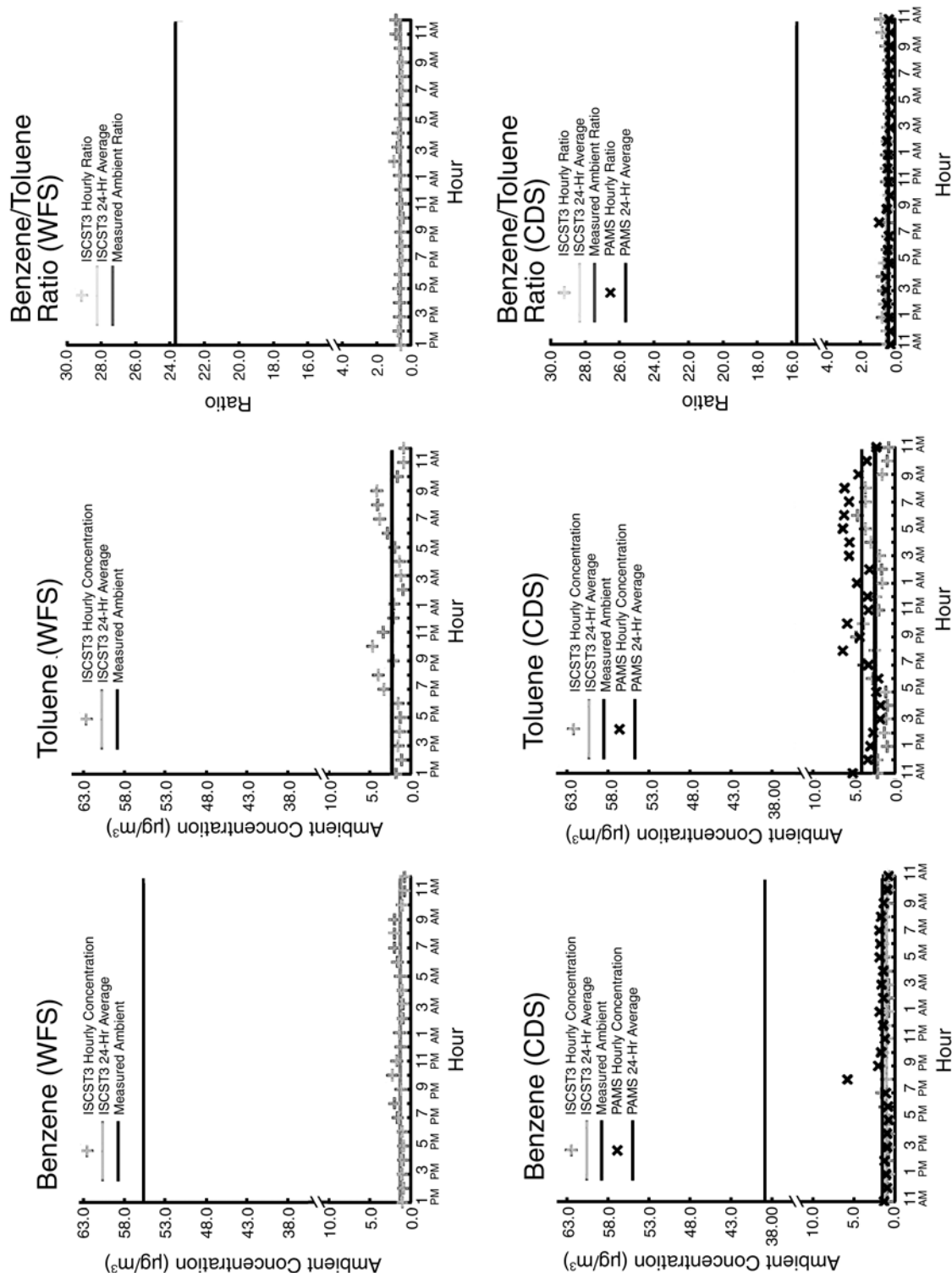


Figure 53. Comparison of ISCST3 predictions for ambient benzene and toluene concentrations with corresponding actual concentrations measured at the fixed monitoring sites in Waterfront South (WFS) and Copewood-Davis (CDS) and at the PAMS site in CDS on July 21, 2004. The sampling date is an example of a day when unusually high benzene measurements made at the fixed sites were not consistent with those made at the PAMS site.

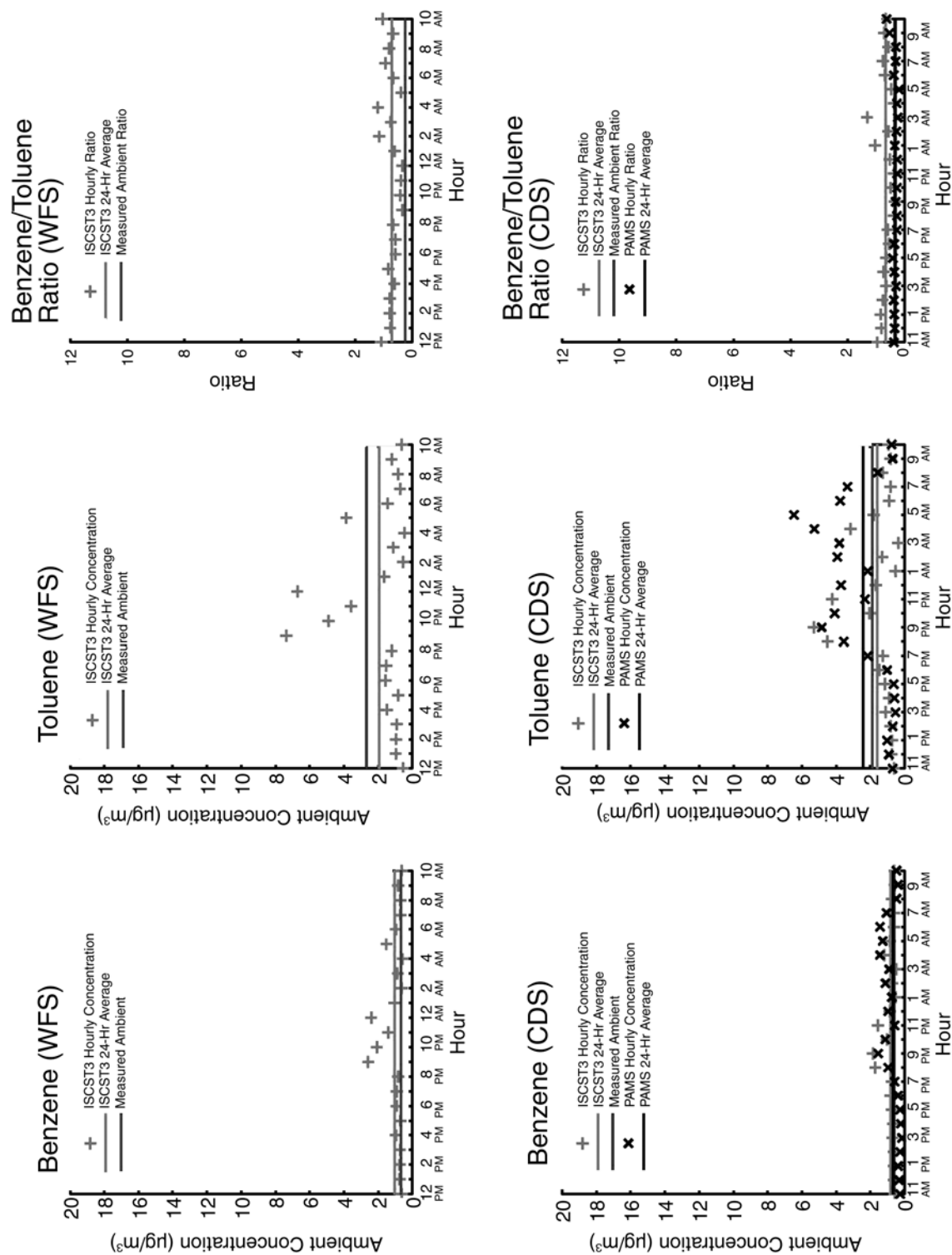


Figure 54. Comparison of ISCST3 predictions for ambient benzene and toluene concentrations with corresponding actual concentrations measured at the fixed monitoring sites in Waterfront South (WFS) and Copewood-Davis (CDS) and at the PAMS site in CDS on July 23, 2005. The sampling date is an example of a day when benzene measurements made at the fixed sites were consistent with those made at the PAMS site.

years of the study were included for toluene and formaldehyde. (For benzene, again, the linear regression was conducted without the data from the 2004 summer hot days.) If the model predictions consistently agree with the measurements, the slope of the regression line will be near 1.

For benzene the slopes of the model predictions regressed against the ambient measurements were 0.92 for ISCST3 and 0.93 for AERMOD. Both of the 95% confidence intervals for the two slopes covered positive values centered on 1 (Table 23), indicating statistical significance for the slopes. For toluene the slopes were 0.94 and 1.0, respectively. The 95% confidence intervals were narrower than those for benzene (Table 23), meaning that the performance of the two dispersion models was better for toluene than for benzene. For formaldehyde the 95% confidence intervals of the slopes covered zero values, indicating that the slopes were not statistically significant.

For both dispersion models the predictions for toluene provided the best performance results among the three air toxics. All of the slopes of the regression lines were less than 1 except that of the AERMOD predictions for toluene (Table 23). For toluene and benzene the model predictions were generally underestimates at high measured concentrations and overestimates at low measured concentrations. For formaldehyde the model predictions were

underestimates across all measurements. For formaldehyde and benzene the variations among the ISCST3 and AERMOD predictions were much smaller than the variations among the ambient measurements ($R^2 = 0.002$ and 0.0015 for formaldehyde and 0.08 and 0.07 for benzene, respectively). This situation improved in the model predictions for toluene, as reflected in the roughly elliptical shapes of the scatter points and the larger R^2 values (0.22 and 0.23). The results of the analyses indicate that both ISCST3 and AERMOD predictions for benzene and formaldehyde did not capture the variability of the actual ambient measurements, because of spatial (the locations of the fixed monitoring sites) and temporal (day of the week) differences. However, the model predictions for toluene showed better agreement with the ambient measurements. Detailed comparisons of model predictions and ambient measurements by location, season, and day of the week are shown in Appendix I.

Central Sites Versus Subjects' Residences As noted earlier, ambient measurements were taken only at the study's two fixed monitoring sites, on the assumption that these measurements would be representative of ambient air toxic concentrations as a whole for the two neighborhoods (and hence for the subjects' residences). Comparisons of model predictions for ambient concentrations with actual ambient measurements made at the fixed sites and the subjects' residences helped us examine and substantiate this assumption. The slopes of the ISCST3 predictions regressed against the fixed sites and subjects' residences were very close to 1 (0.98, 0.98, and 0.94, respectively), with R^2 's of 0.92, 0.92, and 0.94 for benzene, toluene, and formaldehyde, respectively. A similar pattern was observed for the AERMOD predictions (with slopes of 0.89, 0.88, and 0.94 and R^2 's of 0.72, 0.79, and 0.94 for benzene, toluene, and formaldehyde, respectively). These findings show that ambient concentrations measured at the fixed sites in Waterfront South and Copewood–Davis were generally representative of ambient concentrations in the two neighborhoods. However, as shown in our saturation-sampling campaigns, higher values will be observed in Waterfront South on specific days near specific sources and locations.

Analysis of Ambient Source Contributions to Predictions of Ambient Concentrations To assess the impact of local ambient sources of benzene, toluene, and formaldehyde on ambient concentrations in the two neighborhoods, we apportioned the contributions from ambient sources in five source categories — background, point, area, mobile onroad, and mobile nonroad — to the 24-hr averages of the ISCST3 predictions for the three air toxics' ambient concentrations. Figure 55 shows an example of this analysis

Table 23. Linear Regression Coefficients and 95% Confidence Intervals (CI) for Measured Ambient Benzene, Toluene, and Formaldehyde Concentrations Regressed Against ISCST3 and AERMOD Outputs

Chemical / Model	Regression Coefficients	(95% CI)
Benzene (Without 2004 Summer Hot Days)		
ISCST3	Slope: 0.92	(0.40 to 1.45)
	Intercept: 0.08	(0.02 to 0.15)
AERMOD	Slope: 0.93	(0.40 to 1.49)
	Intercept: 0.10	(0.04 to 0.17)
Toluene		
ISCST3	Slope: 0.94	(0.68 to 1.21)
	Intercept: -0.01	(-0.11 to 0.09)
AERMOD	Slope: 1.00	(0.72 to 1.28)
	Intercept: -0.01	(-0.1 to 0.1)
Formaldehyde		
ISCST3	Slope: 0.27	(-0.73, 1.27)
	Intercept: 1.07	(0.89 to 1.26)
AERMOD	Slope: 0.28	(-0.83 to 1.38)
	Intercept: 1.07	(0.88 to 1.26)

for benzene and toluene for the sampling date July 15, 2006. Figure 56 shows the analysis for formaldehyde for July 9, 2005. These results were typical of all the analyses. The source contribution profiles shown in the bar charts in Figures 55 and 56 provide quantitative estimates of the contributions from the five source categories to the predicted ambient concentrations.

For benzene our ISCST3 predictions captured about 70% and 60% of the ambient values measured at the Waterfront South and Copewood–Davis fixed monitoring sites, respectively (Figure 55). For background concentrations, modeled contributions constituted about 20% and 30% of the model's predictions for the two neighborhoods, respectively. Because benzene is a relatively inert chemical, characterizing the contribution from long-range transport (i.e., the background concentration) is essential. The principal local contributors of ambient benzene were mobile onroad sources; these appeared to dominate both in Waterfront South and in Copewood–Davis. In addition, the estimated contribution from mobile onroad sources in Waterfront South was higher than in Copewood–Davis, because Waterfront South is near major roadways. The estimated contributions of benzene from stationary (point and area) sources were relatively small.

For toluene there was no background contribution, because it is a more reactive chemical than benzene. Our ISCST3 predictions captured about 80% and 60% of the ambient values measured at the Waterfront South and Copewood–Davis fixed monitoring sites, respectively (Figure 55). Both mobile and stationary (point and area) sources were relatively small.

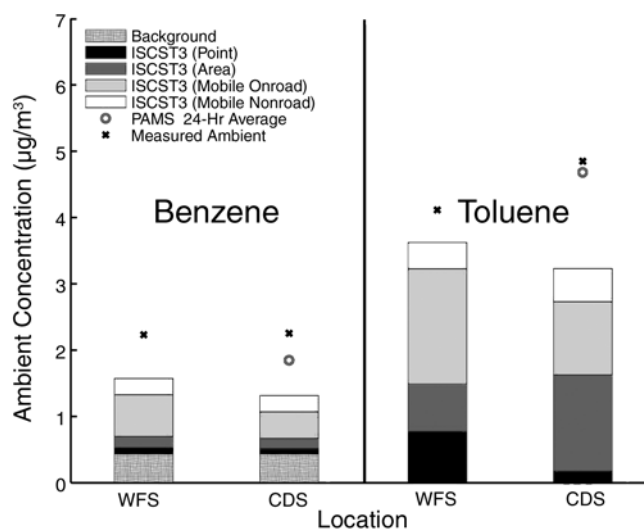


Figure 55. Comparison of ISCST3 predictions for benzene and toluene source contributions from the five source types with actual ambient concentrations measured at the two fixed monitoring sites in Waterfront South (WFS) and Copewood–Davis (CDS) and with PAMS 24-hr averages measured in CDS on July 15, 2006.

sources contributed equally to the estimated ambient concentrations in the two neighborhoods, confirming what had been suggested by the results from both our general study and saturation-sampling campaigns. The higher estimated contribution from mobile onroad sources in Waterfront South again reflected the impact of the neighborhood's proximity to major roadways. The estimated contribution from point sources was much higher in Waterfront South than in Copewood–Davis, because the Waterfront South fixed monitoring site is closer to industrial facilities than is the Copewood–Davis site.

For formaldehyde our ISCST3 predictions significantly underestimated the ambient values measured at the Waterfront South and Copewood–Davis fixed monitoring sites (Figure 56). For background concentrations, modeled contributions constituted about 50% and 70% of the model's predictions for the two neighborhoods, respectively. The other major modeled contributors to the ISCST3 predictions were mobile nonroad sources in both neighborhoods. The model's significant underestimates for formaldehyde can be attributed to the lack of estimates for secondary formaldehyde formation caused by photochemical production and to significant underestimates for mobile source emissions in the region. In a regional-scale modeling study of central Philadelphia, Georgopoulos and colleagues (2005) used Community Multiscale Air Quality modeling to estimate secondary formaldehyde formation at 5 ppb (about 6.13 µg/m³) to 10 ppb (about 12.3 µg/m³) in general for summer 2001. This estimate can make up roughly half of the discrepancy shown in Figure 56

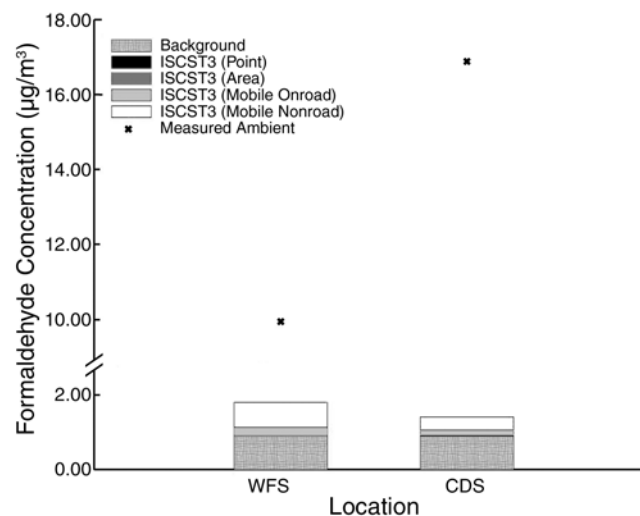


Figure 56. Comparison of ISCST3 predictions for formaldehyde source contributions from the five source types with actual ambient 24-hr averages measured in Waterfront South (WFS) and Copewood–Davis (CDS) on July 9, 2005.

between the ISCST3 predictions and actual ambient measurements. The rest, then, would come from local truck traffic and idling, as suggested, again, by the results from our main study and saturation-sampling campaigns.

To judge from these outcomes, future efforts to improve emissions inventories in the Camden area should focus on mobile sources for benzene, both mobile and stationary sources for toluene, and mobile nonroad and onroad sources or activities related to truck traffic for formaldehyde.

Personal Concentrations of Air Toxics Our ambient measurements and ISCST3 and AERMOD predictions were used as separate ambient-data input options in the microenvironmental and personal exposure modules of the MENTOR system. These data were used to estimate the personal exposure concentrations resulting from ambient sources for each subject. Because the actual personal measurements reflected contributions from both ambient and indoor sources, comparison of the model predictions with the actual measurements would reflect the impact of the ambient sources on the personal exposures. However, our dispersion modeling had significantly underestimated ambient formaldehyde concentrations, which would indicate the underestimation of local outdoor emissions based only on county-level NEI emissions data. The predicted formaldehyde personal concentrations would provide estimates of personal exposures associated with local ambient concentrations (as measured at the fixed monitoring sites) and demonstrate the underestimated exposures based on ambient concentrations estimated from county-level NEI emissions data.

Scatter plots of personal measurements versus model predictions (with linear regression statistics) are shown in Figures 57, 58, and 59 for benzene, toluene, and formaldehyde, respectively.

For benzene (Figures 57a, b, and c) the MENTOR predictions based on ambient measurements performed much better (slope = 0.51, $R^2 = 0.27$) than those based on dispersion modeling (slopes = 0.16 and 0.20 and $R^2 = 0.0028$ and 0.0040, respectively, for ISCST3 and AERMOD). To understand the underlying reason for this, linear regression was conducted for the pairs of personal versus ambient measurements (Figure 57d). The same linear regression statistics (slope = 0.51, $R^2 = 0.27$) were found as for the personal measurements regressed against the MENTOR predictions based on ambient measurements. Model predictions for personal exposures were calculated by adjusting the ambient measurements for linear microenvironmental factors and time spent in the various microenvironments recorded in the subjects' time-activity diaries. This means that there was a linear relationship between the model predictions of personal exposures and the ambient measurements — and

therefore that the correlation between personal measurements and model predictions is attributable to the correlation between the personal and ambient measurements.

For toluene (Figures 58a, b, and c) the MENTOR predictions based on ambient measurements improved the models' performance only marginally (slope = 0.35, $R^2 = 0.10$) compared with the predictions based on dispersion modeling (slopes = 0.47 and 0.46 and $R^2 = 0.05$ and 0.05, respectively, for ISCST3 and AERMOD). This was caused (1) by a weaker association (Figure 58d) between personal and ambient measurements ($R^2 = 0.10$) for toluene than for benzene ($R^2 = 0.42$) and (2) by better model performance when using the dispersion model predictions of ambient concentrations for toluene than for benzene, as reported above.

For formaldehyde (Figures 59a, b, and c) the MENTOR predictions based on ambient measurements improved the models' performance significantly (slope = 0.57, $R^2 = 0.23$) compared with the predictions based on dispersion modeling (slopes = 0.44 and 0.50 and $R^2 = 0.004$ and 0.01, respectively, for ISCST3 and AERMOD). Because the dispersion model predictions had significantly underestimated the ambient concentrations, the MENTOR personal exposure predictions based on these inputs underestimated the personal concentrations in turn. As shown in Figures 59a and b, very narrow vertical bands of scatter points were revealed, indicating that the model predictions of personal exposures could not capture the variation in personal measurements. Significantly better model performance was obtained using the actual ambient measurements, because of their basic correlation with personal measurements (Figure 59c and d). These results suggest that measurements of air toxics at well-characterized fixed monitoring sites are necessary for areas with concentrated sources in order to make accurate estimates of personal exposures in the community.

Comparisons Between Waterfront South and Copewood-Davis To evaluate the model performance of our IBEM-MENTOR personal exposure predictions for the two neighborhoods, the cumulative distribution functions (CDFs) of the predictions, actual personal measurements, and corresponding model-to-measurement ratios were plotted for benzene, toluene, and formaldehyde (Figures 60, 61, and 62, respectively).

For benzene, personal measurements for Copewood-Davis were generally higher than those for Waterfront South (Figure 60). Below the 50th percentile, the IBEM-MENTOR predictions based on all three ambient input values (i.e., the ISCST3 and AERMOD estimates and the actual ambient measurements) were lower than the personal measurements, but there were no significant differences among the predictions. Above the 50th percentile,

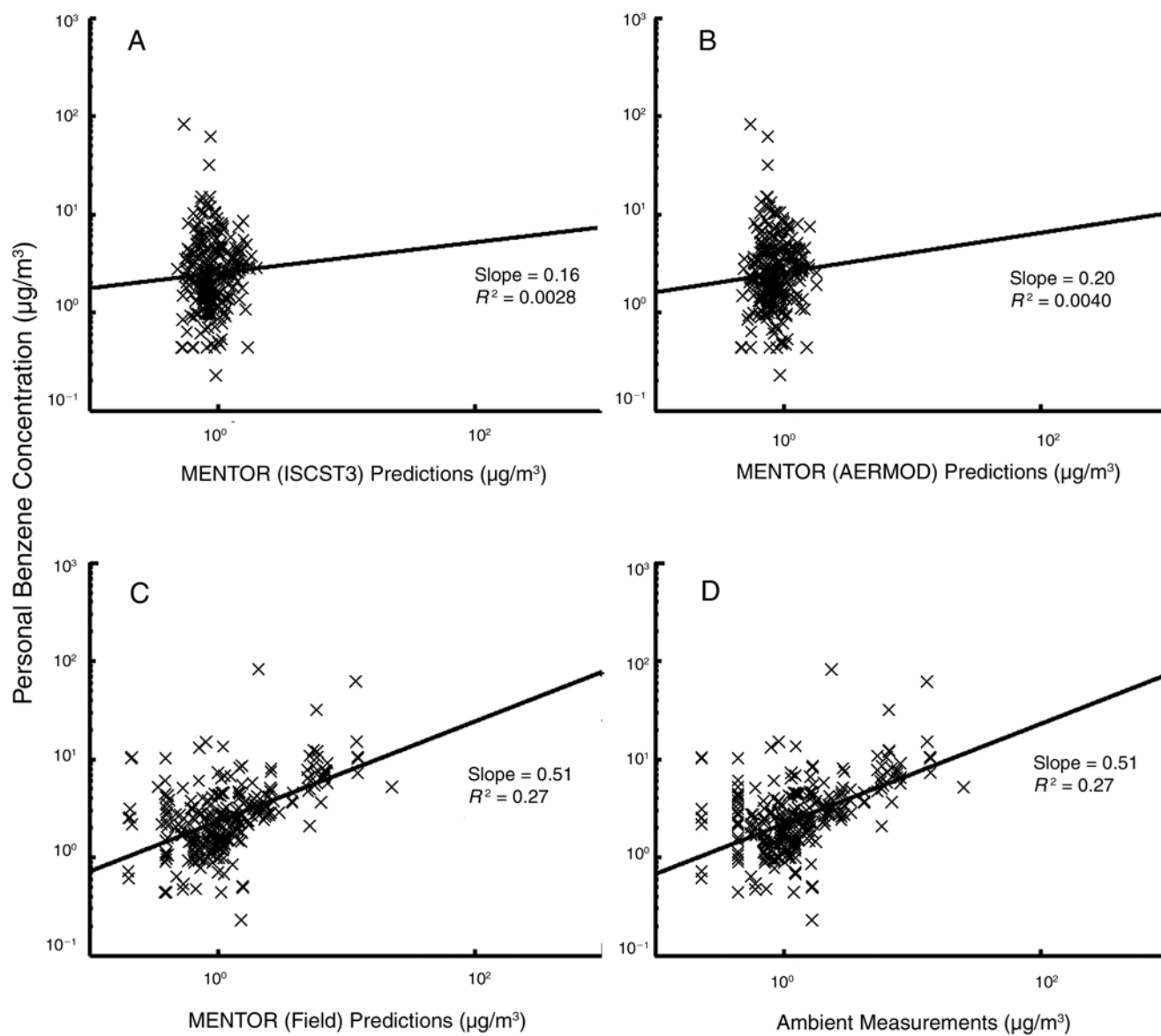


Figure 57. Scatter plots (and linear regression statistics) for personal benzene measurements versus MENTOR predictions for personal exposures caused by ambient sources — using (A) ISCST3 ambient estimates, (B) AERMOD ambient estimates, and (C) actual ambient measurements as inputs — and (D) actual ambient measurements.

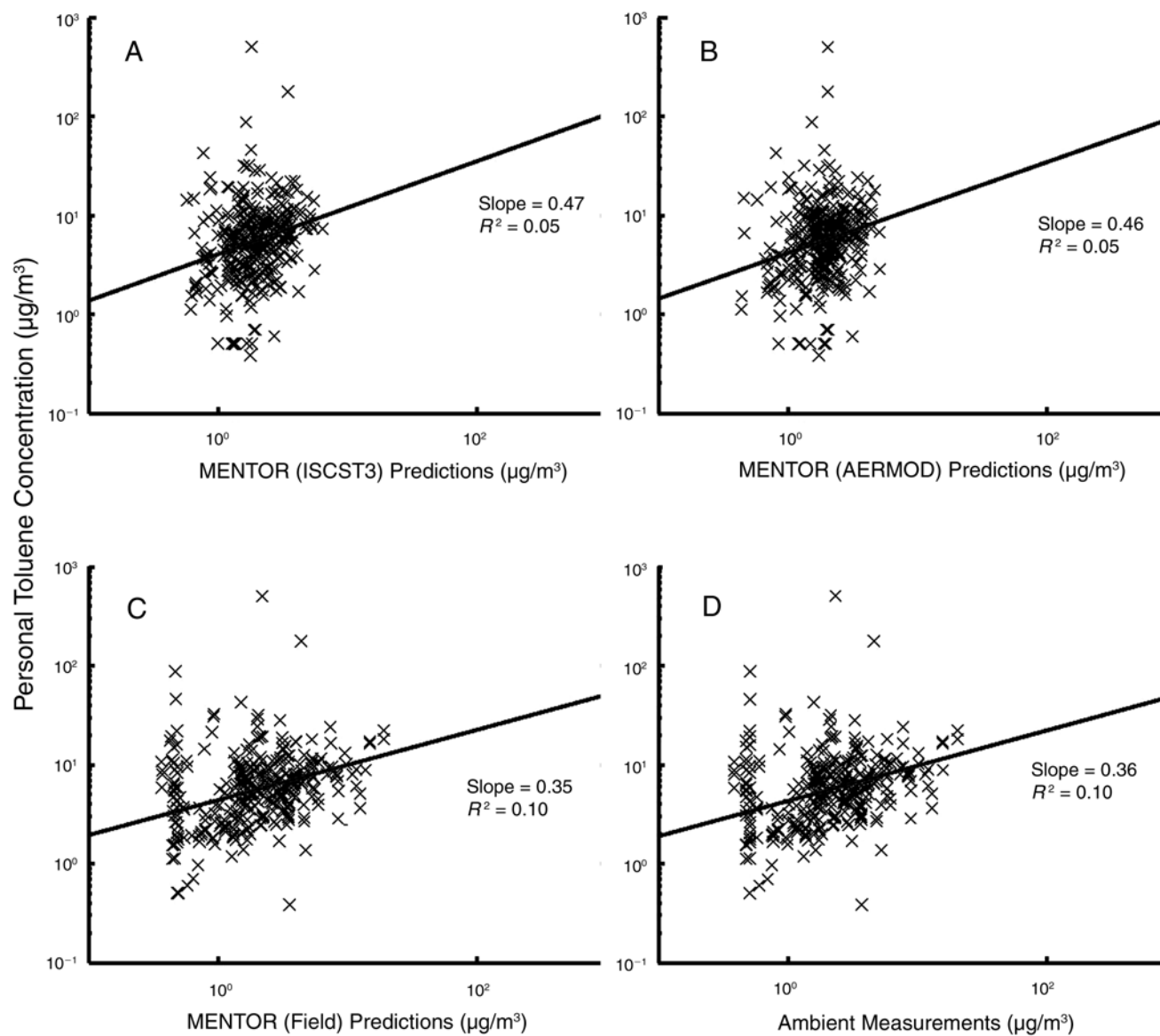


Figure 58. Scatter plots (and linear regression statistics) for personal toluene measurements versus MENTOR predictions for personal exposures caused by ambient sources — using (A) ISCST3 ambient estimates, (B) AERMOD ambient estimates, and (C) actual ambient measurements as inputs — and (D) actual ambient measurements.

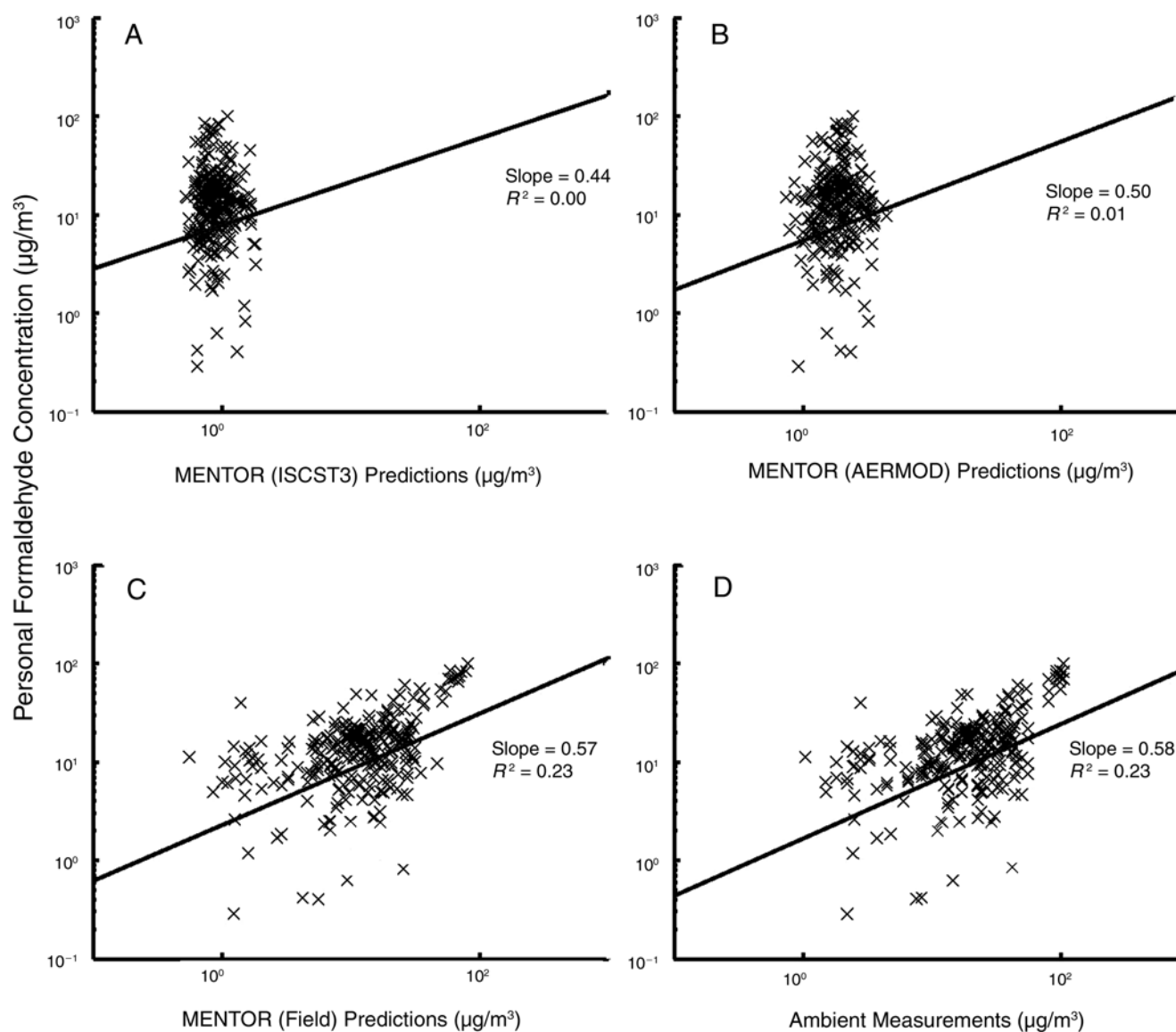


Figure 59. Scatter plots (and linear regression statistics) for personal formaldehyde measurements versus MENTOR predictions for personal exposures caused by ambient sources — using (A) ISCST3 ambient estimates, (B) AERMOD ambient estimates, and (C) actual ambient measurements as inputs — and (D) actual ambient measurements.

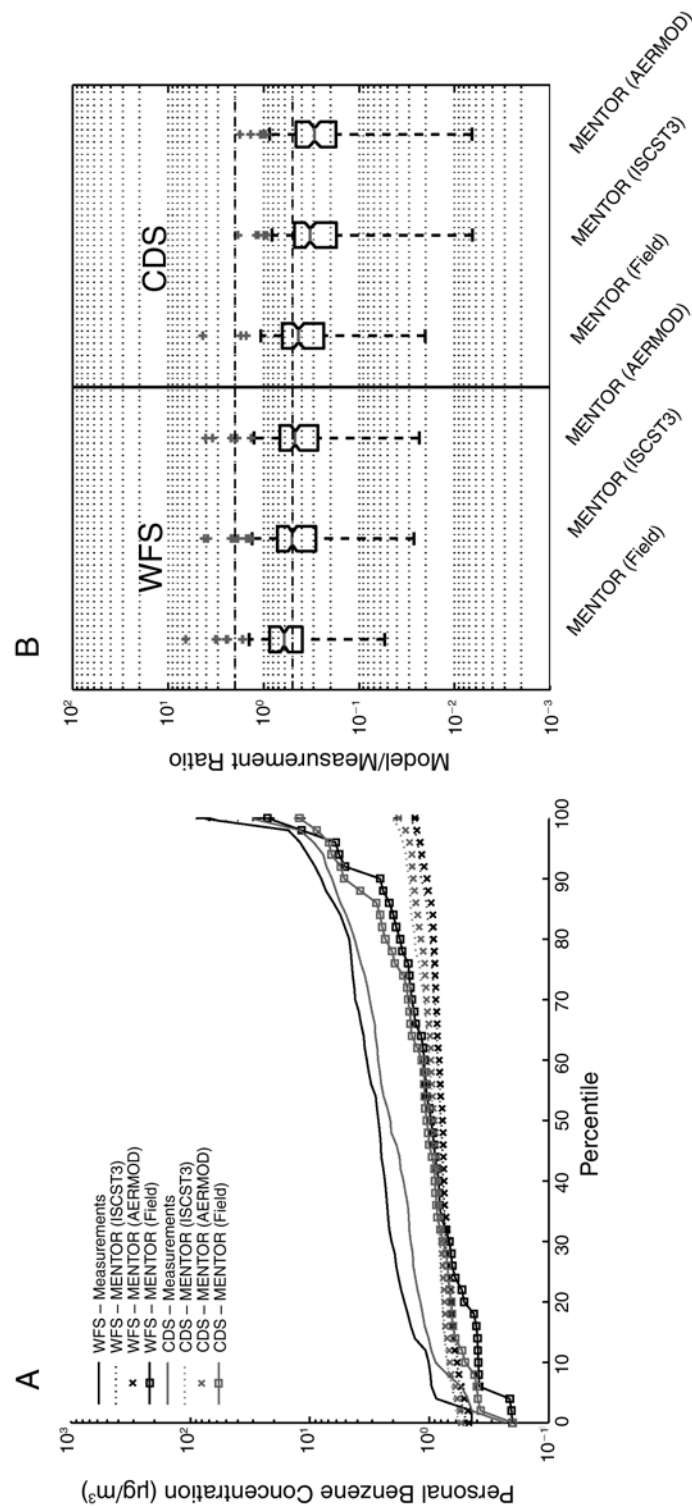


Figure 60. Comparison of cumulative distribution functions for personal benzene measurements and corresponding MENTOR predictions — using ISCST3 estimates, AERMOD estimates, and actual ambient measurements as inputs — for Waterfront South (WFS) and Copewood-Davis (CDS) on all sampling dates except the 2004 summer hot days (A). Box plots of the corresponding model-to-measurement ratios. For ratios that fall between the horizontal dashed lines, the model result agreed with the measurement result to within a factor of 2 (B).

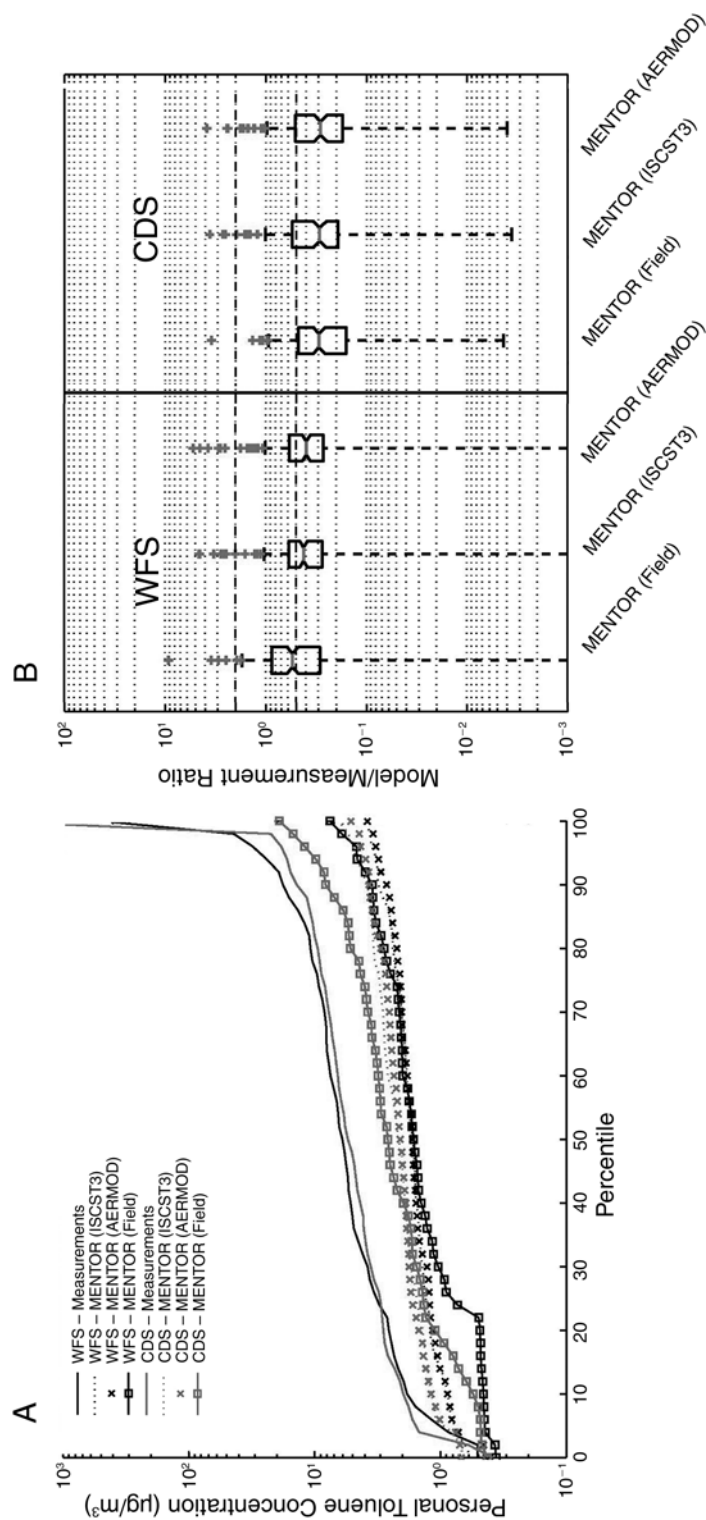


Figure 61. Comparison of cumulative distribution functions for personal toluene measurements and corresponding MENTOR predictions — using ISCST3 estimates, AERMOD estimates, and actual ambient measurements as inputs — for Waterfront South (WFS) and Copewood–Davis (CDS) on all sampling dates except the 2004 summer hot days (A). Box plots of the corresponding model-to-measurement ratios. For ratios that fall between the horizontal dashed lines, the model result agreed with the measurement result to within a factor of 2 (B).

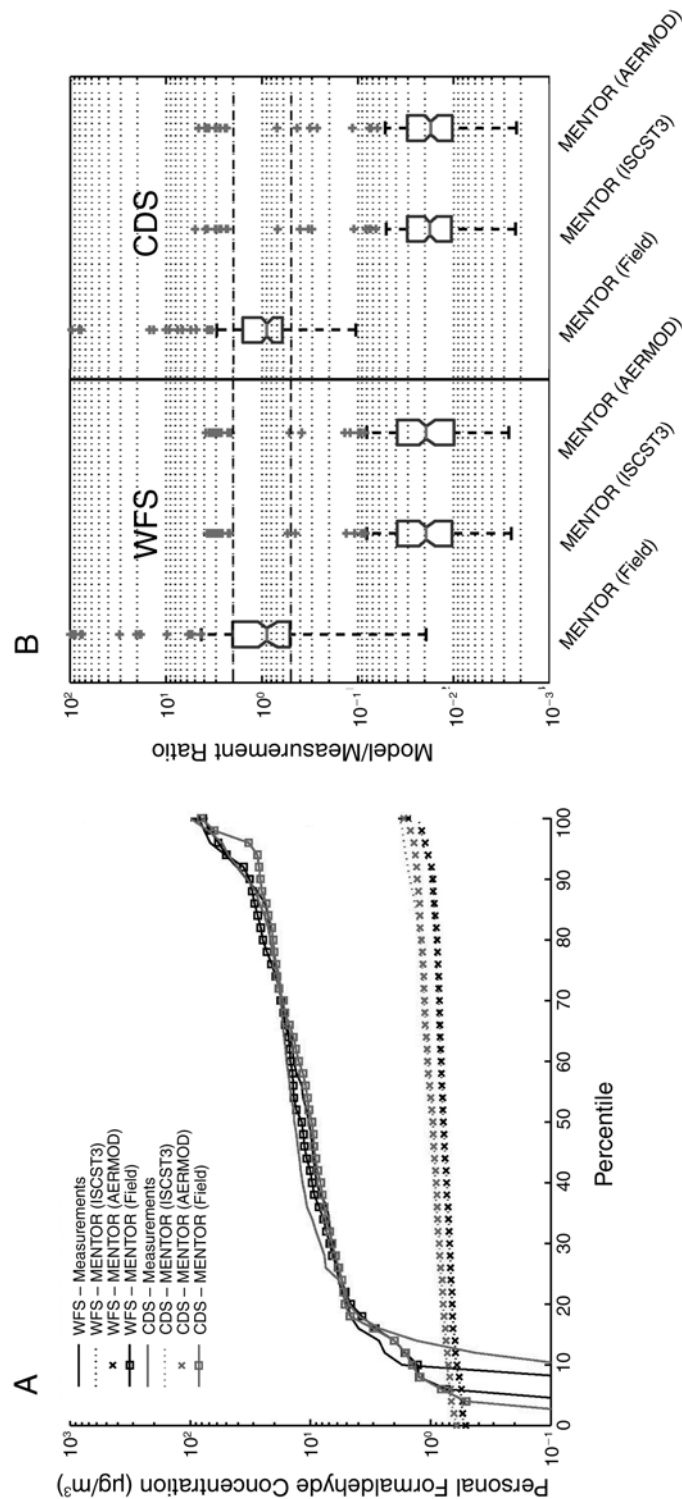


Figure 62. Comparison of cumulative distribution functions for personal formaldehyde measurements and corresponding MENTOR predictions — using ISCST3 estimates, AERMOD estimates, and actual ambient measurements as inputs — for Waterfront South (WFS) and Copewood-Davis (CDS) on all sampling dates except the 2004 summer hot days (A). Box plots of the corresponding model-to-measurement ratios. For ratios that fall between the horizontal dashed lines, the model result agreed with the measurement result to within a factor of 2 (B).

however, the IBEM–MENTOR predictions based on the ambient measurements began to deviate from the other predictions and then caught up with the personal measurements above about the 75th percentile. Here, they showed the differences caused by location of residence. Because the personal exposure predictions only accounted for contributions from ambient sources, the differences between the predictions and the measurements below the 75 percentile could have been caused by contributions from indoor sources. Still, the high-end personal exposures to benzene were dominated by contributions from ambient sources, to judge from the smaller differences seen between the personal measurements and the IBEM–MENTOR predictions derived from the ambient measurements. The model-to-measurement ratios in the box plots represent the estimated percentage of total personal exposures that resulted from ambient sources. The box plots show that the percentage contributions to personal exposures from ambient sources were generally higher in Waterfront South than in Copewood–Davis. Further, the predictions based on the ambient measurements showed higher percentage contributions to personal exposures resulting from ambient sources than did the predictions based on dispersion model estimates of ambient concentrations. This result was caused mainly by the correlation between personal and ambient measurements for benzene (Figure 57), which was stronger than those for toluene and formaldehyde. When the ambient measurements were used as inputs to IBEM–MENTOR, the median percentage contributions from local ambient sources to personal benzene exposures were estimated as being about 60% for Waterfront South and 40% for Copewood–Davis.

For toluene the personal measurements stratified by location showed only minor differences (Figure 61). The IBEM–MENTOR predictions of personal exposures based on all three ambient inputs, however, were underestimated across the entire distribution of the personal measurements. The predictions based only on ambient measurements showed differences by location: those for Waterfront South were higher than those for Copewood–Davis. This was caused primarily by the higher ambient values in Waterfront South. The influence of contributions from indoor or occupational sources to personal toluene exposures was significant for the entire distribution, as shown by the differences between the measurements and the predictions based only on contributions from ambient sources (Figure 61). However, the predictions based on the ambient measurements for Waterfront South subjects were closer to the Waterfront South personal measurements than was the case for Copewood–Davis subjects. This result indicates that the Waterfront South subjects had higher percentages of personal toluene exposures from local ambient sources (as reported earlier, in the section on VOC

measurements). The same result was also found in the box plots of model-to-measurement ratios, where the percentage contributions to personal exposures from ambient sources were generally higher in Waterfront South than in Copewood–Davis; this result was consistent as well with the findings of our saturation-sampling campaigns.

For formaldehyde the personal measurements stratified by location showed only minor differences (Figure 62). The IBEM–MENTOR predictions based on the ambient measurements were generally consistent with the personal measurements (Figure 62). This result clearly demonstrated that personal exposures to formaldehyde in both Waterfront South and Copewood–Davis were dominated by contributions from local ambient emission sources. The box plots of model-to-measurement ratios in Figure 62 show the differences in percentage contributions to personal exposures caused by local ambient concentrations (measured at the fixed monitoring sites) and the predicted ambient concentrations of the dispersion modeling results. The percentage contributions from local ambient concentrations were about one order of magnitude higher than those from the modeling (which had been derived from county-level NEI emissions data), indicating the major impact of local sources on personal formaldehyde exposures.

Comparisons Between Summer and Winter CDF plots and box plots of model-to-measurement ratios stratified by season are shown in Figures 63, 64, and 65 for benzene, toluene, and formaldehyde, respectively. For benzene, personal measurements were higher in summer than in winter; IBEM–MENTOR predictions based on the ambient measurements showed the same trend (Figure 63). Above the 75th percentile, the IBEM–MENTOR predictions for summer were closer to the personal measurements; the predictions for winter were consistently lower than the personal measurements across the entire distribution. These results substantiate the fact that, above the 75% percentile, personal exposures in summer were dominated by contributions from ambient sources. In winter the contributions of indoor sources to personal exposures were larger for the entire study population because the subjects spent more time indoors. For toluene no seasonal differences were observed for the personal measurements; the IBEM–MENTOR predictions showed the same trend (Figure 64). However, the predictions were consistently lower than the personal measurements across the entire distribution because of the influence of indoor sources or occupational sources. For formaldehyde, personal measurements in summer were consistently higher than those in winter (Figure 65).

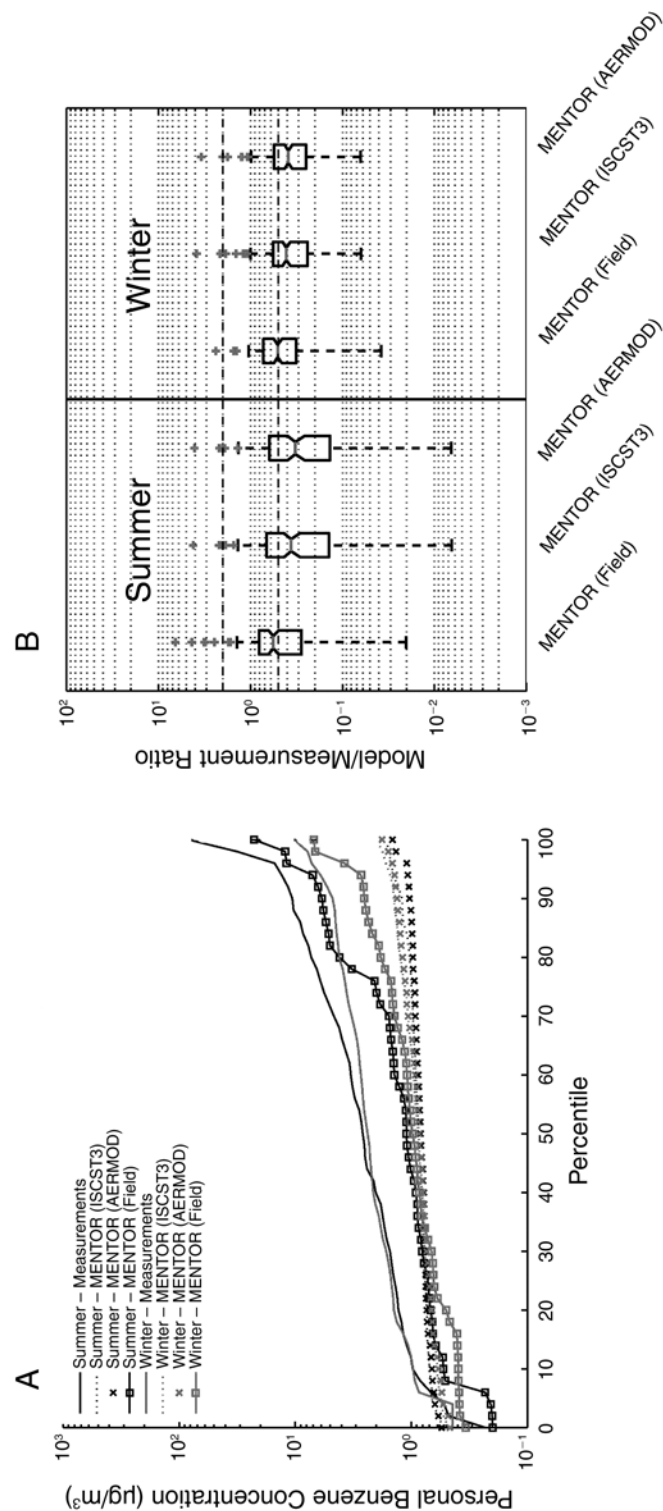


Figure 63. Comparison of cumulative distribution functions for personal benzene measurements and corresponding MENTOR predictions — using ISCST3 estimates, AERMOD estimates, and actual ambient measurements as inputs — for summer and winter on all sampling dates except the 2004 summer hot days (A). Box plots of the corresponding model-to-measurement ratios. For ratios that fall between the horizontal dashed lines, the model result agreed with the measurement result to within a factor of 2 (B).

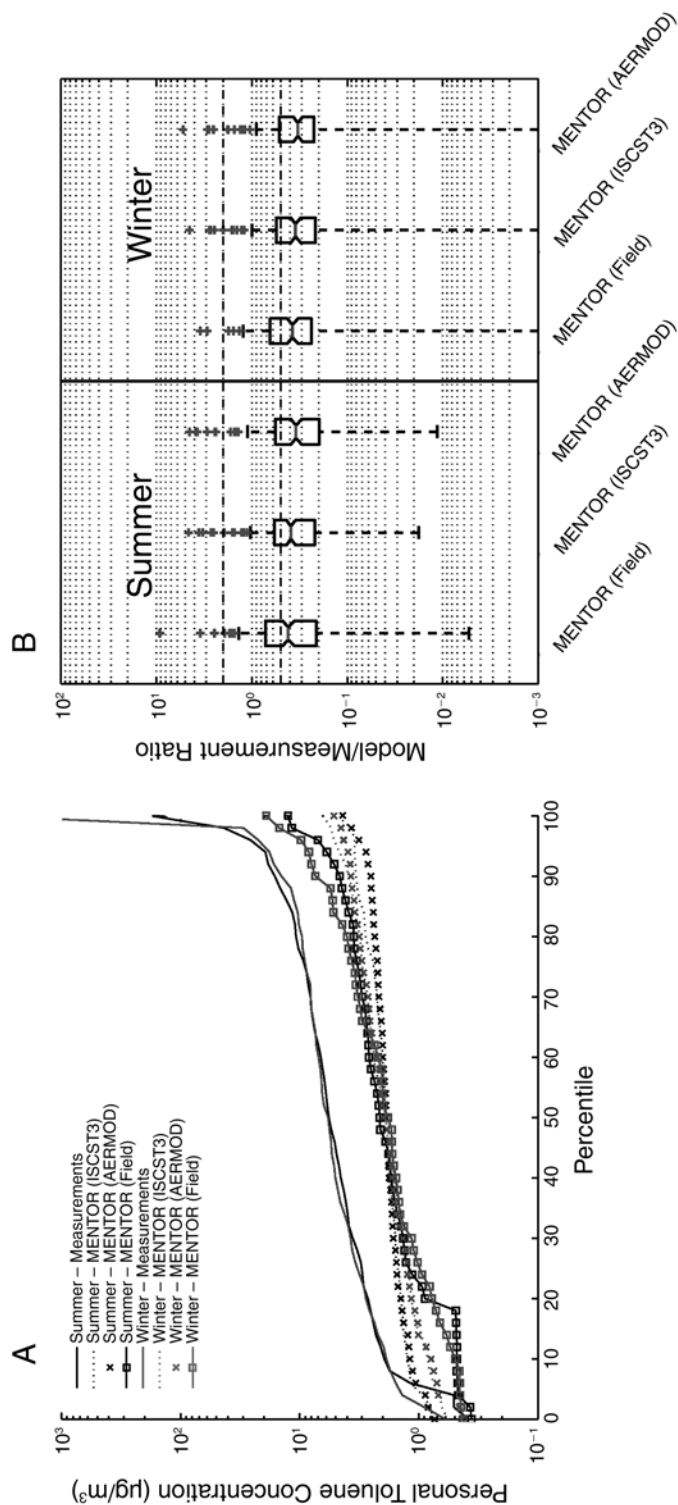


Figure 64. Comparison of cumulative distribution functions for personal toluene measurements and corresponding MENTOR predictions — using ISCST3 estimates, AERMOD estimates, and actual ambient measurements as inputs — for summer and winter on all sampling dates except the 2004 summer hot days (A). Box plots of the corresponding model-to-measurement ratios. For ratios that fall between the horizontal dashed lines, the model result agreed with the measurement result to within a factor of 2 (B).

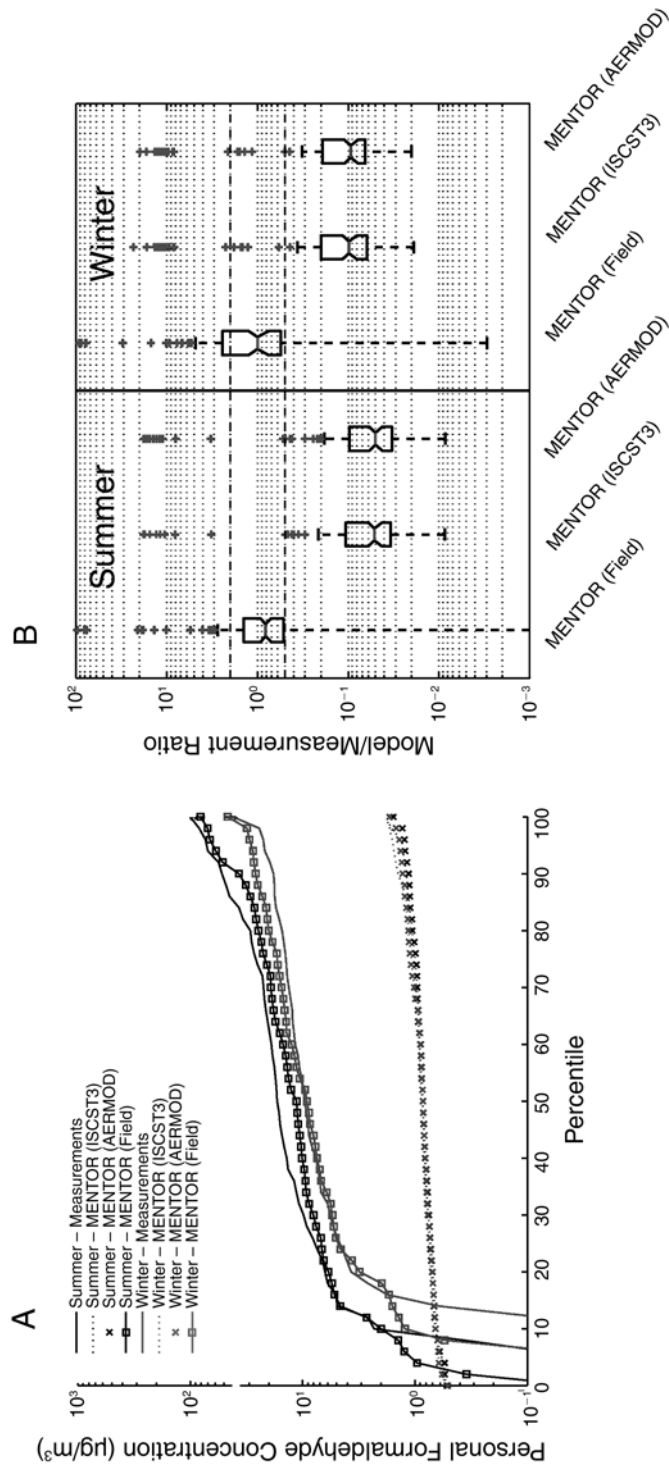


Figure 65. Comparison of cumulative distribution functions for personal formaldehyde measurements and corresponding MENTOR predictions — using ISCST3 estimates, AERMOD estimates, and actual ambient measurements as inputs — for summer and winter on all sampling dates except the 2004 summer hot days (A). Box plots of the corresponding model-to-measurement ratios. For ratios that fall between the horizontal dashed lines, the model result agreed with the measurement result to within a factor of 2 (B).

Analysis of Ambient Source Contributions to Personal

Exposure Predictions Because the IBEM–MENTOR system is able to use various inputs — such as ISCST3 and AERMOD ambient estimates and actual ambient measurements — for estimating personal exposures, the option of using ambient measurements provided a better way of assessing the impact of local ambient sources of certain air toxics on personal exposures in instances where dispersion modeling could not be used to fully characterize the ambient concentrations. Figure 66 shows an example of outputs from a source contribution analysis conducted for benzene and toluene for a subject sampled on July 15, 2006. The differences between the IBEM–MENTOR estimates based on ambient measurements and the personal measurements reflected contributions from indoor and occupational sources; these were higher for toluene than for benzene. The IBEM–MENTOR estimates based on ISCST3 predictions reflected the source contribution profile (based on the five ambient source categories) of the personal benzene and toluene exposures. The relative contributions from the five categories remained the same as for the ambient benzene and toluene concentrations, but the magnitudes of the ambient contributions to the personal concentrations decreased. For toluene, however, as would be expected from the general study and the spatial variation study, outdoor point sources still made significant contributions. Figure 67 shows an example of outputs from a source contribution analysis conducted for personal exposure concentrations of formaldehyde for a subject sampled on July 9, 2005. In this case, the contributions from photochemical production and local (neighborhood) truck activities were not accounted for, and the minor indoor component was not quantified.

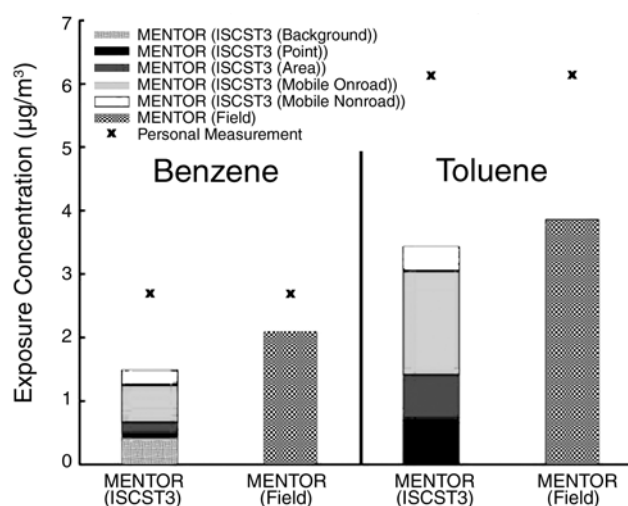


Figure 66. Comparison of MENTOR personal exposure predictions for benzene and toluene (based on ISCST3 source estimates for five source types and actual ambient measurements) with actual personal measurements from a subject sampled on July 15, 2006.

Discussion of Modeling Applications and Measurement Results

Ambient Concentrations of Air Toxics The performance of the ISCST3 and AERMOD models was similar in predicting ambient concentrations of the three air toxics:

- Of the three air toxics, the model predictions for toluene were best. Modeled ambient concentrations of toluene were generally in agreement with ambient measurements by a factor of two or less, a result similar to that reported by Pratt and colleagues (2004).
- For benzene, the model predictions were generally in agreement with the ambient measurements by a factor of two, a result, again, similar to that reported by Pratt and colleagues (2004), who found both under- and over-predictions by factors of two or less. Wheeler and colleagues (2004), however, reported that their ISCST3 predictions had underestimated the monthly averages of monitored ambient benzene values by factors of three or less for one of the three cities studied.
- For formaldehyde, the model predictions were poorest. Our dispersion modeling results significantly underestimated the ambient concentrations and could only account for 4% to 20% of the measured values. Wheeler and colleagues (2004) reported that their ISCST3 predictions had underestimated the monthly averages of monitored ambient formaldehyde values, accounting only for 10% to 30% of them for two of the three cities studied.

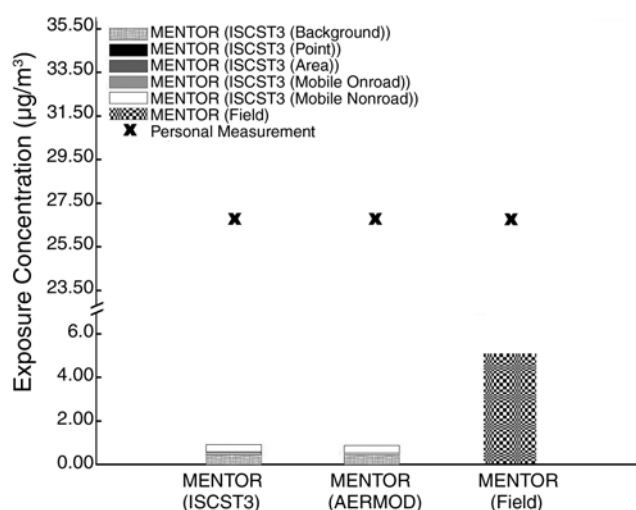


Figure 67. Comparison of MENTOR personal exposure predictions for formaldehyde (based on ISCST3 source estimates from five source types, AERMOD estimates, and actual ambient measurements) with actual personal measurements from a subject sampled on July 9, 2005.

As pointed out by Pratt and colleagues (2004), the performance of dispersion models has rarely been evaluated by comparing their predictions with actual air toxic measurements, especially over averaging times shorter than a year, and that dispersion models perform better as averaging times increase. The dispersion modeling analyses presented in this report provide new insights into model performance by comparing predictions with measurements over 24-hr time periods; these were shorter than the 48-hr and monthly–annual averaging time periods used by Pratt and colleagues (2004) and Wheeler and colleagues (2004), respectively.

As to the effects of location, ambient benzene measurements collected at the two fixed monitoring sites did not show differences between the two neighborhoods. Ambient toluene measurements collected in Waterfront South were consistently higher than those in Copewood–Davis across the entire distribution. Results from the ISCST3 and AERMOD dispersion models both showed spatial differences for benzene and toluene that were consistent with those found in the ambient measurements for the two compounds. For formaldehyde only minor differences in ambient measurements were found between the two fixed monitoring sites. Dispersion modeling results for formaldehyde did not show spatial differences either. Pratt and colleagues (2004) reported a similar finding, namely that ISCST3 predictions could not fully resolve ambient concentrations in areas with sharp emissions gradients for pollutants emitted mainly by mobile sources.

As to the effects of the day of the week, generally very minor weekday–weekend differences were found for ambient benzene and formaldehyde. Dispersion modeling results for benzene and formaldehyde did not show such differences either. Ambient measurements of toluene (which dominated stationary sources in Waterfront South) were consistently higher on weekdays than on weekend days.

As to the effects of season, ambient formaldehyde measurements were higher in summer than in winter. McCarthy and colleagues (2007) reported that, nationally, ambient benzene concentrations tend to be higher in winter than in summer (because of lower mixing heights in winter and higher photochemical removal rates in summer) and that ambient formaldehyde concentrations tend to be higher in summer than in winter (because of higher photochemical production rates in summer). The same trends were observed in earlier studies as well (South Coast Air Quality Management District [SCAQMD] 2000; Battelle 2001; Kinney et al. 2002). The RIOPA study (Weisel et al. 2005), likewise, reported the same trend for outdoor benzene concentrations in Los Angeles and for outdoor formaldehyde concentrations in Los Angeles and Houston. Our trend for benzene did not accord with those of the earlier studies.

Sexton and colleagues (2004) reported no significant seasonal differences for ambient benzene concentrations. Our dispersion modeling for benzene and formaldehyde showed minor seasonal differences.

Ambient toluene measurements generally do not show differences by season (Pratt et al. 2004; Sexton et al. 2004; Weisel et al. 2005), although Kinney and colleagues (2002) found mean outdoor toluene concentrations to be higher in summer than in winter in New York City. The Multiple Air Toxics Exposure Study (SCAQMD 2000) found that ambient concentrations of air toxics related to stationary sources tended to be consistent from season to season and that those related to mobile sources tended to have significant seasonal variations. Because the toluene emissions in our study hot spot had significant contributions from both stationary and mobile sources (as shown by our saturation sampling and personal monitoring data), the effects of the stationary sources might have overwhelmed those of the mobile sources, resulting in the seasonal variations found for toluene.

The key limitation of local-scale dispersion models might be their inputs (i.e., data on emissions, source characteristics, and meteorology). Sax and Isakov (2003) showed that the uncertainties associated with emissions estimates were the largest source of uncertainty in simulating local-scale air toxic impacts. Meteorology, the spatial distribution of sources, and model formulation also contribute to the uncertainty, but they might not be as important as emissions (ENSR 2003). Our underestimates for benzene and toluene at the high-end percentiles and for formaldehyde across the whole distribution were mostly caused by the emissions data used for benzene and toluene and by the emissions data and model formulation (i.e., lack of considering photochemical production) for formaldehyde. The NEI 2002 mobile source emissions inventories were developed at the county level, where mobile onroad emissions are estimated as products of emissions factors (in grams or milligrams per mile) and vehicle miles traveled (VMT). The emissions factors represent long-term county-level vehicle population averages and vehicle activity data. Although these mobile emissions are allocated to census tracts using spatial surrogate factors, such as roadway miles, the resulting emissions estimates might not reflect the actual spatial variations (e.g., local-scale heavy traffic) or temporal variations (e.g., daily traffic patterns) in hot spots like our study areas in Camden — an important point for the design of future hot spot investigations.

For example, according to the NJDEP (2005), more than 100,000 trucks per year pass directly through Waterfront South to deliver materials to industrial facilities located in or near the neighborhood. Moreover, as mentioned earlier,

many of these trucks idle illegally while waiting to pick up or deliver their cargo. Diesel truck emissions are well-known sources of formaldehyde in most urban areas (HEI 1995; Grosjean et al. 2001). For the following four reasons, then, the NEI 2002 mobile source emissions estimates for formaldehyde could have significantly underestimated formaldehyde emissions from local truck traffic in Waterfront South:

- The highest projected growth in daily truck traffic on New Jersey roadways is for Kaighn Avenue in Camden, according to a report (2005) by the Tri-State Transportation Campaign — and the residences of some of our Waterfront South subjects were located adjacent to this street. Estimates of mobile source emissions for the census tract containing the street were obtained by apportioning county-level NEI emissions data using roadway miles as the spatial surrogate factor, resulting in underestimated values.
- According to the same Tri-State Transportation Campaign report (2005), VMTs are increasing for truck traffic in Camden County. NEI 2002 mobile onroad emissions were estimated based on VMTs in 2002, which would underestimate those of our study's time period (2004 to 2006).
- Census-tract-level emissions estimates spread onroad mobile emissions over the entire area of a census tract instead of aligning them with roadways. This could result in underestimated concentrations in dispersion modeling for receptor locations close to roadways. Our two fixed sites were close to roadways.
- Mobile emissions from truck idling were significantly underestimated in the NEI 2002 data. If we assume, for example, that there are 100,000 truck trips per year in the Waterfront South area and (in accordance with a national truck survey [Lutsey et al. 2004]) that each truck trip consumes 1600 gallons of diesel fuel per year when idling, this could result in 136 short tons per year of local formaldehyde emissions, on an emissions factor of 1.7 lb per 1000 gallons of diesel fuel used for combustion (Ventura County Air Pollution Control District 2001). Yet the NEI 2002 estimate of census-tract mobile emissions for formaldehyde was about 11 short tons per year in the Waterfront South area, which is only about 10% of that from the example given above. The difference in truck traffic volumes between the inventory data and our actual measurements is consistent with the percentage underestimate of the model prediction.

Besides primary emissions, a portion of ambient formaldehyde concentrations can also be attributed to secondary formation by photochemical production in summer. In a simulation study of a grid-based photochemical model application in the Philadelphia area, Ching and colleagues (2005) showed that the ratio of secondary formaldehyde formation to primary formaldehyde is from 5 to 10. Further, as mentioned earlier, secondary formation can contribute from 5 ppb (about $6.13 \mu\text{g}/\text{m}^3$) to 10 ppb (about $12.25 \mu\text{g}/\text{m}^3$) in summer (Georgopoulos et al. 2005).

In order to resolve the issue of the mischaracterization of mobile emissions at local scales, more accurate local-scale emissions data need to be collected for areas like Waterfront South and Copewood–Davis. This should be done by using local traffic information to estimate more accurate emissions factors. One source could be vehicle activity data from a travel demand model; this could better characterize hot spots by providing more detailed information on the spatial distribution of traffic by roadway types, vehicle activity, and road speeds (Cook et al. 2006).

Personal Concentrations of Air Toxics

Our IBEM–MENTOR predictions of personal exposures caused by ambient sources were made three ways for each air toxic, using the estimates from ISCST3 and AERMOD dispersion modeling and the actual ambient measurements as inputs. The predictions were then evaluated against the actual personal measurements collected in the field to assess the extent of ambient source contributions to personal exposures. This represents a step forward over the Philadelphia IBEM–MENTOR study by Georgopoulos and colleagues (2005), in which ambient predictions were evaluated against monitored values but, because of a lack of exposure measurements, exposure estimates were not evaluated.

For benzene it was found that the estimated personal exposures above the 75th percentile were dominated by contributions from ambient sources. These exposure estimates were close to the actual personal measurements in this range and much lower than the personal measurements in the lower percentiles, reflecting the influence of indoor sources on personal exposures in the lower percentiles. Several studies (Clayton et al. 1999; Gordon et al. 1999; Pratt et al. 2004; Weisel et al. 2005) have reported personal and indoor benzene concentrations that were higher than outdoor concentrations, suggesting that indoor benzene sources dominated personal and indoor concentrations. In a study of high school students in New York City, however, Kinney and colleagues (2002) reported more similar personal, indoor, and outdoor benzene concentrations, suggesting ambient concentrations as the driving force for personal benzene exposures.

For toluene, estimated personal exposures caused by ambient sources were generally lower than personal measurements, indicating the consistent influence of indoor sources across the entire distribution of the personal exposures. The earlier studies mentioned above (Clayton et al. 1999; Gordon et al. 1999; Pratt et al. 2004; Weisel et al. 2005) also reported that personal toluene exposures were dominated by indoor sources. In the TEACH New York City study, Kinney and colleagues (2002) reported that ambient toluene concentrations were the driving force for personal exposures. However, when the TEACH results for New York City and Los Angeles were combined for data analysis, both ambient and indoor sources were found to be significant for personal toluene exposures (Chillrud et al. 2004).

For formaldehyde, Kinney and colleagues (2002), Chillrud and colleagues (2004), and Weisel and colleagues (2005) found that indoor and personal concentrations were significantly (i.e., one or two orders of magnitude) higher than outdoor concentrations, suggesting strong influences by indoor sources. By contrast, our study found that the estimated personal exposures caused by ambient sources were comparable with the personal measurements. Mean formaldehyde concentrations measured at our fixed monitoring sites were approximately two to three times higher than the ambient values reported by Kinney and colleagues (2002), Chillrud and colleagues (2004), and Weisel and colleagues (2005). Measured personal formaldehyde concentrations were approximately 60% to 80% of the measured ambient concentrations. Our estimated personal exposures resulting from ambient sources were therefore comparable with our personal measurements when taking into account, in IBEM-MENTOR calculations using ambient measurements, the attenuating effect on these concentrations of penetrating from outdoor into indoor environments.

For benzene and formaldehyde the IBEM-MENTOR estimates for personal exposures based on ambient measurements generally predicted higher percentage contributions from ambient sources than the percentages estimated using dispersion modeling. The IBEM-MENTOR results underestimated ambient concentrations of benzene by a factor of two and of formaldehyde by a factor of one order of magnitude as a result of significant underestimates for mobile sources emissions (for which, as discussed, local patterns of heavy truck traffic had not been taken into account). For benzene, junk car recycling might have created an added burden.

The effects of location were shown in differences in percent contributions to personal exposures from local ambient concentrations (as measured at our fixed monitoring sites). The percentages caused by ambient concentrations were generally higher in Waterfront South than in

Copewood-Davis for both benzene and toluene, reflecting the hot spot character of Waterfront South. For formaldehyde, however, major effects of local ambient sources on personal exposures were observed in both neighborhoods.

Seasonal effects were observed for personal measurements of benzene and formaldehyde; they were higher in summer. This trend was driven by local ambient concentrations, because they showed the same seasonal effect as the personal measurements.

As to the day of the week, no effects for benzene or formaldehyde were found in our IBEM-MENTOR predictions or personal measurements. Personal toluene concentrations were higher on weekdays than on weekend days, which might have been caused by the weekly operating cycles of facilities associated with the stationary toluene sources — a point raised previously in this report for various compounds. The IBEM-MENTOR predictions of personal toluene exposures showed the same trend as the personal measurements.

Limitations identified in our modeling analyses include (1) the use of dispersion modeling in estimating local ambient concentrations of air toxics associated with mobile sources of such compounds as benzene and formaldehyde in the hot spot area, (2) the use of estimates of personal exposure derived from ambient sources based on dispersion modeling, and (3) not identifying the potential contributions from indoor sources to personal exposures in the IBEM-MENTOR modeling.

Data gaps identified in our modeling analyses include (1) estimates of contributions from photochemical production to ambient concentrations of such reactive air toxics as formaldehyde, (2) better estimates of mobile emissions and their temporal trends (season and day of the week) by taking local traffic patterns into account, and (3) estimates of indoor source emissions rates in order to conduct microenvironmental and personal exposure modeling to characterize the contributions of indoor sources. Once the performance of emissions-based dispersion modeling is improved, the contributions of the various ambient source categories (background, point, area, mobile onroad, and mobile nonroad) to personal exposures identified by the IBEM-MENTOR source-to-exposure application can be better characterized, leading to improved utilization of data from personal monitoring and ambient monitoring in hot spots. This should also lead to the development of effective control strategies based on generalizations of the results to help reduce personal exposures to ambient sources of air toxics.

DISCUSSION AND CONCLUSIONS

This three-year study has successfully characterized personal and ambient concentrations of PM_{2.5}, VOCs, aldehydes, and PAHs in an air toxics hot spot (Waterfront South) and an urban reference site (Copewood–Davis) in the city of Camden, New Jersey. Spatial variations in VOCs and aldehydes in both neighborhoods were characterized in a spatial variation study consisting of three saturation-sampling campaigns. Baseline and time–activity diary questionnaires were collected from 107 subjects, and time–location information was collected on a subset of subjects to evaluate the application of GPS technology to exposure research.

Our measurements of personal and ambient air samples, combined with the questionnaire data, were successfully

analyzed using both descriptive statistical analysis and mixed-effects modeling to examine the distribution of the target compounds by location, season, and day of the week. Personal and ambient concentrations of the target compounds measured in Waterfront South were compared with those from Copewood–Davis, and measurements from both neighborhoods were compared with national average concentrations. The results are summarized in Table 24. Ambient concentrations of PM_{2.5}, toluene, ethylbenzene, xylenes, and PAHs were significantly higher in Waterfront South than in Copewood–Davis (Table 24). The spatial variation study revealed several hot spots for BTEX and one for MTBE in Waterfront South, indicating that the neighborhood is a hot spot for these pollutants. Based on the study's observations and NJDEP source data, the elevated concentrations of these pollutants can be attributed to the high volume of local truck traffic in the neighborhood as

Table 24. Summary of Ambient Air Toxic Concentrations Found to Be High in Waterfront South and Copewood–Davis

	High in Waterfront South	High in Copewood– Davis	Similar in Waterfront South and Copewood–Davis	Higher Than National Average	
				Waterfront South	Copewood– Davis
VOCs					
MTBE			x		
Hexane			x		
Chloroform			x		
Carbon tetrachloride			x		
Benzene			x	x	x
Toluene	x				
Ethyl benzene	x				
<i>m</i> - & <i>p</i> -Xylenes	x				
<i>o</i> -Xylene	x				
Styrene					
Aldehydes					
Formaldehyde ^a		x		x	x
Acetaldehyde			x	x	x
Acrolein					
Propionaldehyde					
PAHs					
Naphthalene	x				
Phenanthrene	x				
Fluoranthene					
Pyrene	x				
Benzo[<i>a</i>]pyrene	x				
Fine PM					
PM _{2.5}	x			x	x

^a Formaldehyde was found in similar concentrations in the spatial variation study.

well as to a number of local industrial operations, including a sewage plant and a car scrapping facility. In contrast, measurements at our two fixed monitoring sites showed mean concentrations of benzene and MTBE to be slightly lower in Waterfront South than in Copewood–Davis. These results are consistent with the locations of the two neighborhoods relative to the main local roadways. Benzene and MTBE are generated primarily by gasoline-powered vehicles and by evaporation from gasoline and the recycling of junk automobiles. Waterfront South is located upwind of the main roads; Copewood–Davis is downwind. Aldehyde concentrations were found to be high in both neighborhoods, suggesting that both diesel- and gasoline-powered vehicles were the principal direct emission sources of aldehyde pollution in Waterfront South and Copewood–Davis. In addition, PM_{2.5}, BTEX, MTBE, and all the aldehydes in both neighborhoods were found to be higher than New Jersey background and national average concentrations, showing that both neighborhoods are hot spots for these pollutants.

Our field observations and conclusions were confirmed by dispersion modeling results. Modeled concentrations of ambient toluene (which has dominant local stationary sources in Waterfront South) were in reasonable agreement with ambient measurements, although there was an unexplained contribution to personal exposures that was probably caused by personal and occupational sources. However, modeled concentrations of formaldehyde from mobile sources (particularly diesel emissions) were significantly underestimated, accounting for only about 4% to 20% of the measured values; this was consistent with the difference between NEI 2002 estimates for diesel emissions and the actual annual truck traffic passing through and idling in Waterfront South.

Consistent patterns were observed for many compounds, including PM_{2.5}, TEX, and PAHs, namely that concentrations were higher on weekdays than on weekend days, reflecting the higher volumes of vehicle traffic and higher levels of industrial and commercial operations on weekdays. Seasonal variations varied by compound, depending on the compound's sources and its chemical and physical properties. Lighter PAHs, for example, were found to be higher in summer than in winter because of differences in volatilization rates from various surfaces. Heavier PAHs, such as benzo[a]pyrene, were found to be higher in winter than in summer because of increased home heating emissions and the decreased reactivity of benzo[a]pyrene in winter.

For all target compounds, personal exposure concentrations in the two neighborhoods were found to be higher

than or similar to those reported in other studies of other polluted urban areas. For most compounds, personal concentrations were found to be higher than ambient concentrations, suggesting that many of our target pollutants had indoor and possibly occupational sources in addition to outdoor sources. (Personal exposures can be significantly affected by personal activities.) Differences in personal exposures were also observed by location, season, and day of the week. Some of these were consistent with the variations in ambient air concentrations; some were not.

The results from our descriptive statistical analyses, Spearman correlation analyses, and mixed-effects modeling provided a better understanding of the relationships between ambient air pollution and personal exposures. Personal measurements for VOCs and aldehydes that had known local sources, such as toluene, MTBE, and formaldehyde, were highly correlated with ambient concentrations ($R > 0.5$); a substantial portion of the variations in personal concentrations of these compounds was explained by local ambient air pollution based on mixed-effects model analysis ($R^2 > 0.5$). These results indicate the significant impact of these compounds on personal exposures. Ambient measurements from our fixed monitoring sites in each neighborhood were good predictors of average personal exposures to the compounds. For benzene, toluene, and formaldehyde, these observations were consistent with the estimated results obtained from exposure modeling up to and including the 75th percentile for personal measurements predicted by using ambient measurements from the fixed sites. In contrast, poor correlations ($R < 0.4$) were observed between personal and ambient concentrations of PM_{2.5} and PAHs, suggesting that different emission sources might have affected personal exposures to these pollutants.

In Waterfront South, a suspected hot spot for air pollution, large spatial variations were found for pollutants with significant stationary sources, such as MTBE and BTEX; in Copewood–Davis, where there are no local pollution point sources, a relatively homogenous distribution was found for species with primarily mobile or regional sources, such as the targeted VOCs. These results provide detailed support for the notion of an air toxics hot spot, a localized zone of elevated concentrations of air toxics in an area with localized emission sources. Our spatial variation study also showed that emissions from stationary sources have significant effects on adjacent areas (< 300 m); in contrast, the measurements made at our fixed monitoring sites were unable to capture spatial variations in the target compounds, leading to underestimates for

exposures in areas near the stationary sources. The spatial variation study demonstrated that saturation sampling can provide robust data to conduct accurate assessments of area air pollution and personal exposures and help identify major sources of concerns. Personal exposures are also affected by multiple sources and personal activities. For air toxics problems in hot spot areas, then, saturation sampling or personal monitoring are recommended for examining the impact of industrial or other major local sources on community air pollution and personal exposures.

This study has demonstrated the application of IBEM-MENTOR for characterizing ambient and personal exposure concentrations of benzene, toluene, and formaldehyde in a hot spot for air toxics. It successfully characterized the impacts of local ambient concentrations on personal exposure concentrations and facilitated the analysis of effects on personal measurements by location, season, and day of the week. It also built confidence in — and revealed gaps in the data needed for — applying the IBEM-MENTOR approach to exposure modeling in hot spots. The modeling study demonstrated that the use of local ambient measurements and subject-specific data on time-activity patterns is valuable in estimating the percentage contributions of air toxics from ambient concentrations to personal exposures. This is an important new contribution to the science of exposure, because such an analysis had not been conducted in earlier studies.

In conclusion, the study collected data to estimate the impact of emissions from local stationary and mobile sources on ambient air quality and personal exposures in two areas with mixed sources of air toxics. We found that ambient sources of many compounds, such as aldehydes and some VOCs, dominated personal exposures in the study areas. The situation provided a unique database for future studies of health effects associated with exposure to air toxics. In addition, the study investigated the effects of factors such as location, season, and day of the week on ambient concentrations of air toxics and associated personal exposures. The combination of measurement approaches and modeling of individual exposures provided important results; these approaches should continue to be used in studies of the impact of local sources of air pollutants on personal exposures and their potential health effects. Moreover, the study showed that a good understanding of sources and emissions is needed in order to define a location as a hot spot for compounds studied. As shown in Table 24, Waterfront South and Copewood-Davis could both be defined as hot spots for formaldehyde. However, six other compounds were found

at similar concentrations in the two neighborhoods, but they were not hot spot concentrations. It is worth noting that high-exposure episodes for some species were observed intermittently in Waterfront South and that potential acute exposures cannot be well captured by examining mean or median concentrations. Further analysis is therefore recommended to compare the frequency of the occurrence of ambient air toxics in the upper end of the ranges measured in Waterfront South with those from current agency monitoring sites to determine the temporal aspects of Waterfront South as a hot spot for air pollution. Finally, based on the measurements obtained from our study and emissions inventory data from the NJDEP, we have identified diesel truck traffic as the principal source of aldehydes and several industrial facilities as the principal sources of various individual VOCs (such as toluene). These results have been provided to the state of New Jersey to help it develop effective controlling strategies for reducing air pollution in Waterfront South. The state government has already begun to redirect truck traffic, change truck idling procedures, and investigate pollutant source emission practices in the area. More in-depth analyses (including a more comprehensive analysis of our questionnaire data, such as separating the exposure measurements for children and adults) and additional data on indoor sources are needed in order to gain further insight into the observations reported here.

There are limitations to this study. First, the Copewood-Davis location was not a true urban background site, because of its proximity to major roads. In the modern world, however, can one find an urban background site without major roads? It was thus challenging to determine whether Waterfront South was a hot spot for certain pollutants based on comparisons between it and Copewood-Davis. In some cases, measured concentrations were the same in both neighborhoods or lower in Waterfront South. Second, the design of the study was not probability based. Caution should be exercised when making comparisons among the findings obtained from this study and other studies, because differences in personal exposures might also result from differences in study populations. Third, the differences between the personal and ambient concentrations reported in our study and in other studies might be attributable, at least in part, to the various sampling methods and study designs used in the studies. For our spatial variation study, given the limited number of saturation-sampling campaigns conducted, it was not feasible to examine the results for temporal variability or its contribution to spatial variability. However, spatial variation studies of this kind should be explored for future use in both regulatory and exposure health studies.

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APPENDIX A. HEI Quality Assurance Statement

The study was subjected to independent audit by Mr. David Bush of T&B Systems, Inc., and Mr. David Gemmill of Quality Assurance Consulting. Both are experts in quality assurance for air quality monitoring studies and data management. The date of the audit is shown below.

September 22–23, 2008

The auditors conducted on-site audits at the Environmental & Occupation Health Sciences institute (EOHSI) at the University of Medicine and Dentistry of New Jersey. The audit included a review of the project data set utilized in the final report and included a review of procedures for data collection, processing, and analysis. The audit also included an evaluation of the study laboratories, concentrating on standard quality control activities. Specific concerns previously identified by HEI regarding the analysis of carbonyl samples were investigated, with satisfactory results. An issue was identified regarding reported MTBE values resulting from an anomalous analyzer calibration. In addition, it was noted that the PAH detection limits might have been higher than optimum because of sample holding times up to six months. Issues noted during the audit were addressed and corrected in the study's final report.

A written report of the audit was provided to the HEI project manager, who transmitted the findings to the Principal Investigators. The quality assurance audit demonstrated that the study was conducted by an experienced team with a concern for data quality. The final report appears to be an accurate representation of the study.



David H. Bush, Quality Assurance Officer

APPENDIX B. Ambient VOC Concentrations ($\mu\text{g}/\text{m}^3$) in (B.1) Summer and (B.2) Winter for Waterfront South, Copewood–Davis, and NJDEP Monitoring Sites

Table B.1. Ambient VOC Concentrations ($\mu\text{g}/\text{m}^3$) in Summer for Waterfront South, Copewood–Davis, and NJDEP Monitoring Sites

Analyte / Location	N^a	Mean	SD	Minimum	Median	P 75 ^b	P 90 ^c	Maximum
Benzene								
Waterfront South, Camden, NJ	51	2.73	3.79	0.23	1.23	2.23	7.54	15.9
Copewood–Davis, Camden, NJ	44	3.91	6.09	0.15	1.31	2.30	15.8	23.3
Elizabeth, NJ	36	1.22	0.57	0.08	1.18	1.50	1.91	2.97
Camden, NJ	36	1.16	0.49	0.35	1.02	1.49	2.01	2.17
New Brunswick, NJ	36	0.82	0.53	0.35	0.70	0.86	1.44	2.81
Chester, NJ	33	0.47	0.21	0.08	0.48	0.57	0.70	1.15
Carbon Tetrachloride								
Waterfront South, Camden, NJ	28	0.57	0.09	0.40	0.59	0.64	0.67	0.73
Copewood–Davis, Camden, NJ	28	0.58	0.09	0.41	0.60	0.63	0.71	0.77
Elizabeth, NJ	36	0.65	0.21	0.19	0.63	0.75	0.88	1.13
Camden, NJ	36	0.64	0.14	0.44	0.63	0.69	0.88	1.01
New Brunswick, NJ	36	0.63	0.19	0.19	0.63	0.69	0.82	1.13
Chester, NJ	33	0.64	0.20	0.19	0.63	0.69	0.88	1.13
Chloroform								
Waterfront South, Camden, NJ	28	0.28	0.36	0.05	0.19	0.23	0.71	1.85
Copewood–Davis, Camden, NJ	28	0.30	0.32	0.10	0.18	0.28	0.71	1.70
Elizabeth, NJ	36	0.18	0.11	0.01	0.20	0.24	0.29	0.59
Camden, NJ	36	0.14	0.06	0.01	0.15	0.20	0.20	0.29
New Brunswick, NJ	36	0.18	0.11	0.01	0.15	0.20	0.24	0.73
Chester, NJ	33	0.12	0.10	0.01	0.10	0.15	0.15	0.49
Ethylbenzene								
Waterfront South, Camden, NJ	58	0.49	0.39	0.07	0.41	0.69	1.01	2.02
Copewood–Davis, Camden, NJ	50	0.34	0.34	0.07	0.29	0.38	0.50	2.40
Elizabeth, NJ	36	0.58	0.27	0.09	0.56	0.72	0.91	1.30
Camden, NJ	36	0.43	0.19	0.17	0.41	0.56	0.69	0.87
New Brunswick, NJ	36	0.70	0.60	0.13	0.65	0.89	1.39	3.39
Chester, NJ	33	0.21	0.13	0.04	0.17	0.26	0.35	0.61
m- & p-Xylenes								
Waterfront South, Camden, NJ	58	1.42	0.90	0.13	1.25	2.01	2.82	4.27
Copewood–Davis, Camden, NJ	50	0.96	0.69	0.13	0.93	1.21	1.56	4.10
Elizabeth, NJ	36	1.64	0.78	0.11	1.74	2.13	2.91	3.26
Camden, NJ	36	1.20	0.71	0.35	1.04	1.46	2.21	3.56
New Brunswick, NJ	36	1.48	1.89	0.30	1.07	1.52	2.34	11.8
Chester, NJ	33	0.48	0.36	0.09	0.39	0.65	0.87	1.82

Table continues next page

^a N indicates the number of samples collected.

^b P 75 indicates 75th percentile.

^c P 90 indicates 90th percentile.

Table B.1. (Continued). Ambient VOC Concentrations ($\mu\text{g}/\text{m}^3$) in Summer for Waterfront South, Copewood–Davis, and NJDEP Monitoring Sites

Analyte / Location	N ^a	Mean	SD	Minimum	Median	P 75 ^b	P 90 ^c	Maximum
MTBE								
Waterfront South, Camden, NJ	58	2.51	3.22	0.22	2.02	3.16	4.74	23.6
Copewood–Davis, Camden, NJ	50	2.65	3.53	0.24	1.76	3.20	5.57	22.4
Elizabeth, NJ	36	2.20	2.41	0.00	1.86	3.14	6.06	9.64
Camden, NJ	36	1.70	1.91	0.11	1.14	2.78	4.55	8.23
New Brunswick, NJ	36	5.63	29.12	0.00	0.58	1.43	1.77	175
Chester, NJ	33	0.22	0.33	0.00	0.13	0.22	0.43	1.52
o-Xylene								
Waterfront South, Camden, NJ	58	0.50	0.32	0.10	0.44	0.70	0.94	1.63
Copewood–Davis, Camden, NJ	50	0.35	0.26	0.04	0.33	0.43	0.59	1.65
Elizabeth, NJ	36	0.70	0.35	0.09	0.61	0.98	1.17	1.48
Camden, NJ	36	0.49	0.24	0.17	0.43	0.63	0.87	1.09
New Brunswick, NJ	36	0.70	1.03	0.17	0.50	0.70	1.09	6.42
Chester, NJ	33	0.21	0.15	0.04	0.17	0.26	0.35	0.82
Styrene								
Waterfront South, Camden, NJ	58	0.14	0.18	0.00	0.13	0.13	0.24	1.19
Copewood–Davis, Camden, NJ	50	0.11	0.16	0.00	0.08	0.13	0.13	1.14
Elizabeth, NJ	36	0.58	2.23	0.04	0.19	0.30	0.38	13.6
Camden, NJ	36	0.22	0.17	0.04	0.17	0.32	0.43	0.81
New Brunswick, NJ	36	0.34	0.65	0.02	0.19	0.28	0.47	3.88
Chester, NJ	33	0.11	0.10	0.02	0.09	0.13	0.17	0.51
Toluene								
Waterfront South, Camden, NJ	58	4.06	4.95	0.35	2.66	4.22	11.5	27.2
Copewood–Davis, Camden, NJ	50	1.81	1.24	0.35	1.74	2.48	3.62	4.96
Elizabeth, NJ	36	3.12	1.47	0.26	2.96	3.72	5.05	6.90
Camden, NJ	36	2.80	1.60	1.09	2.38	3.13	4.18	10.2
New Brunswick, NJ	36	3.50	6.26	0.60	1.89	2.81	4.64	32.5
Chester, NJ	33	1.06	0.70	0.19	0.90	1.28	2.26	3.36

^a N indicates the number of samples collected.^b P 75 indicates 75th percentile.^c P 90 indicates 90th percentile.

Table B.2. Ambient VOC Concentrations ($\mu\text{g}/\text{m}^3$) in Winter for Waterfront South, Copewood–Davis, and NJDEP Monitoring Sites

Analyte / Location	N ^a	Mean	SD	Minimum	Median	P 75 ^b	P 90 ^c	Maximum
Benzene								
Waterfront South, Camden, NJ	48	1.50	1.29	0.21	1.13	1.74	2.96	7.62
Copewood–Davis, Camden, NJ	42	1.48	1.24	0.44	1.08	1.66	2.89	5.96
Elizabeth, NJ	66	1.68	0.95	0.41	1.47	2.07	2.93	4.47
Camden, NJ	58	1.48	0.70	0.32	1.37	1.63	2.52	4.37
New Brunswick, NJ	64	1.10	0.60	0.32	0.99	1.37	1.75	3.29
Chester, NJ	61	0.78	0.28	0.29	0.73	0.96	1.12	1.56
Carbon Tetrachloride								
Waterfront South, Camden, NJ	41	0.50	0.10	0.37	0.47	0.59	0.63	0.70
Copewood–Davis, Camden, NJ	40	0.52	0.10	0.35	0.50	0.60	0.66	0.73
Elizabeth, NJ	66	0.47	0.21	0.13	0.50	0.63	0.75	0.88
Camden, NJ	58	0.49	0.20	0.06	0.50	0.63	0.69	0.94
New Brunswick, NJ	64	0.48	0.19	0.06	0.50	0.57	0.69	0.88
Chester, NJ	61	0.45	0.19	0.13	0.50	0.57	0.75	0.82
Chloroform								
Waterfront South, Camden, NJ	41	0.12	0.07	0.04	0.10	0.19	0.22	0.27
Copewood–Davis, Camden, NJ	40	0.13	0.13	0.04	0.11	0.16	0.20	0.78
Elizabeth, NJ	66	0.11	0.06	0.01	0.10	0.10	0.15	0.44
Camden, NJ	58	0.09	0.03	0.01	0.10	0.10	0.10	0.15
New Brunswick, NJ	64	0.10	0.03	0.01	0.10	0.10	0.15	0.24
Chester, NJ	61	0.09	0.06	0.01	0.10	0.10	0.10	0.54
Ethylbenzene								
Waterfront South, Camden, NJ	48	0.65	0.55	0.09	0.46	0.87	1.31	2.32
Copewood–Davis, Camden, NJ	42	0.53	0.35	0.16	0.33	0.78	1.13	1.48
Elizabeth, NJ	66	0.68	0.46	0.09	0.56	0.82	1.17	2.39
Camden, NJ	58	0.50	0.28	0.04	0.43	0.65	0.95	1.39
New Brunswick, NJ	64	0.40	0.26	0.04	0.37	0.52	0.65	1.43
Chester, NJ	61	0.20	0.12	0.04	0.17	0.26	0.30	0.61
m- & p-Xylenes								
Waterfront South, Camden, NJ	48	2.06	1.78	0.32	1.48	2.46	4.99	7.42
Copewood–Davis, Camden, NJ	42	1.45	1.07	0.26	1.08	2.15	2.99	4.34
Elizabeth, NJ	66	1.91	1.48	0.22	1.46	2.34	4.17	6.73
Camden, NJ	58	1.29	0.74	0.04	1.13	1.69	2.43	3.47
New Brunswick, NJ	64	0.99	0.67	0.04	0.91	1.29	1.74	3.34
Chester, NJ	61	0.45	0.33	0.04	0.39	0.56	0.69	1.65

Table continues next page

^a N indicates the number of samples collected.

^b P 75 indicates 75th percentile.

^c P 90 indicates 90th percentile.

Table B.2. (Continued). Ambient VOC Concentrations ($\mu\text{g}/\text{m}^3$) in Winter for Waterfront South, Copewood–Davis, and NJDEP Monitoring Sites

Analyte / Location	N ^a	Mean	SD	Minimum	Median	P 75 ^b	P 90 ^c	Maximum
MTBE								
Waterfront South, Camden, NJ	48	2.21	2.17	0.04	1.46	2.76	5.45	9.79
Copewood–Davis, Camden, NJ	42	1.90	1.55	0.04	1.43	2.79	3.36	7.54
Elizabeth, NJ	66	2.15	2.74	0.00	1.32	2.82	4.51	14.6
Camden, NJ	58	1.42	1.51	0.00	1.05	1.95	3.50	6.79
New Brunswick, NJ	64	0.80	0.90	0.00	0.58	1.05	1.84	5.02
Chester, NJ	61	0.23	0.29	0.00	0.13	0.22	0.51	1.62
o-Xylene								
Waterfront South, Camden, NJ	48	0.71	0.69	0.08	0.47	0.87	1.68	3.86
Copewood–Davis, Camden, NJ	42	0.47	0.38	0.11	0.38	0.65	0.90	1.96
Elizabeth, NJ	66	0.87	0.61	0.09	0.69	1.26	1.65	2.95
Camden, NJ	58	0.57	0.34	0.04	0.48	0.78	1.13	1.69
New Brunswick, NJ	64	0.44	0.32	0.04	0.39	0.54	0.78	1.61
Chester, NJ	61	0.21	0.15	0.01	0.17	0.26	0.39	0.74
Styrene								
Waterfront South, Camden, NJ	48	0.15	0.15	0.00	0.14	0.14	0.18	1.03
Copewood–Davis, Camden, NJ	42	0.13	0.10	0.00	0.13	0.14	0.18	0.52
Elizabeth, NJ	66	0.16	0.09	0.02	0.13	0.21	0.30	0.51
Camden, NJ	58	0.29	0.67	0.04	0.17	0.26	0.38	5.24
New Brunswick, NJ	64	0.13	0.06	0.02	0.09	0.17	0.21	0.30
Chester, NJ	61	0.10	0.05	0.02	0.09	0.09	0.17	0.34
Toluene								
Waterfront South, Camden, NJ	48	4.38	5.36	0.57	2.37	5.00	10.3	28.1
Copewood–Davis, Camden, NJ	42	3.49	4.40	0.74	2.13	4.22	6.38	27.5
Elizabeth, NJ	66	3.80	2.95	0.53	2.87	5.05	7.58	13.4
Camden, NJ	58	3.05	2.40	0.23	2.41	3.36	6.22	14.7
New Brunswick, NJ	64	2.16	1.82	0.23	1.66	2.55	3.81	10.1
Chester, NJ	61	1.01	0.68	0.19	0.79	1.09	1.81	3.09

^a N indicates the number of samples collected.^b P 75 indicates 75th percentile.^c P 90 indicates 90th percentile.

APPENDICES AVAILABLE ON THE WEB

Appendices C, D, E, F, G, H, and I contain supplemental material not included in the printed report. They are available on the HEI Web site at <http://pubs.healtheffects.org>.

Appendix C. Baseline Questionnaire

Appendix D. Activity Questionnaire

Appendix E. Time Diary & Activity Log

Appendix F. GeoLogger Evaluation

Appendix G. Time Window Obtained from Each Subject During the Sampling Period

Appendix H. Detailed Modeling Methods

Appendix I. Comparisons of Modeling Results

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Joann Held, Ph.D., is a retired former research scientist for the NJDEP.

OTHER PUBLICATIONS RESULTING FROM THIS RESEARCH

Wu XM, Fan ZT, Ohman-Stickland P. 2010. Time-location patterns of a population living in an air pollution hotspot. *J Environ Pub Health* 2010:625461. Epub 2010 Apr 22.

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ABBREVIATIONS AND OTHER TERMS

AERMOD	American Meteorological Society–U.S. EPA Regulatory Model
ATEOS	Airborne Toxic Element and Organic Substances
BTEX	benzene, toluene, ethylbenzene, and xylenes
CDF	cumulative distribution function
DNPH	2,4-dinitrophenylhydrazine
DNSH	dansylhydrazine
EOHSI	Environmental and Occupational Health Sciences Institute
ETS	environmental tobacco smoke
EXPOLIS	European Exposure Assessment Project
GC–MS	gas chromatography–mass spectrometry
GPS	global positioning system
HPLC	high-performance liquid chromatography
I-676	Interstate 676
IBEM	Individual Based Exposure Modeling

IR	Investigators' Report	QA–QC	quality assurance and quality control
ISCST3	Industrial Source Complex Short Term Version 3	R	Spearman correlation coefficient
MATES	Multiple Air Toxics Exposure Study	R^2	coefficient of determination
MDL	method detection limit	RFA	Request for Application
MENTOR	Modeling Environment for Total Risk	RIOPA	Relationship of Indoor, Outdoor and Personal Air (study)
MTBE	methyl <i>tert</i> -butyl ether	%RSD	relative standard deviation
NEI	National Emissions Inventory	SCAQMD	South Coast Air Quality Management District
NJ-168	New Jersey Route 168	SD	standard deviation
NJDEP	New Jersey Department of Environmental Protection	TEACH	Toxic Exposure Assessment, a Columbia/Harvard (study)
OVM	organic vapor badge	TEAM	Total Exposure Assessment Methodology
PAH	polycyclic aromatic hydrocarbon	TEX	toluene, ethylbenzene, and xylenes
PAKS	passive aldehydes and ketones sampler	UMDNJ	University of Medicine and Dentistry of New Jersey
PAMS	Photochemical Assessment Monitoring Station	U.S. EPA	U.S. Environmental Protection Agency
PM	particulate matter	VMT	vehicle miles traveled
PM _{2.5}	particulate matter with an aerodynamic diameter $\leq 2.5 \mu\text{m}$	VOC	volatile organic compound
PUF	polyurethane foam		

Research Report 160, *Personal and Ambient Exposures to Air Toxics in Camden, New Jersey*, P.J. Lioy et al.

INTRODUCTION

Air toxics comprise a large and diverse group of air pollutants that, with sufficient exposure, are known or suspected to cause adverse effects on human health. The Clean Air Act requires the U.S. Environmental Protection Agency (U.S. EPA*) to characterize, prioritize, and address the effects of air toxics on public health and the environment. However, the characterization of ambient concentrations and personal exposures to air toxics remains challenging, generally as a result of the low ambient concentrations of individual compounds and the multiple indoor and outdoor sources that contribute to exposure. Consequently, the assessment of possible health risks from exposure to air toxics has also been difficult.

HEI has had a longstanding commitment to and involvement in understanding the health effects of air toxics (see Scientific Background, below, for further information). As the Preface accompanying this report makes clear, however, better characterization of exposures to air toxics should be undertaken — especially at sites of possible high exposures — before conducting health effects studies. Thus, in 2003, HEI issued Request for Application (RFA) 03-1, “Assessing Exposure to Air Toxics,” to support research to identify and characterize exposure to air toxics at so called hot spots, areas where concentrations of one or more air toxics are expected to be “higher than those to which the broader public is exposed.” As will be described subsequently, various alternative definitions of a hot spot have been used by other funding agencies and researchers, including the authors of this report.

In response to the RFA, Dr. Paul J. Lioy of the Environmental and Occupational Health Sciences Institute of

Piscataway, New Jersey, and colleagues submitted an application, “Assessing Personal Exposure to Air Toxics in Camden, New Jersey.” The investigators proposed to measure ambient concentrations of and personal exposures to fine particulate matter (PM) — that is, PM with an aerodynamic diameter $\leq 2.5 \mu\text{m}$ (PM_{2.5}) — and several individual air toxics, including volatile organic compounds (VOCs), aldehydes, and polycyclic aromatic hydrocarbons (PAHs), for approximately 100 residents living in two neighborhoods in the city of Camden, New Jersey. They chose one neighborhood with a high density of industrial facilities and mobile sources as their potential air toxics hot spot and a nearby neighborhood without industrial facilities as their control, or comparison, site. Lioy and colleagues also proposed to assess the impact of industrial and mobile sources on ambient concentrations and personal exposures to these air pollutants and to compare measured personal and ambient concentrations of the target compounds with values derived by a model they had developed. The investigators proposed a pilot study of 10 homes to evaluate these proposed methods and to identify and address any potential problems in the main study.

The HEI Research Committee thought Dr. Lioy’s proposal was interesting and generally well designed and that the investigators would be able to get suitable access to the populations they proposed to evaluate in order to make the necessary measurements of pollutant concentrations. After discussions with the Research Committee about issues that needed to be addressed in the pilot study — which included choosing subjects in homes that spanned a wide range of predicted exposures and explaining how concentrations below the limit of detection (“nondetects”) would be handled in the data analysis — Dr. Lioy agreed to make the changes suggested by the Committee, and the Committee recommended the study for funding.

SCIENTIFIC BACKGROUND

As described in the accompanying Preface, air toxics are a large and diverse group of compounds that are generated by multiple sources; understanding exposures to and the effects of air toxics generated by mobile sources is of particular concern to the U.S. EPA. The accompanying

Dr. Paul J. Lioy’s 3-year study, “Assessing Personal Exposure to Air Toxics in Camden, New Jersey,” began in December 2003. Total expenditures were \$937,392. The draft Investigators’ Report from Lioy and colleagues was received for review in May 2007. A revised report was received in May 2009, and the edited report was accepted for publication in April 2011. During the review process, the HEI Health Review Committee and the investigators had the opportunity to exchange comments and to clarify issues in both the Investigators’ Report and the Review Committee’s Commentary.

This document has not been reviewed by public or private party institutions, including those that support the Health Effects Institute; therefore, it may not reflect the views of these parties, and no endorsements by them should be inferred.

* A list of abbreviations and other terms appears at the end of the Investigators’ Report.

sidebar summarizes information about the most common sources of the air toxics and PM that were evaluated in the current study. Summary information about the concentrations of multiple air toxics in various microenvironments relevant to the current study — including benzene, acetaldehyde, formaldehyde, and naphthalene and several other PAHs — can be found in HEI's special report on mobile-source air toxics (MSATs) (HEI Air Toxics Review Panel 2007).

Assessments of ambient concentrations of and personal exposures to air toxics have been challenging, generally, as a result of the low ambient concentrations of individual compounds, multiple indoor sources that contribute to exposure, and lack of adequate sampling methods for some important species. Prior to the current set of studies funded by HEI to characterize possible hot spots, the U.S. EPA conducted the Total Exposure Assessment Methodology (TEAM) studies from 1980 through 1988 to characterize exposures to air toxics at locations across the United States where concentrations were thought to be high (Wallace et al. 1987, 1991). The TEAM studies collected data on personal exposures to VOCs and PM_{2.5} in 10 U.S. cities for more than 1000 persons and showed [a] that, for the most prevalent VOCs and PM, personal exposure concentrations were consistently higher than either indoor or outdoor concentrations and [b] that indoor

concentrations dominated personal exposures, implying that indoor sources (such as consumer products) and personal activities contributed substantially to total exposures (Clayton et al. 1993; Özkaynak et al. 1996).

To better define the relation between outdoor, indoor, and personal exposure concentrations of PM_{2.5} and air toxics, HEI and the Houston-based National Urban Air Toxics Research Center co-funded the Relationships of Indoor, Outdoor, and Personal Air (RIOPA) study (Weisel et al. 2005; Turpin et al. 2007). The RIOPA study was conducted in three urban centers with different weather conditions and air pollution source profiles, namely Los Angeles, California, which is dominated by mobile sources; Houston, Texas, which is dominated by large industrial stationary and area sources (with a portion contributed by mobile sources); and Elizabeth, New Jersey, which has a mixture of mobile, point, and area sources.

In each city approximately 100 homes and 100 adult residents of those homes were selected; the homes were located at various distances from point (large stationary), area, and mobile sources. Samples of VOCs, carbonyls, and PM_{2.5} (as well as some components of PM_{2.5}) were collected inside and outside the homes and in subjects' personal air (breathing zone) during two 48-hr sampling

SOURCES OF POLLUTANTS EVALUATED^a

The compounds measured in the Liroy study are grouped into categories based on their chemical structure. Many have common outdoor and indoor sources and are present in cigarette smoke^b; they are summarized below.

VOCs

BTEX

Benzene, toluene, ethyl benzene, and the xylenes (collectively known as BTEX) are constituents of petroleum products, particularly gasoline, jet fuel, and kerosene. As pollutants, they result from combustion and are emitted from motor vehicle tailpipes, oil refineries, and hazardous waste sites; they also evaporate from vehicles and from solvents in paints and coatings.

MTBE

MTBE was a widely used oxygenate that was added to gasoline to increase octane and reduce emissions of carbon monoxide and benzene. Exposures to MTBE result from both tailpipe and

evaporative emissions (including vapors during refueling). In the United States, MTBE use was phased out in the early 2000s because of concerns about groundwater contamination. At the time samples were collected for the current study and for the earlier RIOPA study, MTBE was still in use in New Jersey.

Hexane

Hexane is used as a solvent in industrial facilities, in the production of elastomers, in metal processing, in automobile repair shops, and in recycling plants.

Chlorinated Compounds

Chloroform in the air is generally a byproduct of chlorine use. The primary sources of chloroform are pulp and paper mills where chlorine is used as a bleach and water treatment plants where chlorine is used as a disinfectant. Chloroform is also emitted from industrial facilities that use chlorine to make other chemicals. Chlorine is present in household bleach products and as a solvent in a variety of household products; it can also derive from chlorinated water.

Carbon tetrachloride is used primarily as a solvent, refrigerant, and propellant. It is released by oil refineries and other industrial facilities. Because of its harmful effects, it has been banned from consumer products and is only used in industrial processes.

Sidebar continues next page

^a Information adapted from the Commentary section of the RIOPA study, Part I (Weisel et al. 2005), and HEI Special Report 16 (HEI Air Toxics Review Panel 2007).

^b The principal air toxics in cigarette smoke are benzene, benzo[a]anthracene, benzo[a]pyrene, PAHs, N-nitrosamines, aldehydes (formaldehyde and acetaldehyde), and 1,3-butadiene.

periods in various seasons between summer 1999 and spring 2001.

Measured concentrations of air toxics were highly variable for all air toxic species within and across the three cities (Weisel et al. 2005). However, considering the wide variety of pollutant sources and weather, the overall relationships among indoor, outdoor, and personal air samples for most compounds were unexpectedly similar for all three cities. Thus, with a few exceptions, mean and median personal exposures to VOCs and carbonyls were similar to their measured indoor concentrations and higher than the outdoor concentrations both within the whole data set and within individual cities. Mean and median personal exposure concentrations of PM_{2.5} were higher than indoor and outdoor concentrations, and indoor and outdoor concentrations were very similar. Several compounds — including methyl *tert*-butyl ether (MTBE) and trichloroethylene — were identified as being primarily of outdoor origin, some (including acetaldehyde, formaldehyde, and chloroform) as being primarily of indoor origin, and some (acrolein, PM_{2.5}, and *m*-, *o*-, and *p*-xylenes) as being derived from both indoor and outdoor sources.

Several years ago, the European Exposure Assessment Project (EXPOLIS) characterized ambient and personal exposures to aldehydes and other carbonyl compounds at the residences and workplaces of 15 residents of Helsinki, Finland (Jurvelin et al. 2001, 2003). The investigators found that personal exposures to formaldehyde and acetaldehyde were lower than indoor residential concentrations and that ambient air concentrations were lower than both indoor residential concentrations and personal exposure concentrations (Jurvelin et al. 2001). The investigators concluded that non-traffic-related emissions were the principal sources of personal exposures to these compounds (Jurvelin et al. 2003). Kinney and colleagues (2002) found quite variable relationships among personal, home indoor, and home outdoor concentrations of VOCs and aldehydes in high school students living in an inner-city neighborhood of New York City.

In the United States, state agencies have also attempted to characterize exposures to air toxics and define potential hot spots. Starting in the 1980s and continuing until 2005, California's South Coast Air Quality Management District set up an air toxics evaluation program to quantify the population exposure risk from existing sources of selected

SOURCES OF POLLUTANTS EVALUATED (Continued)

1,3-Butadiene and Styrene

1,3-Butadiene is derived from the incomplete combustion of petroleum-based fuels. Mobile sources are responsible for most emissions; area and point sources include petroleum refineries, residences, and industrial operations. Vehicles with malfunctioning catalysts emit much higher concentrations than vehicles with functioning catalysts. Other 1,3-butadiene sources are industrial facilities that manufacture tires, other rubber products, resins, plastics and latex paint.

Styrene is released from mobile sources and by industrial facilities that manufacture tires, other rubber products, plastics, and resins.

PAHs

Approximately 100 PAHs have been identified in air, soil, food, and water. They are products of the incomplete combustion of a variety of sources and are produced by high-pressure processes. The sources include motor vehicles (PAHs are a significant component of diesel exhaust), residential heating with fossil fuels, industrial plants, air traffic, and wood combustion.

ALDEHYDES

Several aldehydes, particularly formaldehyde, acetaldehyde,

acrolein, and propionaldehyde, are present in emissions from motor vehicles, power plants using fossil fuels, incinerators, wood combustion, and cooking. They are also naturally formed in ambient air through photochemical oxidation of hydrocarbons in the presence of hydroxyl radicals and ozone. Photochemical oxidation produces more aldehydes than direct emission does, particularly during the daytime hours and on warm, sunny days. Formaldehyde is released from building materials, carpets, ordinary paper products, and indoor combustion sources. Acetaldehyde is used in producing perfumes.

PM

PM is the product of the incomplete combustion of fossil fuels. It is also produced as a result of vehicular brake wear, soil erosion, and sea spray. PM can be generated by outdoor sources and indoor activities (such as cooking, vacuuming, or burning incense and candles). Based on aerodynamic diameter in urban air, PM volume distribution is conventionally classified in three size modes: coarse particles (> 1 µm), fine particles (0.1 to 1 µm), and ultrafine particles (< 0.1 µm). Maximum outdoor ambient concentrations of fine PM are regulated by the U.S. EPA through the National Ambient Air Quality Standards for PM_{2.5}.

air toxics. These studies — the Multiple Air Toxics Exposure Study (MATES) I, II, and III — established and updated an emissions inventory for air toxics and determined the theoretic carcinogenic risk for multiple air toxics at various sites in the South Coast Air Basin of California. For these studies a hot spot was defined as an “area where the concentration of air toxics is at a level where individuals may be exposed to an elevated risk of health effects” (California Air Resources Board 1987). MATES II, conducted in 1999, concluded that the contribution to risk from air toxics in ambient air was dominated by emissions from mobile sources, particularly diesel PM emissions.

In the 1980s New Jersey instituted a program — Airborne Toxic Element and Organic Substances — to measure and reduce exposures to pollutants. The results indicated that Camden and other locations, including Newark and Elizabeth, in the state had higher concentrations of ambient air toxics than other sites in the state (Harkov et al. 1983, 1984; Lioy 1990). These studies suggested that Camden had several sources of air toxics, including mobile sources with a relatively high component of diesel emissions, several industrial and manufacturing sources, and contributions from the pollution plume associated with the Philadelphia metropolitan area. More specifically, the principal sources of air toxics in Camden were located in or near the neighborhood of Waterfront South, where nearly all homes were located 1 km or less from some stationary source of air toxics. As described in the section below and subsequently, the intention of the current study by Lioy and colleagues was to follow up and extend these findings by measuring ambient and personal exposures to air toxics in this potential hot spot.

STUDY OBJECTIVES AND SPECIFIC AIMS

This study was conducted to characterize ambient concentrations and personal exposures to air toxics and PM_{2.5} in two neighborhoods in Camden, New Jersey. The investigators selected a potential hot spot, Waterfront South, with several known industrial and mobile sources of air toxics and, for reference purposes, the nearby neighborhood around the intersection of Copewood and Davis streets, which had no known sources of industrial emissions. The investigators also proposed to assess the impact of local stationary and mobile sources on the air quality of the two neighborhoods and the personal exposures of the participants. The key goals of the study were (a) to demonstrate that Waterfront South is a hot spot for air toxics based on measured ambient concentrations and (b) to determine whether a hot spot assessed by ambient measurements is also associated with elevated personal exposures.

The study had the following specific aims:

1. To quantify personal exposures to air toxics for people living in the Waterfront South and Copewood–Davis neighborhoods;
2. To quantify ambient concentrations of air toxics in the two neighborhoods;
3. To characterize spatial variations in VOCs and aldehydes in both locations;
4. To examine differences in concentration and composition between the targeted air toxics in the two neighborhoods;
5. To compare the temporal (weekdays versus weekend days) and seasonal (summer versus winter) variations in the targeted air toxics in the two neighborhoods;
6. To examine the associations between ambient air pollution and personal exposures; and
7. To assess the impact of local sources of air toxics, particularly diesel emissions, on personal exposures and air quality in the two neighborhoods based on measurements of air toxics with support from exposure modeling, using the authors' source-to-dose Modeling Environment for Total Risk (MENTOR) system (Georgopoulos et al. 2005; Georgopoulos and Lioy 2006).

Lioy and colleagues also evaluated a GeoLogger global positioning system (GPS) device for recording personal movements in order to evaluate the application of GPS technology to exposure research. The investigators considered this evaluation to be exploratory and not part of the main study; results from it are included in Appendix F of the Investigators' Report (available on the HEI Web site).

STUDY DESIGN

CHARACTERISTICS OF WATERFRONT SOUTH AND COPEWOOD–DAVIS

As shown in Figure 1 of the Investigators' Report, Waterfront South — the study's potential hot spot — has an area of about 1 km²; it has a population of approximately 1700. Within or near the neighborhood (Table 1 in the Investigators' Report) are 26 industrial and manufacturing facilities (including a municipal waste and sewage treatment facility, recycling plants, and metal processing facilities). In addition, it has been estimated that several hundred trucks per day travel to, through, or idle in the neighborhood to service local industry. Further, two major roads with heavy traffic volumes (Interstate 676, with 80,000 vehicles/day, and New Jersey Route 168 [NJ-168], with 25,000 vehicles/day) are located close to the eastern boundary of the neighborhood

(New Jersey Department of Transportation 2006). Copewood–Davis, the control area, is residential, with a population of approximately 6,200, and is located about 1 km east of Waterfront South (Figure 1 in the Investigators' Report). There are no identifiable industrial facilities in or near (< 1000 m) the neighborhood, but it is bounded by two local major roads, one of which, NJ-168, the investigators indicated contributes to mobile-source pollution to the west in Waterfront South (see above).

PARTICIPANTS

The investigators recruited 107 participants, 54 from Waterfront South (of whom 17 were children) and 53 from Copewood–Davis (of whom 19 were children), all from non-smoking households. The residents of Waterfront South have been characterized as low income, with more than 33% in poverty (New Jersey Department of Environmental Protection [NJDEP] 2005), and as predominantly nonwhite (69.3% black, 27.2% Hispanic). The residents of Copewood–Davis were similar in socioeconomic status and background (69.3% black, 25.6% Hispanic) to those of Waterfront South. About 80% of the study participants from Copewood–Davis lived within three blocks (< 500 m) of NJ-168.

Nonsmoking households were chosen to reduce the impact of environmental tobacco smoke (ETS) on measured exposures and to improve the ability to evaluate the impact of ambient and other indoor sources. Recruitment for the study was carried out by multiple means, such as through local community leaders, work with local liaisons, attending local community events, advertisements in local newspapers, and by word of mouth.

The investigators modified two questionnaires developed for the RIOPA study and administered them to each participant to obtain information on demographics, time spent in various microenvironments, and contact with ETS and indoor sources of pollution. These questionnaires are shown in Appendices C through E of the Investigators' Report (available on the HEI Web site).

AMBIENT MONITORING SITES

To represent ambient concentrations of air toxics and PM in Waterfront South and Copewood–Davis, a fixed monitoring site was selected in each neighborhood (see Figure 3 in the Investigators' Report). In Waterfront South the site was located at Sacred Heart Church, one block from an intersection that was the entrance for truck traffic traveling through the neighborhood; it was not close (< 50 m) to stationary sources of air toxics. In Copewood–Davis the investigators used the NJDEP's ambient monitoring site, at the

intersection of Copewood and Davis streets, as their fixed monitoring site; it, too, was not close to stationary sources.

AIR POLLUTANT SAMPLING AND MEASUREMENT

For each participant the investigators collected four 24-hr personal air samples between June 2004 and July 2006; two samples were collected in summer (on weekdays and weekend days) and two in winter (on weekdays and weekend days). Simultaneous to the personal sampling, 24-hr ambient air samples were collected at the fixed monitoring sites in each neighborhood. Details of the sample collection techniques are given in the Investigators' Report, in the Measurement of Air Pollutants section.

Liroy and colleagues measured concentrations of PM_{2.5}, VOCs, aldehydes, and PAHs as described below. Generally, they used the same samplers to collect both ambient and personal samples. To calculate the concentrations of individual VOCs, aldehydes, and PAHs in a sample, the investigators set up calibration curves using standard solutions of each compound and also calculated values for solvent, laboratory, and field blanks.

PM_{2.5} PM_{2.5} was collected by an SKC sampling pump and PM_{2.5} impactor onto Teflon filters; PM_{2.5} mass was determined gravimetrically.

VOCs VOCs were collected on passive sampler badges and were extracted and analyzed by gas chromatography combined with mass spectrometry (GC–MS). The extraction efficiency for each VOC was determined and found to be similar to those determined by the manufacturer of the badges. The investigators focused on the compounds MTBE, hexane, chloroform, carbon tetrachloride, benzene, toluene, ethylbenzene, *m*- & *p*-xylenes, *o*-xylene, and styrene. They intended to evaluate 1,3-butadiene concentrations as well, but these were not included in the report because of what the investigators described as “large uncertainties in the measurement method.”

Aldehydes Aldehydes were collected on a dansylhydrazine (DNSH)-coated silica-based bonded sorbent in passive aldehydes and ketones sampler (PAKS) cartridges, which were also used in the RIOPA study (Weisel et al. 2005). Aldehydes react with DNSH to form their corresponding hydrazone derivatives. These compounds were subsequently extracted and quantified using high-performance liquid chromatography with fluorescence detection. Four aldehydes (formaldehyde, acetaldehyde, propionaldehyde, and acrolein) were quantified in this study.

PAHs PAHs were collected on a Teflon filter and two polyurethane foam cartridges connected in series downstream of the filter. Deuterated PAHs were spiked into samples to measure the extraction efficiency. The PAHs were extracted and separated by GC–MS, and 16 target compounds were analyzed: naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benzo[*a*]anthracene, chrysene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, indeno[1,2,3-*cd*]pyrene, benzo[*ghi*]perylene, and dibenzo[*a,h*]anthracene. Results were shown as the sum of each compound's concentrations in both gaseous and particle-bound phases.

The investigators focused their comparative analyses of ambient concentrations and personal exposures to PAHs on four compounds — naphthalene, phenanthrene, pyrene, and benzo[*a*]pyrene — to represent groups of PAHs with from two to five fused benzene rings. Naphthalene and phenanthrene were also analyzed because they were detected at the highest concentrations among the 16 PAHs, and benzo[*a*]pyrene was analyzed because of its known adverse health effects.

Nicotine as a Marker of Exposure to ETS All the recruited participants were nonsmokers, but during the study the investigators realized that exposure to ETS was widespread in both neighborhoods. They reasoned that exposure to ETS would likely contribute to exposures to air toxics and PM and would interfere with the evaluation of exposure to local sources. Thus, from some samples collected for PAH assessment, the investigators used GC–MS to measure nicotine concentrations; they considered a nicotine concentration of 0.5 µg/m³ or greater to reflect exposure to ETS.

SPATIAL SATURATION SAMPLING

To evaluate the spatial variability in the concentrations of air toxics in each neighborhood, the investigators conducted a saturation-sampling substudy in which they measured VOCs and aldehydes at 22 sampling sites in Waterfront South (mostly at street intersections close to local sources, especially major industrial sources) and at 16 sites in Copewood–Davis (mostly along roads, to capture variations in vehicle emissions). The substudy consisted of three campaigns, two in summer (collecting samples over approximately 24 or 48 hr in July and August 2005) and one in winter (collecting samples over approximately 48 hr in December 2005). Hourly meteorologic data (primarily wind speed, wind direction, ambient temperature, and relative humidity) were obtained from a weather monitoring station in Philadelphia.

MODELING OF POLLUTANT CONCENTRATIONS

The investigators estimated ambient and personal air concentrations of three air toxics — benzene (emitted by automobiles), toluene (which the investigators concluded was emitted by both automobile and stationary sources in Waterfront South), and formaldehyde (measured at high concentrations throughout the study) — using the MENTOR modeling system. This system was developed by the investigators for multi-scale source-to-dose modeling of exposures to multiple contaminants (Georgopoulos et al. 2005; Georgopoulos and Lioy 2006).

Ambient concentrations of the three air toxics were estimated at the fixed monitoring sites in the two neighborhoods by two Gaussian plume atmospheric dispersion models, the Industrial Source Complex Short Term Version 3 (ISCST3) and the American Meteorological Society–U.S. EPA Regulatory Model (AERMOD). Inputs for both models included point, area, mobile onroad, and mobile nonroad source emissions data from the 2002 National Emissions Inventory for Camden County and meteorologic data (wind speed, wind direction, stability category, and mixing height) from the Philadelphia International Airport (approximately 8 miles [13 km] away). Annual county-level emissions data were converted to hourly data and apportioned within census tracts for all sources, except onroad emissions, for which the models used road links.

To evaluate the models' performance, Lioy and colleagues compared the concentrations modeled for the fixed monitoring sites with the measured ambient concentrations. Modeled concentrations of benzene and toluene were also compared with those measured at a Photochemical Assessment Monitoring Station in Copewood–Davis during corresponding time periods (17 matched days). To model personal exposures to the three air toxics, Lioy and colleagues used the Individual Based Exposure Modeling [IBEM]–MENTOR exposure model, which uses attributes and activities specific to the study subjects. Inputs included modeled or measured outdoor air toxic concentrations and activity data from diaries completed by the subjects. For model evaluation, the modeled estimates were compared with the measured personal exposure data. In addition, the investigators used the ratio of the measured personal concentrations to the modeled personal concentrations to estimate the percent of the total exposure (i.e., measured personal concentrations) that resulted from outdoor sources.

STATISTICAL ANALYSES

The investigators calculated means (\pm standard deviation [SD]) and medians for ambient and personal concentrations

of each pollutant measured and used scatter plots and calculated Spearman rank correlations to describe associations between the ambient and personal concentrations. They interpreted scatter plots in which the majority of data points were close to the 1:1 regression line to indicate that personal exposures were closely related to ambient pollution concentrations at a given location (e.g., MTBE in Figure 25 and hexane in Figure 26 in the Investigators' Report). Scatter plots in which the majority of points were above the 1:1 line were taken to indicate that personal exposures were influenced by more than outdoor exposures (e.g., *m*- & *p*-xylenes in Figure 22 in the Investigators' Report). Scatter plots in which the data were randomly distributed around the 1:1 line but in which more points were above the line were interpreted to indicate that personal exposure came from both indoor and outdoor sources (e.g., benzene in Figure 23 in the Investigators' Report). Lioy and colleagues also used mixed-effect models to test and estimate (1) differences in ambient and personal concentrations between the two neighborhoods and (2) associations between personal exposures and ambient air pollution. Analyses comparing Waterfront South and Copewood–Davis incorporated location, day of the week (i.e., weekdays versus weekend days) and season (i.e., summer versus winter) as predictors. A random effect for the day of the week accounted for correlations between the paired measurements taken simultaneously at each location. Interactions between factors were included in the model if they were significant.

A second set of analyses used a nested analysis of variance (hierarchical linear model) to examine associations between personal exposures and ambient concentrations, including interactions with season or day of the week. Four models were applied — (1) controlling location, season, and day of the week; (2) examining personal versus ambient associations for each location separately; (3) examining personal versus ambient associations for each season separately; and (4) examining personal versus ambient associations for weekdays and weekend days separately. All models treated sampling date, subject, and season within subject as random effects. Each model yielded a slope, which reflected the change of personal exposure per unit change of ambient concentration, and a *P* value. An *R*² was obtained by regressing the measured exposure on the modeled exposure, which could be used to interpret how much of the variation in personal exposure could be explained by each model.

For the study subgroup for which measurements of nicotine were available, the confounding effects of ETS exposure on associations between personal and ambient concentrations were evaluated by incorporating the measured nicotine concentrations into the regression analyses for all target compounds. Statistical Analysis System program version 9

was used to perform the data analysis. For the statistical analysis of pollutant concentrations below the method detection limit (MDL), the investigators used concentration values that were half of the MDL. When more than 60% of the data points were below the MDL, only graphical or descriptive analyses were performed.

Before performing these statistical analyses, quantile–quantile probability plots and Shapiro–Wilk tests were conducted to examine the normality of the concentrations for all target compounds. Except for ambient PM_{2.5}, measurements for all the target compounds showed many extreme right-skewed concentrations. The concentration data for the other pollutants were therefore log-transformed prior to using the mixed model.

For the spatial variation substudy, the investigators calculated means and standard deviations of the VOC and aldehyde concentrations from all the sampling sites during each sampling period. Because the sample size was small and the data were highly skewed, the investigators used the nonparametric Wilcoxon rank sum test to compare the spatial variability and the mean of the VOCs between Waterfront South and Copewood–Davis. To examine the spatial variability, then, differences in variation between the two locations were evaluated by applying the Wilcoxon rank sum test to the absolute deviations of each measurement from the medians for each day and location. Spearman correlation analysis was also performed among all species to provide information to identify potential sources of VOCs in the two neighborhoods.

HEALTH REVIEW COMMITTEE EVALUATION

GENERAL COMMENTS

In its independent review of the study, the HEI Health Review Committee found that Lioy and colleagues' study made an important contribution to the characterization of possible hot spots for air toxics. Although the participants had not been selected at random as a "proportional sample" of the entire community (as was done, for example, in the U.S. EPA's TEAM air toxics studies [Wallace et al. 1987, 1991]), the study provided valuable information about the concentrations of and relationships among ambient and personal exposures to PM and several air toxics in two neighborhoods in Camden, New Jersey.

KEY FINDINGS

The study generated many findings on personal and ambient concentrations of multiple pollutants; they are summarized (and compared with results from the RIOPA

study) in the Commentary Table, discussed in the Results section of the Commentary Appendix, and tabulated comprehensively in the Commentary Appendix Table.

The Committee considered the following to be the study's most important findings:

1. Based on ambient measurements at a fixed monitoring site, Waterfront South had consistently higher concentrations than Copewood–Davis of $PM_{2.5}$, toluene and the xylenes (known collectively as TEX), and PAHs ($P = 0.003$ for $PM_{2.5}$, $P < 0.0005$ for TEX, and $P < 0.0161$ for PAHs in the comparisons of summary data for the two areas). Although the investigators originally considered Copewood–Davis to be a control, or background, site for measurements in Waterfront South, they recognized during the study that some ambient pollutant concentrations — acetaldehyde and some VOCs (MTBE, benzene, hexane, chloroform, and carbon tetrachloride) — were similar in both neighborhoods or, in some cases, even higher in Copewood–Davis than in Waterfront South.
2. In comparison with other sites in New Jersey and other parts of the United States, *both* neighborhoods had elevated concentrations of $PM_{2.5}$, benzene and TEX (known collectively as BTEX), aldehydes, and MTBE (but not PAHs).
3. Trends in the data suggested that $PM_{2.5}$, TEX, and PAH concentrations were higher on weekdays than on weekend days.
4. Seasonal patterns were as predicted: concentrations of MTBE and formaldehyde were higher in summer, and concentrations of low-volatility PAHs were higher in winter.
5. Personal exposures to most pollutants were similar for residents in both neighborhoods. Higher personal exposures were measured in Waterfront South than in Copewood–Davis for $PM_{2.5}$ and toluene, which might reflect differences in ambient concentrations. For all measured pollutants, there was little impact of day-to-day (or temporal) variability in ambient concentrations on day-to-day (or temporal) personal exposures, suggesting that, even where ambient hot spots exist, the elevated ambient concentrations do not necessarily result in elevated (i.e., hot spot) personal exposures.
6. The spatial variation substudy, in which measurements of air toxics were made in three campaigns at multiple monitors across the two neighborhoods, revealed several locations in Waterfront South with elevated concentrations of BTEX and one of MTBE. In addition, compared with Copewood–Davis, more spatial variability was observed in Waterfront South on days with low wind speed.

INTERPRETATION OF FINDINGS

The Committee considered the following points to be critical to the evaluation and interpretation of Lioy and colleagues' findings:

Hot Spots: Alternative Definitions

The investigators' rationale for conducting the study was to determine whether the Waterfront South neighborhood of Camden, with its multiple industrial and mobile sources of pollution, was a hot spot for selected air toxics. In a publication that reported Lioy and colleagues' VOC data from the current study (Zhu et al. 2008), Zhu and colleagues defined a hot spot as "an area where the average concentrations of air pollutants are higher than those in the surrounding areas." The nearby Copewood–Davis neighborhood in Camden, with no endogenous industrial or other major point pollution sources, was therefore considered to be a suitable control site for the study. A priori, the investigators expected that residents of Copewood–Davis would be exposed to lower concentrations of pollutants than would residents of Waterfront South.

In view of Zhu and colleagues' (2008) definition and the finding that Waterfront South had consistently higher ambient concentrations than Copewood–Davis of several pollutants — $PM_{2.5}$, TEX, and PAHs — the Committee agreed with Lioy and colleagues that, for these pollutants, Waterfront South could be considered a hot spot in comparison with Copewood–Davis. However, the Committee noted that in defining a hot spot neither the investigators nor the original RFA specified either the spatial extent (e.g., "surrounding areas") or how much higher (using appropriate statistical tests) pollutant concentrations need to be for an area to constitute a hot spot in comparison with a control area. Future discussions of hot spots will need to consider these key issues.

As mentioned above, the investigators recognized during the study that both neighborhoods had similarly high concentrations of certain measured pollutants (and thus referred to Copewood–Davis as an "urban reference site" in the report). These findings underscored the difficulty in trying to define a hot spot in terms of a neighboring "control" community. Because they realized that Copewood–Davis had high concentrations of some pollutants, Lioy and colleagues also used an alternative definition of a hot spot in the report: a location with high concentrations compared with measurements made at other, more distant locations in New Jersey or other sites in the United States. By this criterion, the Committee agreed with the investigators that *both* Waterfront South and Copewood–Davis could be considered hot spots for $PM_{2.5}$, TEX, and MTBE.

Commentary Table. Summary Mean and Median Pollutant Concentrations ($\mu\text{g}/\text{m}^3$) for Waterfront South, Copewood–Davis, and Elizabeth, New Jersey^a

Pollutant	Waterfront South		Copewood–Davis		Elizabeth	
	Ambient	Personal	Ambient	Personal	Ambient	Personal
PM _{2.5}	31.3/29.7	61.9/53.3	25.3/24.0	84.1/49.1	20.4/18.2	44.8/37.4
VOCs						
MTBE	2.31/1.53	4.15/2.59	2.35/1.63	6.39/2.41	5.76/4.32	14.7/5.51
Hexane	226/2.16	215/2.81	266/1.87	539/3.16	NA	NA
Chloroform	0.19/0.13	0.96/0.49	0.20/0.14	1.96/0.52	0.30/0.17	2.02/0.85
Carbon tetrachloride	0.53/0.55	0.51/0.52	0.54/0.55	0.47/0.44	0.84/0.69	0.87/0.64
Benzene	2.18/1.20	2.90/2.12	2.86/1.26	4.73/2.81	1.44/1.22	2.80/1.76
Toluene	4.27/2.46	28.0/5.83	2.34/1.95	11.4/6.19	6.77/3.02	20.8/11.3
Ethylbenzene	0.55/0.39	1.41/0.89	0.43/0.33	1.57/0.98	1.34/0.99	2.91/1.40
<i>m</i> - & <i>p</i> -Xylenes	1.66/1.35	4.06/2.45	1.19/0.95	4.50/2.82	3.23/2.34	7.93/4.04
<i>o</i> -Xylene	0.59/0.44	1.25/0.77	0.41/0.34	1.42/0.93	1.70/0.94	3.07/1.56
Styrene	0.14/0.13	0.27/0.14	0.12/0.13	0.37/0.18	NA	NA
Aldehydes						
Formaldehyde	20.2/15.5	16.8/14.1	24.8/20.4	16.0/11.5	6.35/7.09	21.9/20.6
Acetaldehyde	12.5/6.1	15.5/11.4	14.6/8.3	16.6/15.4	8.88/7.86	17.3/15.1
Acrolein	0.8/0.2	1.2/0.5	0.6/0.2	1.4/0.4	0.89/0.39	0.74/0.26
Propionaldehyde	1.1/0.4	2.2/0.8	1.6/0.4	1.9/0.9	1.21/1.06	1.69/1.51
PAHs						
Naphthalene	13.1/9.78	64.4/22.3	11.7/7.62	133/32.9	NA	NA
Phenanthrene	15.5/9.14	24.9/17.5	12.4/7.47	23.4/17.5	NA	NA
Pyrene	1.47/0.99	1.28/0.94	1.23/0.91	1.56/0.88	NA	NA
Benzo[a]pyrene	0.36/0.18	0.38/0.05	0.21/0.12	0.42/0.05	NA	NA

^a Data for Waterfront South and Copewood–Davis are from the current study (Tables 12 and 15 in the Investigators' Report). Data for Elizabeth are from the RIOPA study (Weisel et al. 2005; Turpin et al. 2007).

NA indicates not assessed.

The Committee noted, however, that any comparisons of pollutant concentrations across studies need to be treated cautiously because studies differ in sampling periods, sampling and analysis methods, types of sampling sites, meteorologic conditions (temperature and wind speed and direction are particularly relevant), and averaging periods.

By applying the same criterion of comparison with previous measurements at other sites across the United States, the investigators also considered *both* Waterfront South and Copewood–Davis to be hot spots for benzene and the aldehydes formaldehyde and acetaldehyde. Although recognizing that all air toxics studies have had some difficulty in making accurate measurements of aldehyde concentrations, the Committee had some specific concerns

(described below, in the section on Reliability of Measurements of Individual Air Pollutants) about the accuracy and reliability of the concentrations of formaldehyde and acetaldehyde — as well as benzene — measured in the study.

Lioy and colleagues did not indicate whether either neighborhood was a hot spot for the four PAHs they focused on (naphthalene, phenanthrene, pyrene, and benzo[a]pyrene; see Table 24 in the Investigators' Report). Although ambient concentrations of these PAHs were higher in Waterfront South than in Copewood–Davis, the concentrations in Waterfront South were not higher than those measured at other U.S. urban sites, and therefore the Committee did not consider either location to be a PAH hot spot as defined by the alternative definition.

Sources of Air Pollutants in the Two Neighborhoods

Ambient Concentrations of Pollutants The investigators' expectation was that, because Waterfront South had multiple industrial sources, ambient concentrations of pollutants measured at the study's fixed monitoring site in each neighborhood would be higher in Waterfront South than in Copewood–Davis. Indeed, ambient concentrations of several pollutants (PM_{2.5}, TEX, and all the measured PAHs) were found to be higher in Waterfront South. Based on source emissions data collected by the NJDEP (2005), the authors attributed the high concentrations of TEX in Waterfront South to localized emissions sources, including paint applicators, several metal processing companies, and automobile repair shops. The Committee's view was that both industrial and mobile sources in the Waterfront South area were likely contributors to ambient concentrations of these compounds in this neighborhood. By contrast, the investigators attributed the high concentrations of TEX in Copewood–Davis — which were relatively homogeneously distributed in the spatial-saturation campaigns (discussed in more detail below) — primarily to regional sources, including those situated in Waterfront South and mobile sources on roads within and near the neighborhood. These conclusions were considered reasonable.

The study's finding that PM_{2.5} concentrations were higher in Waterfront South than in Copewood–Davis was surprising. Because PM mass typically has relatively little spatial heterogeneity (in part because it is a mixture derived from many emissions sources but also because it has a relatively long atmospheric lifetime), it was not clear why the neighborhoods differed in PM concentrations. PM composition was not evaluated in the study, making it difficult to know if Waterfront South experienced greater effects of mobile or industrial (or indeed other) sources than did Copewood–Davis. The Committee suggested that the trend of higher concentrations of the four measured PAHs in Waterfront South and the differences in weekday and weekend concentrations pointed specifically to emissions from weekday diesel-fueled truck traffic to and from industrial sites in Waterfront South.

The investigators reported that ambient concentrations of some VOCs (e.g., the aldehydes) were very similar in both neighborhoods and were in some cases actually higher in Copewood–Davis than in Waterfront South (e.g., benzene and MTBE). The Committee considered that the finding that not all the pollutants were higher in Waterfront South was difficult to interpret in terms of emissions sources. For the aldehydes, the investigators suggested that both neighborhoods were exposed to similar concentrations of emissions from diesel- and gasoline-powered

vehicles on roads adjoining the neighborhoods and from the urban pollution plume of Philadelphia (some 20 miles [32 km] distant). For benzene and MTBE, however, they interpreted the higher concentrations of these compounds in Copewood–Davis than in Waterfront South as reflecting mobile-source emissions (primarily gasoline-powered vehicles and evaporation from gasoline) carried downwind from the local main roads on most sampling days.

For several reasons, the Committee was not entirely convinced by the investigators' explanations of why Copewood–Davis might have had concentrations of some VOCs as high as or higher than those of Waterfront South, particularly because Copewood–Davis had no comparable level of industrial sources within the community. First, the meteorologic data did not indicate that Copewood–Davis was consistently downwind of Waterfront South. Second, the roads in or near Waterfront South carried substantially more traffic than the roads in or near Copewood–Davis, so the impact of traffic-related pollution would have been expected to be higher in Waterfront South. The Committee thought that alternative explanations need to be considered: Industrial sources in Waterfront South, for example, might have contributed little to overall VOC concentrations in Waterfront South, Copewood–Davis might have had unidentified point sources of VOCs, the concentrations of VOCs emitted by mobile sources on roads near Copewood–Davis might have been as high as or higher than the combined VOC concentrations from mobile and stationary sources in Waterfront South, or the fixed monitoring site in Copewood–Davis might have been more directly affected by mobile sources. Whatever the explanation, the Committee thought that the authors' findings that concentrations of many air toxics were as high as or higher in Copewood–Davis than those in Waterfront South appeared to contradict their original hypothesis that pollutant concentrations would be higher in Waterfront South as a result of emissions from industries in or near the neighborhood.

Another factor might also have contributed to the findings that Copewood–Davis had high concentrations of certain air toxics. The investigators characterized NJ-168, which runs north–south along the western edge of the neighborhood, as one of the major local roads in the Camden area (with a traffic volume of 25,000 cars/day). Although in setting up the study Liroy and colleagues considered it likely that mobile source emissions from NJ-168 would contribute to pollutant concentrations in Waterfront South, they apparently did not consider that these same emissions would also contribute to pollutant concentrations in Copewood–Davis. However, the results of the study bore out the likely important contribution of mobile source emissions from nearby major roads to exposures in

Copewood–Davis: 80% of the participants from Copewood–Davis lived within 500 m of NJ-168, and the fixed monitoring site in the neighborhood was located < 100 m from NJ-168 and < 100 m from a local road with moderate traffic (8000 vehicles/day). As the investigators noted, the detailed measurements they made in Copewood–Davis in the three campaigns of the spatial variation substudy indicated that the highest concentrations of BTEX and MTBE were found at sites located along NJ-168 (see Figures 43 through 45 in the Investigators' Report), in particular at a site with a high frequency of stopping and idling traffic.

As also became clear, the two nearly adjacent neighborhoods were both exposed to the pollutant plume from Philadelphia and were both affected by the same regional meteorologic conditions. The Committee noted that, in addition, the neighborhoods were similar in not being topographically complex. All these factors likely contributed to the similarities in the concentrations of ambient PM_{2.5} and many air toxics measured in the neighborhoods during the study.

The investigators' comparisons of weekday and weekend measurements were intended to provide some understanding of the contribution of mobile sources and particularly local industrial sources to variability of air pollutant concentrations. Higher concentrations were expected on weekdays, particularly in Waterfront South, with its industrial sources and the vehicles that serviced them. However, as the Commentary Appendix Table shows, the investigators' assumption held only for certain PAHs, such as phenanthrene and pyrene. This might point to diesel-powered trucks as their source, because truck traffic should be heavier on weekdays and diesel-fueled vehicles emit more PAHs than do gasoline-fueled vehicles. Day-of-the-week disparities were not found for ambient concentrations of certain pollutants, however, including PM_{2.5}, aldehydes, and BTEX. As a result, the relative contributions of local mobile and industrial sources to the measured concentrations of these pollutants remain uncertain.

The concentrations of several compounds were also expected to differ between summer and winter. VOCs such as TEX, for example, were expected to be higher in winter than in summer because they are depleted by photochemical reactions in the atmosphere at a much higher rate in summer. In fact, however, winter TEX concentrations were only higher in Copewood–Davis. For PAHs, the investigators concluded that seasonal variations differed for individual compounds, depending on the sources and the chemical reactivity and physical properties (particularly volatility) of each compound. Gas-phase concentrations of the lighter PAHs phenanthrene and pyrene were found to be higher in summer than in winter; the reverse was found for the heavier multi-ring PAHs naphthalene and benzo[*a*]pyrene. The

investigators attributed the winter increases to higher emissions from home heating, lower photochemical reactivity, and reduced dispersion; the Committee thought these interpretations were reasonable.

In summary, the Committee thought that the findings that the two neighborhoods had similarly high concentrations of numerous pollutants illustrated the difficulties of trying to define specific areas as hot spots — or controls for hot spots — without measuring actual pollutant concentrations in the areas. The fact that the neighborhoods were so close together geographically, within the same pollutant airshed, and not topographically complex made it likely, after all, that many similarities in ambient air pollutant concentrations would be observed.

Thus, choosing Copewood–Davis as the control area for the study was not ideal. Although Waterfront South could be considered a hot spot as expected, Copewood–Davis could also be considered a hot spot by some criteria, most likely as a result of the high concentrations of mobile-source emissions from roads in or near the area as well as of the emissions transported from adjacent neighborhoods, including Waterfront South.

Relationship of Personal Exposures to Ambient Concentrations

The study provided useful information on personal exposures in the two neighborhoods. Personal concentrations for most of the pollutants measured were higher than the corresponding ambient concentrations, suggesting contributions from sources other than outdoor (such as indoor or occupational).

Differences between personal and ambient concentrations were found when the data were analyzed by location, season, and day of the week. As Lioy and colleagues pointed out, some of these differences were consistent with temporal variation patterns in local ambient air concentrations, but some were not. These findings highlight an important issue that has also been identified in previous studies and is relevant to policy making and future health effects studies, namely the difficulty of relating personal exposures in a community to ambient concentrations measured at a central monitoring site in the community. The findings also support much earlier findings from the TEAM (e.g., Clayton et al. 1993; Özkaynak et al. 1996) and RIOPA (Weisel et al. 2005) studies about the dominance of the contribution to personal exposure of sources other than those captured by ambient monitoring.

Spatial Saturation Sampling

The saturation-sampling substudy, in which measurements of air toxics were made in three campaigns at multiple monitors in the two neighborhoods, provided new

and useful information about the spatial variability of air toxics concentrations over small distances. For example, the study revealed several locations with elevated concentrations of BTEX and one of MTBE in Waterfront South that were not detected using the fixed monitor. Thus, even within a possible hot spot neighborhood, spatial variability in ambient concentrations could be observed. This suggests that people in some locations in the neighborhood are likely to have been exposed to concentrations higher than those detected with the fixed monitor.

Comparisons with the RIOPA Study

The Committee thought it would be useful to compare the findings from the current report with those obtained by the same team of investigators using the same methods and sample-averaging periods that were used in the three-city RIOPA study in Elizabeth, New Jersey; Houston, Texas; and Los Angeles, California (Weisel et al. 2005) (see the Commentary Table). For reasons that are not clear, the average ambient $PM_{2.5}$ concentrations in Waterfront South and Copewood–Davis (31.3 and 25.3 $\mu g/m^3$, respectively) were approximately double those of the mean ambient $PM_{2.5}$ concentration for the three RIOPA cities (15.5 $\mu g/m^3$) and higher than the average for Elizabeth (20.4 $\mu g/m^3$).

On the other hand, BTEX and MTBE concentrations in Elizabeth in the RIOPA study were higher than those in Waterfront South and Copewood–Davis (Commentary Table). Aldehyde concentrations, however, particularly acetaldehyde and formaldehyde concentrations, appear to have been considerably higher in the current study than those in Elizabeth in the RIOPA study, even though both studies used similar measurement methods. As described below, the Committee had some additional concerns about the validity of the aldehyde measurements in the current study.

Reliability of the Measurements of Individual Air Pollutants

Although agreeing with the investigators that the methods used to measure concentrations of $PM_{2.5}$ and most of the VOCs likely produced accurate values, the Committee expressed caution about interpreting comparisons between concentrations of aldehydes and the VOCs benzene and MTBE from this study and those of other studies, as described below:

Ambient Aldehydes Concentrations of some of the aldehydes — especially formaldehyde and acetaldehyde — in the study were two to three times higher than those reported for other studies in the United States or Europe (Jurvelin et al. 2001; Sax et al. 2004) and even for the three cities of the RIOPA study (Weisel et al. 2005), which used similar sampling times and assay methods. Concentrations

were also much higher than those reported in studies in Brazil, where ethanol is widely used as a fuel; ethanol has been shown to lead to increased emissions of formaldehyde and acetaldehyde (reviewed in HEI Air Toxics Review Panel 2007).

Liroy and colleagues suggested that the differences in values among the studies might have resulted from the different methods used to capture aldehydes. The Committee considered this as a possible explanation; however, in the Commentary that accompanied the RIOPA I report (Weisel et al. 2005), HEI's Health Review Committee cautioned against comparing RIOPA's absolute aldehyde values (using a DNSH derivative of aldehydes) with those of other studies, given that the method had not been evaluated or compared externally with other methods (such as the 2,4-dinitrophenylhydrazine derivative method [Zhang et al. 2000; Herrington et al. 2005]).

For formaldehyde the investigators determined, after the samples were analyzed, that the liquid standards used to calibrate the gas chromatograph had partially polymerized and that the DNSH agent used in the cartridges had reacted with polymerized aldehydes. The investigators developed a correction factor for the concentration values based on a comparison of the original calibration curve with a new curve developed using an aldehyde monomer standard in the gas phase. The Committee was not convinced, however, that the correction factor was accurate, because the investigators had not measured the degree of polymerization in the field samples when they were originally collected and, as a result, could not gauge how extensively polymerization had proceeded in the time between sample collection and analysis. For this reason the Committee has expressed caution about the interpretation of the formaldehyde concentrations reported in the study.

VOCs For benzene the investigators reported that concentrations were detected in blank samples and varied considerably from batch to batch. The Committee thought that one possible explanation was that benzene had not been removed as a contaminant from the desorbing solvent (acetone–carbon disulfide) used to extract benzene from the sampler. It was not clear whether the investigators had taken steps to identify this as the source of the problem, and the investigators did not provide an explanation for their findings. In the light of this uncertainty, the Committee did not have great confidence in the absolute concentration values reported for benzene or in results indicating differences in benzene concentrations between Waterfront South and Copewood–Davis. However, the Committee was reassured that the benzene results might be reliable because of similarities in the observed pattern

of benzene concentrations with those of toluene, ethylbenzene, and xylene concentrations.

For MTBE some questions arose during the Quality Assurance audit of the data (Appendix A) about measurements made during the saturation-sampling campaign of August 2005. The investigators reported that just prior to making these measurements they had found an anomalous analyzer calibration with values three to four times higher than those of any other calibration. As the investigators explained in the section on Concentrations and Spatial Distribution of BTEX and MTBE in the Investigators' Report, they used this anomalous calibration for the values obtained in August 2005 in order to be able to compare values across different sites. However, for personal and ambient sampling, they decided to use the previous calibration factor instead. Thus, as the investigators indicated, comparisons of MTBE concentrations in the July and August 2005 campaigns cannot easily be made.

Nicotine as a Marker of ETS

The investigators measured personal nicotine in a subset of the study population (234 measurements) but did not include the measurements in the report, and hence the Committee did not evaluate the findings. It is not clear how representative these measurements were of the entire study population. Liroy and colleagues reported that nicotine concentrations in Copewood–Davis were more than double the average concentrations in Waterfront South ($0.61 \pm 0.17 \mu\text{g}/\text{m}^3$ compared with $0.26 \pm 0.16 \mu\text{g}/\text{m}^3$, $P < 0.01$), but they did not know why the two neighborhoods differed so much in this respect. They noted, and the Committee agreed, that because nicotine adheres to and is constantly re-emitted by indoor surfaces measured nicotine concentrations on a given day might not reflect actual ETS exposures on that day. For reasons that are not clear, however, the investigators set a nicotine concentration of $0.5 \mu\text{g}/\text{m}^3$ as the cutoff to indicate an exposed population, suggesting that only individuals in Copewood–Davis were exposed to significant concentrations of ETS. The Committee considered that individuals in both neighborhoods were likely exposed to ETS, because earlier studies have suggested a much lower cutoff ($0.1 \mu\text{g}/\text{m}^3$) for ETS exposures of nonsmokers in a smoking work environment (e.g., Jenkins and Counts 1999).

Because including nicotine as a confounder in models of associations between ambient and personal VOC concentrations did not affect the associations in Waterfront South but did in Copewood–Davis (they became less significant), Liroy and colleagues concluded that ETS, a likely source of BTEX, was not a major contributor to personal VOC concentrations in Waterfront South but was in Copewood–Davis. However,

given the apparently large exposure to mobile source emissions in both locations, the Committee believed that more analysis was needed before concluding that exposure to ETS is a significant contributor to the observed personal exposures to BTEX — and PAH compounds — in Copewood–Davis and not Waterfront South.

Performance of the MENTOR Modeling System

The Committee thought that the authors' modeling work was interesting but exploratory and not central to the main objective of the study, namely to characterize pollutant concentrations in an area that might be a hot spot for exposure to air toxics. The Committee noted that the results (see Figures 60 through 62 in the Investigators' Report) illustrated the concordances between the predictions made using two different dispersion models, ISC3ST and AERMOD, indicating that these models are internally consistent.

The performance of the MENTOR model was poorest for formaldehyde. The predictions significantly underestimated the measured ambient concentrations and could account for only 4% to 20% of the variability in the measured ambient values. Although average modeled toluene and benzene concentrations were within a factor of two of the measured ambient concentrations, the low correlation between modeled and measured values indicated that the current modeling framework has a limited ability to capture the actual variability of pollutant concentrations, though without more exhaustive model evaluation it is unclear what specific parameterizations or inputs were major contributors to the performance. Identifying why the correlations were so poor would be valuable, because these models are widely used.

The Committee concluded that the application of IBEM–MENTOR for estimating personal exposures was inaccurate, at least in part as a result of the poor performance of the air quality dispersion models in predicting concentrations of the three air toxics (benzene, toluene, and formaldehyde) from emissions data. Although the use of ambient measurements instead of emissions data to help estimate personal exposures resulted in significantly improved model estimates, the performance was similar to simply using the observed concentrations directly. The lack of information on emissions specific to the study area at the time of sampling was likely to be an important factor in the poor performance of the models, as discussed by the authors. Other factors also appeared to be involved, however, because the simulated and observed concentrations were not well correlated.

The Committee agreed with the investigators' general conclusion that, given the current state of the IBEM–MENTOR system, measured ambient concentrations are needed to

obtain more accurate exposure information. However, the Committee also noted that the overall model fit was weak, even with the measured concentrations as inputs in the model. At present, the personal exposure component of the system does not appear to significantly improve estimated exposures; the reasons for the lack of improved estimates need to be identified.

This application of the IBEM–MENTOR system highlights general issues that need to be addressed in developing and applying comprehensive exposure modeling approaches. Although the investigators ascribed much of the poor model performance to emissions errors, it is difficult to accept that this is the primary reason without performing significantly more model testing to develop confidence in each component of the modeling process.

Implications for Health Effects Studies

Consistent with findings from several previous studies, correlations between measurements of personal exposure and ambient pollutant concentrations at the current study's two fixed monitoring sites were not consistent between the two neighborhoods. This poses challenges for future health effects studies, because even when there are spatial differences in ambient concentrations of specific pollutants, these differences might not be generally reflected in personal exposures. The lack of correlation between ambient concentrations and personal exposures most likely resulted from contributions to personal exposures by sources other than ambient (e.g., indoor). The lack of correlation might also have resulted from population mobility, a factor not addressed in the current report: Even in sets of locations where there are differences in ambient concentrations (including well-defined hot spots) that individuals might live in, travel through, and work or study in, personal exposures reflect the time-weighted average of ambient concentrations in all such locations. Thus, to obtain more precise and detailed information about personal exposures, information such as time–activity diaries in various microenvironments for individual participants will be required.

SUMMARY AND CONCLUSIONS

HEI's Health Review Committee, which undertook an independent review of the study, thought the study — conducted in two nearby neighborhoods in Camden, New Jersey — had made an important contribution to the characterization of possible air toxics hot spots.

The investigators reported that one community, Waterfront South, had consistently higher ambient concentrations

than the other, Copewood–Davis, of PM_{2.5}, toluene, xylenes, and PAHs. Thus, by the investigators' original definition of a hot spot (i.e., having elevated concentrations compared with those of a nearby control or comparison area with fewer industrial sites), Waterfront South could be considered a hot spot for these pollutants. However, ambient concentrations in Copewood–Davis of several other pollutants — benzene, MTBE, chloroform, carbon tetrachloride, hexane, and acetaldehyde — were found to be as high as or higher than those in Waterfront South. The Committee generally considered the measurements of the air pollutants to have been accurate and reliable. However, they were concerned about the validity of the absolute concentrations of benzene, formaldehyde, and acetaldehyde, because they were much higher than those reported in other studies, and there appeared to be some specific problems with the measurement method for formaldehyde. The Committee concurred with the investigators' conclusion that, by their alternative definition of a hot spot (i.e., having elevated concentrations compared with those of other areas in New Jersey and across the United States), *both* neighborhoods could be considered hot spots for PM_{2.5}, benzene, toluene, xylenes, MBTE, and aldehydes. At the same time, the Committee cautioned that comparisons of pollutant concentrations across studies are difficult because studies differ in averaging periods, sampling and analysis methods, types of sampling sites, and meteorologic conditions (temperature and wind speed and direction are particularly relevant). The Committee also noted that, although ambient concentrations of PAHs were higher in Waterfront South than in Copewood–Davis, they were not higher than those measured at other urban sites in the United States. Although comparison of findings among different studies is challenging, neither neighborhood was a hot spot for PAHs by the alternative definition.

The Committee concluded that the study had provided useful information on personal exposures in the two neighborhoods. For most of the pollutants, measured personal concentrations were higher than the respective ambient concentrations measured at the study's two fixed monitoring sites, suggesting contributions from sources other than outdoor (i.e., indoor or occupational). In general, variations in personal concentrations did not correspond with variations in ambient concentrations. The Committee considered that this highlighted an important issue for policy making and future health effects studies of air toxics, namely the difficulty of relating personal exposures to ambient concentrations measured at a central monitoring site.

The Committee found that the investigators' saturation-sampling substudy, in which measurements of ambient air

toxics were made in three campaigns at several monitoring sites in each neighborhood, provided valuable information about the spatial variability at small scales of pollutant concentrations that could be compared with information from each neighborhood's fixed monitoring site. The results showed that, even within a possible hot spot, spatial variability in ambient concentrations can be found, suggesting that people in some locations within a neighborhood are likely to be exposed to much higher concentrations than those recorded at a fixed monitoring site in the same neighborhood. This finding again underscores the importance of individualized personal monitoring of pollutants. Liroy and colleagues also provided useful information that showed temporal (weekday versus weekend) and seasonal (summer versus winter) variability in individual pollutant concentrations.

The study was the first in a potential hot spot to attempt to compare measured concentrations of selected air toxics — benzene, toluene, and formaldehyde — with estimated concentrations derived from a model developed by the investigators. Correlations between modeled and measured concentrations were not very good; however, the Committee considered the modeling work to be interesting but exploratory and not central to the main objectives of the study.

The Committee concluded that in retrospect choosing Copewood–Davis as the control area for the study was not ideal. Although the neighborhood was free of industrial facilities, it was subject to high concentrations of mobile-source emissions from traffic on roads in or near the area as well as of emissions transported from adjacent areas (including Waterfront South). The two neighborhoods were also subject to the same regional meteorologic conditions and the same pollutant plume from Philadelphia (some 20 miles [32 km] away), and both were topographically simple.

In summary, the current study provided valuable information about ambient and personal concentrations of PM_{2.5} and a large number of air toxics and demonstrated elevated ambient concentrations (compared with other areas in New Jersey and across the United States) of some air toxics in both of these lower-socioeconomic-status neighbourhoods. At the same time, the findings illustrate the difficulties of defining an area a priori as a potential hot spot — or as a control location. The design of future exposure and health effects studies in hot spots will need to take multiple pollutant sources and meteorologic factors into consideration to achieve sufficient contrasts in pollutant concentrations between appropriately chosen hot spots and background locations.

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COMMENTARY APPENDIX

This Appendix summarizes the study's many findings on ambient and personal concentrations of multiple pollutants and indicates where in the Investigators' Report the original data can be found. The Commentary Appendix Table provides a compilation of mean and median concentrations for all pollutants measured, comparing summary, day-of-the-week, and seasonal ambient and personal pollutant data for Waterfront South and Copewood–Davis.

METEOROLOGIC DATA

The investigators reported that wind speed and direction data (Figure 11 in the Investigators' Report) for the study's four principal sampling periods (summer weekdays and weekends and winter weekdays and weekends) showed that southwest and south winds dominated in summer and that northwest and west winds dominated in winter.

During the spatial variation substudy, discussed in more detail below, temperature and wind speeds differed (they were lower and higher in winter, respectively, than in summer) across the substudy's three saturation-sampling campaigns (Table 6 in the Investigators' Report), but there was no rain or snowfall. The dominant wind directions during the saturation-sampling campaigns were from the west or southwest–northwest; the August sampling campaign also had winds from the east.

Note: In its review of the study, HEI's Health Review Committee agreed with the investigators' general interpretation and use of regional meteorologic data but noted that the data were obtained from the Philadelphia airport, eight miles from the study's sampling sites, and thus did not capture microscale variability in relation, for example, to specific sources or roadways.

AMBIENT AND PERSONAL POLLUTANT CONCENTRATIONS

Mean and median concentrations (in $\mu\text{g}/\text{m}^3$) of every ambient and personal pollutant measured in the study are shown in the Commentary Table, comparing summary, day-of-the-week (weekdays versus weekend days), and seasonal (summer versus winter) pollutant data for Waterfront South and Copewood–Davis. Additional information about pollutant concentrations reported in the study is outlined below:

PM_{2.5} and Nicotine

Data for ambient and personal PM are shown in Tables 12a, 13, 15a, and 16 in the Investigators' Report and are summarized below:

Ambient PM_{2.5} Mean (\pm SD) PM_{2.5} concentrations were $31.3 \pm 12.5 \mu\text{g}/\text{m}^3$ in Waterfront South and $25.3 \pm 11.9 \mu\text{g}/\text{m}^3$ in Copewood–Davis, which the investigators indicated was a significant difference. Based on average U.S. PM_{2.5} concentrations of 11–18 $\mu\text{g}/\text{m}^3$, the investigators calculated that ambient PM_{2.5} concentrations were 1.7 to 2.8 times higher than the national average in Waterfront South and 1.4 to 2.3 times higher in Copewood–Davis. In both neighborhoods there was little difference in average ambient PM_{2.5} concentrations when comparing weekdays with weekend days or summer with winter.

Personal PM_{2.5} Mean personal PM_{2.5} concentrations were higher than ambient concentrations in both Waterfront South and Copewood–Davis. The average was somewhat higher in Copewood–Davis (attributed to a few highly exposed individuals), but the median personal concentrations did not differ significantly between the neighborhoods.

No differences in personal PM_{2.5} concentrations were found between the two neighborhoods when comparing weekdays with weekend days or summer with winter. Nicotine concentrations, a marker of ETS, in Copewood–Davis were more than double those in Waterfront South (0.61 versus 0.26 $\mu\text{g}/\text{m}^3$).

Relationships Between Personal and Ambient PM_{2.5}

As shown in Figure 14 and Tables 18 and 19a through 19c in the Investigators' Report, scatter plots for personal versus ambient PM_{2.5} and mixed-effects models indicated that personal PM_{2.5} concentrations were higher than ambient concentrations in both neighborhoods. Excluding ETS-exposed participants in both neighborhoods from the models decreased mean personal exposure, but it was still higher than the ambient concentrations. The Spearman correlation coefficient (*R*) for all personal and ambient PM_{2.5} data from the two neighborhoods was 0.30. It was higher in summer (0.51), lower in winter (0.06), and higher on weekdays (0.46) than on weekend days (0.23).

VOCs

Personal and ambient VOC concentration data are shown Tables 12a, 14, and 15a and Figures 15 through 18 in the Investigators' Report and are summarized below. The percent of samples above the method detection limit was > 60% for all compounds except styrene (33%).

Ambient Average ambient concentrations of toluene, ethylbenzene, *m*- & *p*-xylenes, and *o*-xylene (known collectively as TEX) were higher in Waterfront South than in Copewood–Davis, especially in summer. They were higher in winter than in summer in both neighborhoods, but the

Commentary Appendix Table. Mean and Median Concentrations ($\mu\text{g}/\text{m}^3$) for All Pollutants Measured, Comparing Summary, Day-of-the-Week, and Seasonal Ambient and Personal Data for Waterfront South and Copewood-Davis

Compounds	Ambient					Personal				
	Summary	Weekday	Weekend	Summer	Winter	Summary	Weekday	Weekend	Summer	Winter
Waterfront South										
VOCs										
Methyl <i>tert</i> -butyl ether (MTBE)	2.31/1.53	2.11/1.36	2.57/1.68	2.48/1.95	2.09/1.44	4.15/2.59	3.49/2.32	4.91/2.74	4.74/3.10	3.51/2.25
Hexane	226/2.16	5.91/2.17	496/1.83	435/3.87	1.46/0.84	215/2.81	3.54/2.53	428/3.12	454/3.27	3.08/2.03
Chloroform	0.19/0.13	0.17/0.14	0.21/0.13	0.28/0.18	0.11/0.10	0.96/0.49	1.06/0.47	0.86/0.51	1.08/0.55	0.85/0.46
Carbon tetrachloride	0.53/0.55	0.53/0.54	0.54/0.57	0.57/0.58	0.50/0.45	0.51/0.52	0.51/0.53	0.51/0.52	0.56/0.55	0.46/0.46
Benzene	2.18/1.20	2.39/1.28	1.92/1.15	2.77/1.25	1.49/1.17	2.90/2.12	3.06/2.29	2.73/1.99	3.02/1.79	2.79/2.47
Toluene	4.27/2.46	4.95/3.36	3.38/1.97	4.08/2.63	4.53/2.37	28.0/5.83	47.0/6.55	6.23/4.97	8.62/5.39	49.1/6.45
Ethyl benzene	0.55/0.39	0.58/0.47	0.50/0.36	0.49/0.39	0.63/0.40	1.41/0.89	1.76/1.05	1.01/0.75	1.55/0.78	1.26/0.96
<i>m</i> - & <i>p</i> -Xylenes	1.66/1.35	1.87/1.60	1.40/1.13	1.42/1.24	2.0/1.41	4.06/2.45	5.13/3.0	2.83/2.14	4.50/2.15	3.58/2.78
<i>o</i> -Xylene	0.59/0.44	0.66/0.49	0.50/0.42	0.50/0.43	0.71/0.47	1.25/0.77	1.55/0.95	0.90/0.72	1.38/0.74	1.10/0.89
Styrene	0.14/0.13	0.15/0.13	0.14/0.13	0.14/0.13	0.15/0.14	0.27/0.14	0.31/0.18	0.24/0.13	0.18/0.13	0.37/0.32
Aldehydes										
Formaldehyde	20.2/15.5	20.2/15.6	20.2/13.6	20.7/15.5	19.6/14.4	16.8/14.1	16.7/13.7	16.8/14.3	20.7/16.3	11.7/12.0
Acetaldehyde	12.5/6.1	10.5/3.3	15.1/8.3	8.7/1.2	17.7/11.2	15.5/11.4	12.9/9.7	18.6/12.7	12.9/7.8	18.9/16.2
Acrolein	0.8/0.2	0.8/0.2	0.7/0.2	0.8/0.1	0.6/0.2	1.2/0.5	1.3/0.6	1.1/0.4	1.3/0.5	1.0/0.5
Propionaldehyde	1.1/0.4	0.9/0.4	1.4/0.4	0.9/0.4	1.4/0.4	2.2/0.8	2.9/0.8	1.4/0.6	1.5/0.4	3.1/1.5
Particulate matter										
PM _{2.5}	31.3/29.7	33.3/23.5	29.4/26.1	31.4/24.7	31.2/29.7	61.9/53.3	67.4/54.7	56.4/50.3	58.3/47.7	66.6/56.4
PAHs										
Naphthalene	13.1/9.78	14.1/10.6	12.0/7.8	5.23/3.90	20.3/16.0	64.4/22.3	58.8/19.4	70.2/26.5	80.2/23.3	46.9/20.0
Acenaphthylene	3.37/1.78	4.41/2.04	2.17/1.04	1.98/0.46	4.63/3.02	5.34/2.49	5.08/2.23	5.61/3.10	4.39/2.02	6.39/3.57
Acenaphthene	5.78/3.03	7.33/3.19	3.99/2.93	8.09/2.97	3.69/3.04	12.3/7.33	12.2/6.77	12.4/8.05	17.6/10.5	6.47/5.40
Fluorene	11.8/8.38	13.9/9.5	9.52/5.71	18.1/11.3	6.18/4.75	39.8/26.1	40.1/21.6	39.4/28.9	59.3/37.6	18.1/15.9
Phenanthrene	15.5/9.14	17.5/9.6	13.3/7.63	26.9/23.2	5.23/4.32	24.9/17.5	22.7/15.2	27.2/20.4	36.2/31.6	12.4/10.1
Anthracene	1.06/0.96	1.16/0.96	0.95/0.92	1.31/0.97	0.84/0.96	1.92/1.30	1.93/1.26	1.92/1.32	2.64/2.02	1.12/0.96
Fluoranthene	2.44/1.91	3.06/2.01	1.73/1.41	3.76/2.56	1.25/1.11	1.97/1.75	2.21/1.86	1.72/1.66	2.57/1.97	1.32/1.25
Pyrene	1.47/0.99	1.94/1.18	0.94/0.81	1.91/1.22	1.08/0.74	1.28/0.94	1.43/0.99	1.11/0.93	1.61/1.25	0.90/0.60
Benzo[<i>a</i>]anthracene	0.38/0.15	0.46/0.18	0.29/0.15	0.28/0.15	0.48/0.21	0.36/0.15	0.47/0.15	0.25/0.15	0.46/0.15	0.25/0.15
Chrysene	0.61/0.54	0.71/0.54	0.48/0.45	0.47/0.43	0.73/0.54	0.58/0.54	0.70/0.54	0.46/0.48	0.63/0.53	0.52/0.54
Benzo[<i>b</i>]fluoranthene	0.37/0.11	0.43/0.12	0.31/0.10	0.14/0.05	0.58/0.38	0.40/0.05	0.49/0.09	0.29/0.05	0.48/0.05	0.30/0.25
Benzo[<i>k</i>]fluoranthene	0.24/0.09	0.25/0.13	0.22/0.08	0.08/0.05	0.38/0.21	0.18/0.05	0.17/0.05	0.19/0.05	0.15/0.05	0.22/0.09
Benzo[<i>a</i>]pyrene	0.36/0.18	0.44/0.20	0.26/0.08	0.12/0.05	0.57/0.40	0.38/0.05	0.32/0.05	0.44/0.05	0.15/0.05	0.64/0.33
Indeno[1,2,3- <i>cd</i>]pyrene	0.23/0.05	0.28/0.05	0.17/0.05	0.11/0.05	0.33/0.09	0.23/0.05	0.33/0.05	0.13/0.05	0.18/0.05	0.29/0.05
Benzo[<i>ghi</i>]perylene	0.11/0.05	0.12/0.05	0.09/0.05	0.06/0.05	0.15/0.05	0.23/0.05	0.23/0.05	0.23/0.05	0.23/0.05	0.24/0.05
Dibenzo[<i>a,h</i>]anthracene	0.21/0.05	0.22/0.05	0.19/0.05	0.10/0.05	0.30/0.13	0.32/0.05	0.35/0.05	0.30/0.05	0.18/0.05	0.48/0.06

Table continues next page

Commentary Appendix Table (Continued). Mean and Median Concentrations ($\mu\text{g}/\text{m}^3$) for All Pollutants Measured, Comparing Summary, Day-of-the-Week, and Seasonal Ambient and Personal Data for Waterfront South and Copewood–Davis

Compounds	Ambient					Personal				
	Summary	Weekday	Weekend	Summer	Winter	Summary	Weekday	Weekend	Summer	Winter
Copewood–Davis										
VOCs										
Methyl <i>tert</i> -butyl ether (MTBE)	2.35/1.63	2.26/1.51	2.47/1.70	2.63/1.74	1.98/1.55	6.39/2.41	5.38/2.44	7.42/2.36	9.55/2.39	2.94/2.44
Hexane	266/1.87	3.84/1.74	583/1.99	524/3.34	1.31/0.98	539/3.16	4.50/3.02	1066/3.25	1245/3.95	3.36/2.23
Chloroform	0.20/0.14	0.16/0.13	0.24/0.15	0.30/0.17	0.13/0.11	1.96/0.52	1.85/0.55	2.07/0.51	2.84/0.52	1.42/0.48
Carbon tetrachloride	0.54/0.55	0.54/0.57	0.53/0.54	0.58/0.59	0.51/0.50	0.47/0.44	0.48/0.45	0.46/0.43	0.51/0.51	0.44/0.39
Benzene	2.86/1.26	2.81/1.24	2.92/1.27	3.98/1.32	1.55/1.09	4.73/2.81	4.67/2.70	4.80/3.04	6.83/3.36	2.77/2.30
Toluene	2.34/1.95	2.76/2.13	2.13/1.72	1.79/1.73	3.40/2.12	11.4/6.19	13.6/6.18	9.12/6.19	9.15/6.41	13.8/5.75
Ethyl benzene	0.43/0.33	0.42/0.33	0.43/0.33	0.34/0.28	0.55/0.33	1.57/0.98	1.56/1.0	1.59/0.95	1.40/0.99	1.76/0.97
<i>m</i> - & <i>p</i> -Xylenes	1.19/0.95	1.21/1.02	1.17/0.93	0.96/0.92	1.51/1.24	4.50/2.82	4.38/2.87	4.62/2.75	4.15/2.87	4.88/2.82
<i>o</i> -Xylene	0.41/0.34	0.43/0.36	0.39/0.30	0.35/0.33	0.49/0.40	1.42/0.93	1.41/0.93	1.44/0.93	1.38/1.0	1.47/0.86
Styrene	0.12/0.13	0.11/0.13	0.14/0.13	0.11/0.08	0.14/0.13	0.37/0.18	0.39/0.19	0.34/0.15	0.26/0.13	0.48/0.32
Aldehydes										
Formaldehyde	24.8/20.4	22.9/16.3	27.4/21.2	27.7/23.3	21.3/15.3	16.0/11.5	16.3/11.8	15.7/10.2	21.4/16.6	10.6/9.6
Acetaldehyde	14.6/8.3	10.2/5.1	20.3/13.9	11.6/4.3	18.4/10.4	16.6/15.4	14.7/14.4	18.6/16.6	12.5/8.9	20.7/18.1
Acrolein	0.6/0.2	0.7/0.2	0.6/0.2	0.6/0.1	0.8/0.2	1.4/0.4	1.7/0.6	1.1/0.2	1.8/0.7	0.9/0.2
Propionaldehyde	1.6/0.4	1.4/0.4	1.8/0.4	1.6/0.3	1.6/0.4	1.9/0.9	2.0/0.7	1.8/1.5	2.1/1.3	1.8/0.8
Particulate matter										
PM _{2.5}	25.3/24.0	26.7/24.9	23.8/19.8	26.1/24.8	24.6/22.7	84.1/49.1	81.1/50.2	87.0/47.8	92.2/48.5	76.6/49.5
PAHs										
Naphthalene	11.7/7.62	11.7/8.54	11.7/6.21	4.81/3.95	17.6/14.4	133/32.9	194/31.1	66.9/36.1	206/35.8	63.7/29.9
Acenaphthylene	2.40/1.20	2.60/1.74	2.15/0.53	1.49/0.23	3.19/2.80	4.71/2.63	4.76/2.71	4.66/2.57	4.04/1.80	5.36/3.29
Acenaphthene	3.75/2.71	4.14/2.71	3.24/2.67	4.64/3.16	2.98/2.59	14.1/9.08	14.8/10.1	13.4/7.49	15.6/9.26	12.7/7.17
Fluorene	9.25/6.12	9.77/8.30	8.59/4.98	13.8/13.0	5.38/4.14	56.6/28.2	54.0/29.9	59.4/27.2	69.8/33.2	44.0/24.8
Phenanthrene	12.4/7.47	12.9/7.95	11.6/6.88	20.6/19.3	5.26/5.29	23.4/17.5	22.9/17.0	24.1/18.10	33.5/25.1	13.9/14.5
Anthracene	1.03/0.96	1.14/0.96	0.89/0.94	1.28/0.96	0.82/0.89	2.38/1.64	2.20/1.65	2.56/1.40	3.22/2.27	1.58/1.13
Fluoranthene	2.28/1.79	2.46/1.91	2.05/1.71	3.65/2.95	1.11/1.0	2.32/1.64	1.97/1.66	2.70/1.44	3.31/1.91	1.38/1.13
Pyrene	1.23/0.91	1.29/0.87	1.14/0.94	1.60/1.28	0.91/0.51	1.56/0.88	1.50/1.00	1.63/0.68	2.17/1.44	0.99/0.56
Benzo[<i>a</i>]anthracene	0.22/0.15	0.23/0.15	0.21/0.15	0.24/0.15	0.21/0.15	0.58/0.15	0.34/0.15	0.84/0.15	0.95/0.15	0.23/0.15
Chrysene	0.46/0.42	0.46/0.45	0.45/0.39	0.42/0.35	0.49/0.49	0.75/0.54	0.81/0.54	0.68/0.54	0.96/0.54	0.55/0.54
Benzo[<i>b</i>]fluoranthene	0.26/0.08	0.29/0.16	0.21/0.05	0.14/0.05	0.35/0.23	0.38/0.05	0.33/0.05	0.44/0.05	0.49/0.05	0.28/0.14
Benzo[<i>k</i>]fluoranthene	0.15/0.07	0.18/0.10	0.11/0.05	0.07/0.05	0.21/0.12	0.29/0.05	0.35/0.05	0.22/0.05	0.37/0.05	0.21/0.09
Benzo[<i>a</i>]pyrene	0.21/0.12	0.26/0.15	0.14/0.05	0.08/0.05	0.32/0.25	0.42/0.05	0.50/0.05	0.33/0.05	0.44/0.05	0.40/0.20
Indeno[1,2,3- <i>cd</i>]pyrene	0.14/0.05	0.17/0.05	0.11/0.05	0.05/0.05	0.22/0.06	0.43/0.05	0.49/0.05	0.37/0.05	0.59/0.05	0.28/0.05
Benzo[<i>ghi</i>]perylene	0.06/0.05	0.06/0.05	0.06/0.05	0.05/0.05	0.07/0.05	0.68/0.05	0.42/0.05	0.96/0.05	1.01/0.05	0.37/0.05
Dibenzo[<i>a,h</i>]anthracene	0.12/0.05	0.12/0.05	0.11/0.05	0.05/0.05	0.17/0.08	0.47/0.05	0.34/0.05	0.60/0.05	0.56/0.05	0.38/0.05

difference was statistically significant only in Copewood–Davis. Average concentrations of benzene, MTBE, chloroform, and carbon tetrachloride and median concentrations of hexane did not differ between the two neighborhoods. For both neighborhoods, VOC concentrations were not significantly different on weekdays compared with weekend days, although they were higher on weekdays in Waterfront South.

Comparisons with Other Sites in New Jersey Lioy and colleagues compared their ambient data from Waterfront South and Copewood–Davis with data from various New Jersey Department of Environmental Protection (NJDEP) monitoring sites for time periods in 2004–2006 (Figures 19 through 21 and Appendix B in the Investigators' Report) that paralleled the sampling seasons in their own study. The NJDEP sites were located in Elizabeth, New Brunswick, Camden (at a site co-located with the investigators' fixed monitor in Copewood–Davis), and Chester (which the investigators considered to be the background for air pollution in the state). All VOC concentrations (except for chloroform in winter) were higher in Waterfront South than in Chester. Ambient VOC concentrations in Waterfront South were similar to those in Elizabeth and Camden. For most VOCs, Copewood–Davis had higher concentrations than Chester did but lower than those of Elizabeth. Copewood–Davis had significantly higher median benzene concentrations in summer and higher MTBE concentrations than those of New Brunswick. VOC concentrations in Copewood–Davis, although measured at a site co-located with the NJDEP site, were lower than those measured at the NJDEP site (Appendix B in the Investigators' Report).

Personal Mean and median personal benzene concentrations were higher in Copewood–Davis than in Waterfront South, both by day of the week and by season, except in winter (Tables 16 and 17 in the Investigators' Report). Personal MBTE and TEX concentrations did not differ between the two neighborhoods.

Relationships Between Personal and Ambient VOCs

As shown in Figures 22 through 26 and Table 18 in the Investigators' Report, VOCs in Waterfront South generally held close to the 1:1 regression line, and VOCs in Copewood–Davis tended to scatter above the 1:1 line. The *R*'s for ambient and personal concentrations of all target compounds in Waterfront South were higher than or similar to those in Copewood–Davis: Fairly good correlations between personal exposures and ambient concentrations were found for MTBE (0.82 in Waterfront South, 0.65 in Copewood–Davis), hexane (0.68, 0.66), and carbon tetrachloride (0.73, 0.85).

Weaker correlations were found for benzene, toluene, ethylbenzene, and xylenes (known collectively as BTEX), ranging from 0.47 to 0.56 in Waterfront South and 0.26 to 0.45 in Copewood–Davis. Correlations for MTBE, hexane, and benzene by season were relatively higher in summer than in winter for both neighborhoods. The *R*'s for MTBE in summer, for example, were 0.83 in Waterfront South and 0.79 in Copewood–Davis; they decreased in winter to 0.77 and 0.43, respectively. When using the mixed-effect models, personal exposures to MTBE, hexane, benzene, toluene, and *o*-xylene were found to be closely associated with ambient concentrations for both neighborhoods (Table 19a).

When nicotine measurements were included in the analysis of personal and ambient associations, no change was found in regression coefficients or *P* values in Waterfront South, but the *P* values became less significant in Copewood–Davis, changing from 0.0002 to 0.0458 for *o*-xylene, for example, and from 0.0010 to 0.0877 for toluene. The regression coefficient of the ambient VOC concentrations also decreased, from 0.56 to 0.44 for benzene, 0.39 to 0.25 for toluene, and 0.51 to 0.38 for *o*-xylene.

Aldehydes

Data for personal and ambient aldehyde concentrations are shown in Tables 12a, 13, 14, 15a, 16, and 17 and Figures 27 and 28 in the Investigators' Report and are summarized below:

Ambient From descriptive analyses, average formaldehyde and acetaldehyde concentrations were similar in Waterfront South and Copewood–Davis, ranging from 12 to 25 $\mu\text{g}/\text{m}^3$ (i.e., 10 to 20 times higher than propionaldehyde and acrolein concentrations in both neighborhoods). When using the mixed-effect models, formaldehyde concentrations were actually higher in Copewood–Davis (24.8 $\mu\text{g}/\text{m}^3$) than in Waterfront South (20.2 $\mu\text{g}/\text{m}^3$), but acetaldehyde concentrations remained similar. However, in the spatial variation substudy discussed in the Investigators' Report, all aldehydes — except for acrolein in the substudy's July saturation-sampling campaign — were found either at similar concentrations in both neighborhoods or at slightly higher concentrations in Waterfront South than in Copewood–Davis (Table 20 in the Investigators' Report).

To assess the effects of season and day of the week, Lioy and colleagues combined data from the two neighborhoods (indicating that “stratified data by location did not converge”). Ambient mean acetaldehyde concentrations were lower in summer (10.3 $\mu\text{g}/\text{m}^3$) than in winter (18.3 $\mu\text{g}/\text{m}^3$), but formaldehyde concentrations did not differ significantly

by season. Weekdays versus weekends did not significantly affect either acetaldehyde or formaldehyde concentrations.

Personal Like ambient concentrations of acetaldehyde and formaldehyde, mean personal concentrations were in the same range — 15 to 17 $\mu\text{g}/\text{m}^3$ — in both Waterfront South and Copewood–Davis (i.e., one to two orders of magnitude higher than concentrations of acrolein and propionaldehyde). When using the mixed-effect models for combined data from the two neighborhoods, personal acetaldehyde exposures were higher in Copewood–Davis (16.6 $\mu\text{g}/\text{m}^3$) than in Waterfront South (15.5 $\mu\text{g}/\text{m}^3$). Personal acetaldehyde exposures were higher in winter (19.0 $\mu\text{g}/\text{m}^3$) than in summer (12.5 $\mu\text{g}/\text{m}^3$) for the combined data; personal formaldehyde exposures were higher in summer (19.8 $\mu\text{g}/\text{m}^3$) than in winter (10.9 $\mu\text{g}/\text{m}^3$). Similar results were observed when comparing the data by season within each neighborhood (Table 16 in the Investigators' Report). Weekdays versus weekends did not affect acetaldehyde or formaldehyde concentrations.

When using the mixed-effect models after excluding subjects with ETS exposure (nicotine level > 0.5 $\mu\text{g}/\text{m}^3$), differences in personal exposures to acetaldehyde in the two neighborhoods were not significant, meaning that location changed from being a highly significant factor to a marginal factor. In this analysis, season was a statistically significant factor — acetaldehyde exposures were higher in winter (18.8 $\mu\text{g}/\text{m}^3$) than in summer (12.0 $\mu\text{g}/\text{m}^3$), and formaldehyde exposures were higher in summer (19.6 $\mu\text{g}/\text{m}^3$) than in winter (10.9 $\mu\text{g}/\text{m}^3$). Weekdays versus weekends did not significantly affect acetaldehyde or formaldehyde concentrations.

Relationships Between Personal and Ambient

Aldehydes Scatter plots of personal versus ambient formaldehyde concentrations (Figure 29 in the Investigators' Report) indicated that the ambient concentrations were significantly higher than the personal concentrations; this held true even after exclusion of ETS-exposed study subjects in both neighborhoods. Personal acetaldehyde concentrations (Figure 30 in the Investigators' Report) were moderately but statistically significantly higher than ambient acetaldehyde concentrations (15.0 versus 13.9 $\mu\text{g}/\text{m}^3$). Exclusion of the ETS-exposed subjects did not change the relationships between personal and ambient concentrations of formaldehyde or acetaldehyde.

The R 's and mixed-effect models (Tables 18 and 19a through 19c in the Investigators' Report) indicated that personal exposures to formaldehyde and acetaldehyde were significantly correlated with ambient concentrations.

When using combined data from the two neighborhoods, the R 's were 0.55 and 0.68 for formaldehyde and acetaldehyde, respectively (Table 18 in the Investigators' Report). When using the mixed-effect models (Tables 19a through 19c in the Investigators' Report), most of the coefficients of determination (R^2) for associations between personal and ambient aldehyde concentrations were close to or larger than 0.8.

No common associations could be found for variations in R with (a) ETS exposure (0.51 for nonsmoking subjects versus 0.65 for smoking subjects for formaldehyde; 0.73 for nonsmoking subjects versus 0.51 for smoking subjects for acetaldehyde); (b) location (0.48 for Waterfront South versus 0.61 for Copewood–Davis for formaldehyde; 0.68 for Waterfront South versus 0.68 for Copewood–Davis for acetaldehyde); (c) season (0.68 for summer versus 0.42 for winter for formaldehyde; 0.71 for summer versus 0.62 for winter for acetaldehyde); or (d) day of the week (0.56 for weekdays and 0.54 for weekend days for formaldehyde; 0.70 for weekdays and 0.66 for weekend days for acetaldehyde). However, in all cases the correlation coefficients were statistically significant. The associations between personal exposures and ambient concentration were stronger in summer than in winter.

PAHs

The investigators presented data for personal concentrations (Tables 12 through 14 in the Investigators' Report) and ambient concentrations (Tables 15 through 17 in the Investigators' Report) of 16 PAHs. They focused their comparative analyses on naphthalene, phenanthrene, pyrene, and benzo[a]pyrene.

Ambient Ambient concentrations of all 16 PAHs were significantly higher (36% to 86%) in Waterfront South than in Copewood–Davis. In Waterfront South the differences between weekday and weekend concentrations of pyrene and benzo[a]pyrene, but not of phenanthrene or naphthalene, were significant. In Copewood–Davis, weekday and weekend concentrations of these four PAHs did not differ. For naphthalene and benzo[a]pyrene, mean ambient concentrations were about four to five times higher in winter than in summer (Table 14 and Figures 35 through 38 in the Investigators' Report).

Personal A wide range of personal PAH concentrations was detected (Table 15b in the Investigators' Report), with several extremely high concentrations of some compounds in some participants. The investigators suggested that some of these exposures might have been occupational. The median personal naphthalene concentration in

Copewood–Davis (32.9 ng/m³) was 48% higher than that in Waterfront South (22.3 ng/m³) (Table 15b in the Investigators' Report); median personal phenanthrene and pyrene concentrations were similar in the two neighborhoods. Weekday and weekend personal PAH concentrations did not differ in either of the two neighborhoods. Personal exposures differed by season for all the PAHs except naphthalene (Table 17 in the Investigators' Report). In both neighborhoods, phenanthrene and pyrene were higher in summer than in winter, and benzo[a]pyrene was higher in winter.

Note: The HEI Review Committee commented that ambient PAH concentrations were significantly higher in Waterfront South than in Copewood–Davis but, as indicated by the investigators, were not high compared with those reported in studies of other urban areas. The Committee also noted that the differences in ambient PAH concentrations between the two neighborhoods were generally not reflected in the patterns of personal PAH concentrations in the two neighborhoods.

Relationships Between Personal and Ambient PAHs

Generally, the scatter plots for data from Waterfront South and Copewood–Davis followed similar patterns. Phenanthrene and pyrene stayed close to the 1:1 line (Figures 40 and 41, respectively, in the Investigators' Report), and naphthalene and benzo[a]pyrene tended to scatter above the 1:1 line (Figures 39 and 42, respectively, in the Investigators' Report). Both the *R*'s and the mixed-effect models yielded relatively poor correlations (Tables 18 and 19a through 19c in the Investigators' Report) for the associations between personal and ambient PAHs for all the target compounds in both neighborhoods. No statistically significant differences were found for associations between personal and ambient PAHs by season or day of the week (Tables 19b and 19c in the Investigators' Report).

After including nicotine concentrations in the model, associations between personal and ambient concentrations in Copewood–Davis became less significant (the *P* value increased from 0.0093 to 0.0263 for phenanthrene and from 0.0104 to 0.4552 for pyrene); the *P* value in Waterfront South was not affected, indicating that there was no significant confounding effect of ETS on personal–ambient PAH associations.

SPATIAL VARIATION SUBSTUDY

Spatial variations in VOC concentrations were investigated in a substudy consisting of three saturation-sampling campaigns. The observed degrees of variability varied by compound, by day of the week, and by location (Table 20 and Figures 43 through 45 [for toluene, MTBE, and benzene only] in the Investigators' Report). In Waterfront

South, large variations in spatial distribution were found in concentrations of ethylbenzene, xylenes, MTBE, and especially toluene (Table 20 in the Investigators' Report); spatial distributions were relatively uniform in Copewood–Davis.

Mean benzene concentrations and spatial variability (which was relatively low) were found to be similar in the two neighborhoods. However, the mean concentrations and spatial variability of BTEX as a whole were higher in Waterfront South than in Copewood–Davis on sampling days with low wind speed, indicating, according to the investigators, the impact on Waterfront South of BTEX emission sources in the neighborhood. **Note:** The Committee agreed with the investigators' supposition but pointed out that it was based on limited data.

In the July and August sampling campaigns, mean concentrations and spatial variability of TEX and MTBE were approximately two times higher than those in the December campaign (Table 20 in the Investigators' Report). **Note:** The investigators attributed the difference to wind speed. The Committee thought this was one possible explanation but considered volatility to be another potentially important factor. In addition, the Committee also noted that, if overall concentrations were lower in winter, there would also be less spatial variability.

Spatial distributions of chloroform and carbon tetrachloride concentrations were comparable between Waterfront South and Copewood–Davis and, according to the investigators, were close to regional background concentrations in North America.

Concentrations of formaldehyde, acetaldehyde, and acrolein measured at many of the sampling sites in both neighborhoods were higher than ambient concentrations measured in other urban areas (Table 20 and Figures 46 through 49 in the Investigators' Report).

MODELING RESULTS

The ISCST3 and AERMOD dispersion models used by the investigators performed similarly in predicting ambient concentrations of benzene, toluene, and formaldehyde (see Figure 52 in the Investigators' Report for benzene and toluene).

Predicted toluene concentrations were mostly within a factor of two of actual measurements from a U.S. EPA photochemical assessment monitoring station in Copewood–Davis and the investigators' own measurements in the two neighborhoods. The model predictions for benzene differed across days (Figures 53 and 54 in the Investigators' Report) and led the investigators to remove 2004 summer days with unusually high benzene concentrations from their subsequent analyses.

Analysis of scatter plots for the ambient concentrations predicted by the two models versus the corresponding measured concentrations indicated that the slopes for benzene and toluene were between 0.92 and 1; the slopes for formaldehyde were not significant (Table 23 in the Investigators' Report). The investigators also found that the 95% confidence interval of the slope for toluene for both models was smaller than that for benzene. For benzene and toluene, the model predictions underestimated high measured concentrations and overestimated low measured concentrations. The models underestimated formaldehyde concentrations across all measurements.

The R^2 's were poor for toluene (0.22 and 0.23) and benzene (0.08 and 0.07) and even poorer for formaldehyde, leading the investigators to conclude that the models did not capture the variability of their ambient benzene and formaldehyde measurements but made better predictions for toluene.

Scatter plots of predicted versus measured personal exposure concentrations of the three compounds showed

that the MENTOR model performed better when measured, rather than modeled, ambient concentrations were used as inputs (particularly for benzene and formaldehyde). However, the R^2 's were generally very low — 0.27 for benzene, 0.1 for toluene, and 0.23 for formaldehyde using the measured ambient concentrations and < 0.1 for all three pollutants when using the modeled ambient concentrations. The investigators concluded that measurements at well-characterized monitoring sites are necessary to provide accurate estimates of personal exposures in areas with concentrated sources.

The scatter plots of the modeled versus measured personal concentrations were also used to estimate the contributions of outdoor sources to personal exposures, on the assumption that the modeled concentrations represented personal exposures to outdoor sources only. The results showed varying contributions from outdoor sources depending on the model used and the location.

RELATED HEI PUBLICATIONS: AIR TOXICS AND RISK ASSESSMENT

Number	Title	Principal Investigator	Date*
Research Reports			
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156	Concentrations of Air Toxics in Motor Vehicle–Dominated Environments	E.M. Fujita	2011
153	Improved Source Apportionment and Speciation of Low-Volume Particulate Matter Samples	J.J. Schauer	2010
150	Mutagenicity of Stereochemical Configurations of 1,3-Butadiene Epoxy Metabolites in Human Cells	R.Q. Meng	2010
149	Development and Application of a Sensitive Method to Determine Concentrations of Acrolein and Other Carbonyls in Ambient Air	T.M. Cahill	2010
144	Genotoxicity of 1,3-Butadiene and Its Epoxy Intermediates	V.E. Walker	2009
143	Measurement and Modeling of Exposure to Selected Air Toxics for Health Effects Studies and Verification by Biomarkers	R.M. Harrison	2009
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132	An Updated Study of Mortality among North American Synthetic Rubber Industry Workers	E. Delzell	2006
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	<i>Part I.</i> Collection Methods and Descriptive Analyses	C.P. Weisel	2005
	<i>Part II.</i> Analyses of Concentrations of Particulate Matter Species	B.J. Turpin	2007
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