Published in final edited form as:

Atmos Environ (1994). 2012 September ; 57: 72–79. doi:10.1016/j.atmosenv.2012.04.029.

Exposures to volatile organic compounds (VOCs) and associated health risks of socio-economically disadvantaged population in a “hot spot” in Camden, New Jersey

Xiangmei (May) Wu^{a,b,1}, Zhihua (Tina) Fan^{a,b,*}, Xianlei Zhu^{a,b,2}, Kyung Hwa Jung^{a,b,3}, Pamela Ohman-Strickland^{a,c}, Clifford P. Weisel^{a,b}, and Paul J. Liroy^{a,b}

^aUniversity of Medicine and Dentistry of New Jersey (UMDNJ), USA

^bUMDNJ – Robert Wood Johnson Medical School, USA

^cUMDNJ – School of Public Health, USA

Abstract

To address disparities in health risks associated with ambient air pollution for racial/ethnic minority groups, this study characterized personal and ambient concentrations of volatile organic compounds (VOCs) in a suspected hot spot of air pollution – the Village of Waterfront South (WFS), and an urban reference community – the Copewood/Davis Streets (CDS) neighborhood in Camden, New Jersey. Both are minority-dominant, impoverished communities. We collected 24-h integrated personal air samples from 54 WFS residents and 53 CDS residents, with one sample on a weekday and one on a weekend day during the summer and winter seasons of 2004–2006. Ambient air samples from the center of each community were also collected simultaneously during personal air sampling. Toluene, ethylbenzene, and xylenes (TEX) presented higher ($p < 0.05$) ambient levels in WFS than in CDS, particularly during weekdays. A stronger association between personal and ambient concentrations of MTBE and TEX was found in WFS than in CDS. Fourteen to forty-two percent of the variation in personal MTBE, hexane, benzene, and TEX was explained by local outdoor air pollution. These observations indicated that local sources impacted the community air pollution and personal exposure in WFS. The estimated cancer risks resulting from two locally emitted VOCs, benzene and ethylbenzene, and non-cancer neurological and respiratory effects resulting from hexane, benzene, toluene, and xylenes exceeded the US EPA risk benchmarks in both communities. These findings emphasized the need to address disparity in health risks associated with ambient air pollution for the socio-economically disadvantaged groups. This study also demonstrated that air pollution hot spots similar to WFS can provide robust setting to investigate health effects of ambient air pollution.

© 2012 Elsevier Ltd. All rights reserved.

*Corresponding author. EOHSI, 170 Frelinghuysen Road, Piscataway, NJ 08854, USA. Tel.: +1 732 445 3194; fax: +1 732 445 0116. zfan@eohsi.rutgers.edu (Z.(Tina) Fan).

¹Current address: Department of Public Health, University of California, Davis, California, USA.

²Current address: College of Geosciences, China University of Petroleum-Beijing, Beijing, China.

³Current address: College of Physicians and Surgeons, Columbia University, New York City, New York, USA.

Keywords

VOCs; Community air pollution; Hot spot; Personal exposure; Health risks; Socio-economically disadvantaged population

1. Introduction

Volatile organic compounds (VOCs) are a group of air pollutants that are emitted from multiple types of anthropogenic sources, such as power plants, gas stations, autobody and paint shops, and diesel and gasoline-powered vehicles. Numerous VOCs are listed as hazardous air pollutants by the U.S. Environmental Protection Agency (USEPA) (2009). Several VOCs are known or suspected carcinogens while other VOCs may affect the immune system, central nervous system (brain), liver, and kidneys. Some studies have suggested associations between ambient VOCs and adverse health outcomes, such as asthma (Delfino et al., 2003; Wichmann et al., 2009). However, the impact of ambient VOC air pollution on public health has not been clearly defined. One reason for this is that the ambient VOC concentrations are relatively low compared to occupational settings. Also, populations are exposed to many pollutants at once, placing a challenge on epidemiological studies of health effects from environmental VOC exposure. Moreover, indoor sources significantly contribute to personal exposure to many VOCs, making it difficult to differentiate the health effects associated with exposure to ambient air pollution from those associated with exposure to indoor sources. Therefore, characterization of personal and ambient concentrations of VOCs in a hot spot of air pollution, where ambient air pollution is expected to be a significant contributor to personal exposure, may provide a more robust setting to evaluate the health effects of ambient air pollution.

In an air pollution hot spot, elevated air pollution levels as well as a large spatial variation are expected due to many localized emission sources (Jia et al., 2008; Ohura et al., 2006; Smith et al., 2007; Spicer et al., 1996; Touma et al., 2006; Zhu et al., 2008). Thus, measurements obtained from the national and state monitoring programs may underestimate personal exposure and thus health risks in air pollution hot spots. Minority and low socio-economic groups often live in areas that are air pollution hot spots, and may suffer greater health risks associated with ambient air pollution than the general population (Apelberg et al., 2005; Brown, 1995; Brulle and Pellow, 2006; D'Souza et al., 2009; Marshall, 2008; Morello-Frosch et al., 2002). Therefore, characterization of exposures and health risks of air pollution for those highly exposed individuals is imperative. Such data can help regulatory agencies to better address community concerns about air pollution and aid in the development of effective strategies to reduce community exposure and health risks, particularly for the subgroups living in hot spots of air pollution.

The Village of Waterfront South (WFS) neighborhood in Camden, NJ is a suspected hot spot of air pollution given the large number of local industrial sources and a high volume of diesel truck traffic. It was suspected that outdoor air pollution in WFS was above the urban background levels or the neighboring communities and may significantly contribute to personal exposures. The WFS neighborhood is comprised of a significant percentage of African Americans (61.2%) and impoverished population (33%), percentages much higher

than the national average (~13% for both categories) (Wu et al., 2010). Though health data for the WFS neighborhood are not available, some diseases, e.g., asthma and cancer, have been found to be at a much higher rate in Camden County than the NJ State average (NJDHSS, 2006, 2008). To address the community concerns about exposure to local ambient air pollution and associated health risks, we conducted a study to characterize ambient and personal exposures to a suite of air toxics in WFS (Liroy et al., 2011; Zhu et al., 2011). This paper focuses on personal and ambient concentrations of VOCs in WFS and a nearby urban reference site – Copewood/Davis Streets (CDS) in Camden, NJ, examining variations in ambient and personal concentrations by season and by day-of-the-week, determining the contribution from local ambient air pollution to personal exposure, and estimating the health risks associated with VOC exposures for this racial/ethnic minority groups.

2. Methods

2.1. Site description

The WFS neighborhood is characterized as an 800 m × 1200 m rectangular in the southwest of Camden, NJ (NJDEP, 2005). The sources of air pollution located in and near the WFS neighborhood include industrial sources, mobile sources, and urban mixed air pollution sources from Philadelphia (NJDEP, 2005). The majority of homes in the WFS are located less than 200 m from one or more stationary sources of air pollution, and a major highway I-676, with ~80,000 vehicles day⁻¹ (NJDOT, 2007), is located ~100 m east of WFS. Several hundred trucks traveled through and idled within WFS to serve local industries. The urban reference site, the CDS neighborhood, is located ~1000 m east of the WFS, without any identifiable nearby (<1000 m) industrial facilities. However, CDS is surrounded by a state highway with ~25,000 vehicles day⁻¹ and a local road with ~8000 vehicles day⁻¹. About 80% of CDS residences live within three blocks (<500 m) from the two roads. In addition, the dominant wind direction in this area is from southwest. On most sampling days WFS, although closer to I-676, was upwind of the three roads while CDS was downwind of the state highway and I-676. Thus, the impact of mobile sources on the air concentration from the major thoroughfares in the area might have been greater for CDS than WFS. The demographic information of the two neighborhoods is similar, with both containing a majority of socio-economically disadvantaged residents (Wu et al., 2010).

2.2. Sample collection

The study protocol and the consent form were approved by the Institutional Review Board of the University of Medicine and Dentistry of New Jersey prior to the start of the project. A total of 107 subjects were recruited from non-smoking homes for this study, among which 54 subjects were from WFS and 53 subjects were from CDS. While the recruitment was not a strict stratified randomization procedure, the study cohort was typical of the neighborhood population, except that females were over-sampled by 10%–20% (Wu et al., 2010). Field sampling was conducted on a total of 92 sampling days between June 2004 and July 2006. Four 24-h integrated air samples were collected from each subject: two in summer and two in winter; on one weekday and on one weekend day during each season. During each sampling day, two to six subjects were monitored. Each subject wore an OVM 3500 badge

(3M Company, St Paul, MN) close to the breathing zone for 24 h, and the sampling start time from all the subjects during each sampling day was usually within 2 h of each other. During the personal sampling, neighborhood ambient air samples were collected concurrently from the two fixed monitoring site in WFS and CDS, one in each neighborhood. Given the concern of potential exposure to secondhand environmental tobacco smoke (ETS) in these communities, nicotine, a marker of ETS, was also measured in a subset of personal samples (N = 234).

A Baseline Questionnaire and a Time/Activity Diary Questionnaire were administrated during each sampling day to obtain information that may affect personal exposure to VOCs, such as the demographics of the participants, house characteristics and location, utility type, personal activities, time spent in different microenvironments, etc.

2.3. VOC measurement

After sampling, the OVM 3500 badges were extracted with 2:1 (volume ratio) acetone/carbon disulfide mixture, and the extract was analyzed by an HP 6890/5973 GC/MS equipped with a RTX 624 capillary column (60 m $\times$ 0.25 mm inner diameter 1 μ m thickness). The detailed sample processing and analytical procedures can be found in Chung et al. (1999). The target compounds included methyl tert-butyl ether (MTBE), hexane, chloroform, carbon tetrachloride, benzene, toluene, ethylbenzene, *m/p*-xylenes, *o*-xylene, and styrene. Most compounds selected have known emission sources in the WFS area, e.g. diesel emissions, a waste combustor, and paint application. Nicotine level was measured using Polyurethane foam (PUF) sampler worn by the subjects, and the detailed method can be found in Liroy et al. (2011).

For quality assurance and quality control purpose, field blanks (15%) and duplicate samples (12%) were collected across the entire study period. The method detection limits (MDLs) of the VOCs, determined as three times of the standard deviation (SD) of all field blanks values collected from the entire project, ranged from 0.1 to 1 μ g m⁻³ for a sampling duration of 24 h using the OVM sampling badge (Liroy et al., 2011). Except styrene, the measurement precision (CV%) for all of the target compounds was within 20%.

2.4. Data analyses

Summary statistics were calculated for both ambient concentrations and personal exposures of the target VOCs. Concentrations below MDLs were assigned a value of one-half of the MDL. Mixed effect models were used to examine the location differences in ambient and personal concentrations between the WFS and CDS, and temporal variability, i.e., seasonal variation and day-of-the-week variation. The log-transformed concentrations were used in the analyses, since most VOC concentrations were right-skewed. Location (WFS vs. CDS), season (summer vs. winter), and day-of-the-week (weekday vs. weekend) were considered fixed effects, and subject was considered as a random effect. The confounding effect of ETS exposure on the association between personal and ambient concentrations was evaluated by incorporating the measured nicotine level as a covariate into the model.

The contribution of the ambient VOC air pollution to personal exposure was quantified using:

$$P_i = \alpha + \beta \cdot A_i + \varepsilon_i \quad (1)$$

where P_i is the measured personal exposure concentration to an individual VOC for participant i , A_i is the concentration of a VOC measured at the fixed ambient site in each neighborhood on the same day as P_i . This model actually presents the overall impact of ambient air pollution to personal exposure, including both direct exposure outdoors and the exposure to ambient air transported into the indoor environment by ventilation.

Our previous study showed that the study population spent more than twice the amount of time outdoors than the U.S. general population (Wu et al., 2010). Given the dense sources of VOCs in WFS and the proximity to those sources, local residents may be subject to a higher health risk resulting from exposure to ambient air pollution outdoors than the U.S. general population. To quantify the contribution of ambient VOC air pollution to personal exposure through direct exposure outdoors, we incorporated the percentage of time spent outdoors ($t_{a,i}$ %) in the neighborhood into the model (1),

$$P_i = \alpha + \beta \cdot (A_i \cdot t_{a,i} \%) + \varepsilon_i \quad (2)$$

“ $A_i \cdot t_{a,i} \%$ ” was considered as the outdoor exposure of the participant. Using ambient exposure instead of ambient concentration in the model, the R^2 of the model (2) provided a more accurate estimate of the variation in personal exposure to VOCs caused by the change of direct exposure to VOCs in ambient microenvironment, and the regression coefficient indicated whether ambient exposure was a significant factor contributing to personal VOC exposures. The outliers due to occupational exposure or ETS were identified and excluded on the basis of residual plots. Only samples ($N = 266$) in which the subject spent more than 30% of his/her time in the neighborhood and $t_{a,i} \% > 0$ during the sampling period were included in the analysis.

Health risks associated with VOC exposures were estimated based on ambient VOC concentrations for both cancer and non-cancer endpoints, using the conventional approaches developed by the USEPA (2005; 2007). The detailed description of the method can be found in the Supplementary materials (S1). Summary statistics, including mean, median, and upper percentile of the risk estimates were calculated. All of the analyses were performed using SAS (version 9.1; SAS Institute Inc., Cary, NC).

3. Results

The descriptive statistics of ambient and personal VOC concentrations are presented in Tables 1 and 2, respectively. Except for styrene, the concentrations of all of the target compounds in >60% of the samples collected are above the MDL. Given the high percentage (>50%) of non-detects for styrene, it was not included in the subsequent analysis.

3.1. Ambient VOCs concentrations

Ambient VOC concentrations in WFS and CDS communities varied by compound, with the highest mean for toluene and the lowest mean for styrene (Table 1). Extremely high concentrations were observed for hexane and benzene on a number of sampling days in both neighborhoods. Additional QA/QC of the data did not suggest contamination in those samples, which indicated localized sources on those sampling days. Note that the high concentration may not be very accurate due to potential loss by back diffusion of OVM samplers.

The mean concentrations of toluene, ethylbenzene, and xylenes (TEX) were found significantly higher ($p = 0.01$) in WFS than in CDS, with greater differences in the summer. The location differences between WFS and CDS for TEX were greater ($p = 0.01$) on weekdays than on weekends. Unlike TEX, no statistical significant differences were found for benzene and MTBE between WFS and CDS. Chlorinated compounds were present at the same level in both WFS and CDS.

A seasonal difference was observed, with the winter concentrations of TEX in CDS being significantly higher ($p < 0.05$) than the summer concentrations. However, the ambient TEX levels in WFS were not significantly different by season (Table 1). The median benzene concentrations were similar in the two seasons ($1.25 \mu\text{g m}^{-3}$ in the summer vs. $1.30 \mu\text{g m}^{-3}$ in the winter), but a few individual high concentrations observed in summer in both communities resulted in the mean benzene levels being higher in the summer than in the winter. No significant seasonal variation was observed for MTBE.

The difference in mean concentration of toluene and xylenes in the WFS by day-of-the-week approached statistical significance ($p = 0.05$ – 0.08), with weekday concentrations exceeding weekend days. No difference was observed for the other compounds by day-of-the-week in WFS. The only compound that had differences by day-of-the-week in CDS was hexane, which was higher on weekend days than weekdays ($p = 0.05$).

3.2. Personal VOCs concentrations

Personal exposure levels to VOCs were generally much higher than ambient concentrations in both neighborhoods (Table 2). Extremely high individual personal concentrations of toluene and benzene levels were observed in several personal samples. For example, the highest benzene concentration was found to be $83 \mu\text{g m}^{-3}$ in one of the CDS subject. No ETS exposure was reported for those samples in the questionnaire. The peak personal benzene levels coincided with the highest CDS ambient benzene levels on days when the levels of TEX were relatively low. These results suggested that the benzene peaks were probably caused by benzene emission sources other than automobiles.

The trend in mean personal exposure and ambient levels in WFS and CDS were not as expected. Both mean and median concentrations for all VOCs but toluene showed slightly higher values in CDS than in WFS, although not statistically significant. Personal benzene concentrations were higher in CDS than in WFS, and the difference was significant in summer ($p < 0.01$), driven by the few high concentrations found in CDS subjects. The mean personal concentration of toluene was higher in WFS than in CDS ($p < 0.01$), but the

difference was primarily due to several high values observed in WFS. We examined the impact of ETS on personal exposure based on the nicotine measurements. The nicotine concentration was significantly higher ($p < 0.01$) in CDS ($0.61 \pm 0.17 \mu\text{g m}^{-3}$) than in WFS ($0.26 \pm 0.16 \mu\text{g m}^{-3}$), indicating that higher personal levels of TEX and benzene for the CDS participants were partially attributed to ETS exposure.

No statistically significant seasonal difference was observed for personal exposure of TEX and MTBE in either community (except *o*-xylene in WFS). The occasional high personal benzene concentration discussed above caused significantly higher mean benzene concentrations in summer compared to winter. In addition, chloroform and carbon tetrachloride had significantly higher concentrations in the summer than in the winter. In WFS, TEX concentrations were higher during weekdays than those on weekend days, but this pattern was not observed in CDS.

3.3. Contribution of ambient VOC levels to personal exposure

Results of regression model (1) and (2) are presented in Table 3. The R^2 of Model (1) showed that the portion of exposure associated with an ambient air pollution origin explained 18–77% of the variation in several VOCs in WFS. The R^2 s of Model (1) calculated for the VOCs measured in WFS, except for hexane, toluene, and carbon tetrachloride, were greater than those in CDS. This suggests that there is a greater impact of outdoor air pollution on personal exposure in WFS than in CDS. We further examined the impact of ambient VOCs exposure occurring outdoors on the total exposure by incorporating the time spent outdoors in the neighborhood in the regression model (2). The ambient VOC exposure occurring outdoors explained 14–42% of the variation of personal exposure to MTBE, hexane and TEX in WFS, while the contribution of exposure outdoors was much lower in CDS. For TEX, the outdoor exposure contribution became insignificant (<5%) in CDS. Our results showed that the WFS neighborhood residents were subject to higher exposures from local ambient VOC pollutants during the time they stayed outdoors in the neighborhood, particularly for MTBE and TEX that have known local sources in the WFS.

Since TEX and benzene are components of ETS, the concentrations of nicotine, a marker of ETS exposure, was incorporated into the model as a covariate to evaluate the potential impact on the personal/ambient association. There was no change in regression coefficient or the p values of the model for WFS, re-affirming that ambient sources of TEX and benzene were important contributors to personal exposure in WFS and not ETS. However, the personal/ambient association in CDS became less significant and the regression coefficient of the ambient TEX and benzene concentrations decreased, indicating that exposure to ETS contributed considerably to the personal exposures in CDS.

3.4. Cumulative risk assessment

The cumulative cancer risk based on the mean ambient concentrations of the measured carcinogenic VOCs was 27 ± 16 per million in WFS and 26 ± 15 per million in CDS (Table 4). For the lower range of the cumulative carcinogen risks, benzene was the major contributor. The upper range of the cumulative cancer risk (90th percentile) was majorly

driven by chloroform and carbon tetrachloride, which were from indoor sources or regional background. However, additional risks were contributed by benzene and ethylbenzene, which had local outdoor sources. The mean and median cumulative non-cancer risk estimates were generally below the threshold for adverse effects; however, the upper range (95th percentile) non-cancer risks of neurological and respiratory effects were of concern. Non-cancer risks of neurological and respiratory effects were 2–17 times higher than the EPA benchmark on 5–10 sampling days in both communities. The risk of neurological effects was related to several target compounds, including hexane, benzene, toluene and xylenes, while the risk of respiratory effects was associated with exposures to hexane and toluene.

4. Discussion

This study reported VOC exposure levels and associated health risks for two socio-economically disadvantaged communities: one is an air pollution hot spot with dense local emission sources of several VOCs, and the other can be characterized as an urban reference site with no identified point sources of VOCs. Our results indicated that both disadvantaged communities were influenced by local air pollution sources, but the impact was stronger in the air pollution hot spot community WFS. Particularly, higher ambient TEX levels were measured in WFS than in CDS. TEX are widely used in industry as solvents and commonly found in consumer products, such as paint and glue. NJDEP (2005) identified more than ten industrial facilities in WFS that emit TEX, including paint applicators, several metal processing companies, and automobile repair shops, and a municipal waste combustion facility. The lack of seasonal variation of TEX supported that local point and area emission sources of TEX were the major contributors to WFS TEX air pollution, since industrial sources are relatively constant across seasons. In contrast, a strong seasonal pattern was observed for ambient TEX in CDS, with higher concentrations in the winter than in the summer. This pattern is consistent with the trend reported in many urban areas with dominant mobile sources (Adgate et al., 2004; Jia et al., 2008; Wallace et al., 1991). Furthermore, higher weekday toluene and xylene concentrations than weekend concentrations were observed in WFS but not in CDS. The variation by day-of-the-week was consistent with the industrial operation schedule pattern and traffic pattern.

For benzene and MTBE, the absence of differences by location, season, or day-of-the-week indicated that they were influenced primarily by mobile sources in both neighborhoods. As described in the method section, the relative position of the two neighborhoods versus the major traffic roads as well as the dominant wind direction in this area should result in both neighborhoods being influenced by mobile sources to a similar extent and therefore the air concentration of two automobile related compounds without other local industrial sources should be similar in both WFS and CDS.

For the two chlorinated compounds, chloroform and carbon tetrachloride, similar levels were found in the two neighborhoods. The concentrations of carbon tetrachloride in both neighborhoods were also consistent with the national-wide ambient background levels (McCarthy et al., 2006). The findings were expected because neither compound has any

significant sources in this region. Carbon tetrachloride, in particular, has been phased out in consumer products since 1970.

The above arguments about the VOC sources were confirmed by our spatial variation study of VOCs conducted during the same study period (Zhu et al., 2008). Based on the analysis of proximity of multiple sampling sites to the local point sources and roadways, an industrial paint shop and a local recycling plant were identified as the major sources of TEX in WFS (Zhu et al., 2008). In addition, ambient PAHs were measured simultaneously with the VOC measurement (Zhu et al., 2011). We found significant correlations between MTBE and total PAH ($p < 0.05$), and between benzene and fluoranthene and pyrene, while no correlations were observed between TEX and PAHs. Since mobile sources were found to be the major sources of PAHs in WFS and CDS (Zhu et al., 2011), the correlation analysis results re-affirmed that mobile sources were the significant contributors to ambient MTBE and benzene pollution, and local industrial sources were dominant sources of local ambient TEX in WFS.

This study was conducted in two minority-dominated and socio-economically disadvantaged communities. We found that the cohort who participated in our study spent significantly more time outdoors than the U.S. general population, which may lead to higher exposures to local ambient air pollution and associated health risks (Wu et al., 2010). By comparing the R^2 of model (1) and model (2), the ambient concentrations explained 15–77% of the variation of personal exposure to various VOCs when the total exposure (both indoors and outdoors) were considered (Model 1); while the direct exposure to ambient VOCs occurred outdoors explained 5–42% of the variation of personal exposure (Model 2). For compounds with dense ambient sources in WFS, namely TEX, personal exposure contributed by direct outdoor exposure (14–21%) comprised of a significant proportion of the total exposure to ambient air origin (21–33%) in WFS, while the contribution from the direct outdoor exposure to personal exposure to TEX was not significant in CDS (<5%). These results suggested that WFS, an area with dense local industrial sources, provided a better setting to study the impact of ambient air pollution on personal exposure and health effects than CDS, although the personal exposure levels in CDS were comparable to those in WFS.

The ambient and personal VOCs concentrations measured in WFS were compared to those measured in other urban areas in the US, including Elizabeth, NJ (Weisel et al., 2005), South Baltimore, MD (Buckley et al., 2005), New York City, NY (Kinney et al., 2005), and Dearborn, MI (Jia et al., 2008). All of the four urban areas also have significant sources of VOCs. The arithmetic mean ambient benzene concentrations were $2.18 \pm 3.01 \mu\text{g m}^{-3}$ in WFS and $2.86 \pm 4.72 \mu\text{g m}^{-3}$ CDS, higher than those in Elizabeth ($1.44 \pm 1.57 \mu\text{g m}^{-3}$), South Baltimore ($1.88 \pm 1.06 \mu\text{g m}^{-3}$), and Dearborn ($1.71 \mu\text{g m}^{-3}$). The mean personal benzene concentration in CDS ($4.73 \pm 8.30 \mu\text{g m}^{-3}$) was ~25% higher than the other places. The median concentrations of toluene in WFS ($5.83 \mu\text{g m}^{-3}$) was higher than Dearborn but similar to the rest of three areas, and the large arithmetic standard deviations of personal toluene concentrations found in WFS ($28.0 \pm 286 \mu\text{g m}^{-3}$) and CDS ($11.4 \pm 40 \mu\text{g m}^{-3}$) suggests that the residents living in these two neighborhoods experienced more high-end exposures to toluene than in other locations. For the remaining species, such as ethylbenzene, the ambient and personal median concentrations measured in WFS and CDS were

found to be lower than those observed in the other four studies. Note that the TEACH study, the South Baltimore study and the RIOPA study were conducted on average four years prior to our study. The regional ambient air pollutants levels have decreased in the past 10 years (Lioy and Georgopoulos, 2011). Thus, considering the decreased regional background ambient levels of benzene and toluene, their levels in WFS were still found similar to or higher than in other urban areas, suggesting the strong local sources of benzene and toluene in WFS.

Despite the ambient and personal concentrations of several VOCs in WFS being similar or slightly lower than in the other urban areas, the contributions of outdoor sources to personal exposures appeared to be more significant in WFS. For example, we made a comparison of the scatter plots of personal and ambient toluene concentrations from the WFS and CDS neighborhoods and the available data from the studies conducted in Elizabeth (Weisel et al., 2005) and Baltimore (Buckley et al., 2005). As shown in Fig. 1, the contributions of outdoor sources to personal exposure of TEX and benzene in WFS were similar to or greater than in the other two areas because more data points measured in WFS were closer to 1:1 line than in other two cities. These results further demonstrated a greater impact of local ambient air pollution on personal exposure in the WFS neighborhood, and suggested that WFS is an adequate community for studying the impact of ambient air pollution on community health.

It is important to note that secondhand smoking is unavoidable in these underserved communities. Nicotine was found in personal air samples from both neighborhoods, although our subjects were from non-smoking households. Our results suggested that nicotine should be measured in personal samples for study population in this type of community so that the impact from ETS exposure can be taken into account. In addition, our risk assessment was based on limited species of VOCs. Many other VOCs and air toxics, e.g., carbonyls and PAHs, measured in the Camden study were not included in the risk assessment. A comprehensive risk assessment is needed based on the air toxics measurements collected from the Camden study. Such information can assist in developing effective controlling strategies to reduce air pollution and community exposure and better addressing health burden associated with ambient air pollution for the minority and low-income communities located in hot spots of air pollution.

Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

Acknowledgments

We sincerely thank the participants for their time and interest in our study. We thank Marta Hernandez and other colleagues at EOHSI for their support of field sampling. This work was supported by the Health Effects Institute (HEI Agreement Number: 4703-RFA03-1/03-15). Drs. Fan, Lioy, Ohman-Strickland and Weisel are also partially supported by the NIEHS sponsored UMDNJ Center for Environmental Exposures and Disease, Grant# NIEHS P30ES005022.

Appendix A: Supplementary data

Supplementary data related to this article can be found online at doi:10.1016/j.atmosenv.2012.04.029.

References

- Adgate JL, Church TR, Ryan AD, Ramachandran G, Fredrickson AL, Stock TH, Morandi MT, Sexton K. Outdoor, indoor, and personal exposure to VOCs in children. *Environmental Health Perspectives*. 2004; 112:1386–1392. [PubMed: 15471730]
- Apelberg BJ, Buckley TJ, White RH. Socioeconomic and racial disparities in cancer risk from air toxics in Maryland. *Environmental Health Perspectives*. 2005; 113:693–699. [PubMed: 15929891]
- Brown P. Race, class, and environmental-health – a review and systematization of the literature. *Environmental Research*. 1995; 69:15–30. [PubMed: 7588491]
- Brulle RJ, Pellow DN. Environmental justice: human health and environmental inequalities. *Annual Review of Public Health*. 2006; 27:103–124.
- Buckley, TJ.; Payne-Sturges, D.; Kim, SR.; Weaver, V. VOC Exposure in an Industry-Impacted Community NUATRC Research Report. 2005. Available at: https://sph.uth.tmc.edu/mleland/attachments/Buckley_Report4.pdf
- Chung CW, Morandi MT, Stock TH, Afshar M. Evaluation of a passive sampler for volatile organic compounds at ppb concentrations, varying temperatures, and humidities with 24-h exposures. 2. Sampler performance. *Environmental Science & Technology*. 1999; 33:3666–3671.
- D'Souza JC, Jia C, Mukherjee B, Batterman S. Ethnicity, housing and personal factors as determinants of VOC exposures. *Atmospheric Environment*. 2009; 43:2884–2892.
- Delfino RJ, Gong H, Linn WS, Pellizzari ED, Hu Y. Asthma symptoms in Hispanic children and daily ambient exposures to toxic and criteria air pollutants. *Environmental Health Perspectives*. 2003; 111:647–656. [PubMed: 12676630]
- Jia C, Batterman S, Godwin C. VOCs in industrial, urban and suburban neighborhoods, part 1: indoor and outdoor concentrations, variation, and risk drivers. *Atmospheric Environment*. 2008; 42:2083–2100.
- Kinney, P.; Chillrud, S.; Sax, S.; Ross, JM.; Pederson, DC.; Johnson, D.; Aggarwal, M.; Spengler, JD. Toxic Exposure Assessment: A Columbia-Harvard (TEACH) Study (The New York City Report) NUATRC Research Report. 2005.
- Lioy PJ, Georgopoulos PG. New Jersey: a case study of the reduction in urban and suburban air pollution from the 1950s to 2010. *Environmental Health Perspectives*. 2011; 119:1351–1355. [PubMed: 21622086]
- Lioy, PJ.; Fan, Z.; Zhang, J.; Georgopoulos, P.; Wang, SW.; Ohman-Strickland, P.; Wu, XM.; Zhu, X.; Harrington, J.; Tang, X.; Meng, Q.; Jung, KW.; Kwon, J.; Hernandez, M.; Bonnanno, L.; Held, J.; Neal, J. Personal and Ambient Exposures to Air Toxics in Camden, New Jersey HEI Agreement Number: 4703-RFA03-1/03-15. 2011. Available at: <http://pubs.healtheffects.org/getfile.php?u=659>
- Marshall JD. Environmental inequality: air pollution exposures in California's South Coast air Basin. *Atmospheric Environment*. 2008; 42:5499–5503.
- McCarthy MC, Hafner HR, Montzka SA. Background concentrations of 18 air toxics for North America. *Journal of the Air & Waste Management Association*. 2006; 56:3–11. [PubMed: 16499141]
- Morello-Frosch R, Pastor M, Porras C, Sadd J. Environmental justice and regional inequality in southern California: implications for future research. *Environmental Health Perspectives*. 2002; 110:149–154. [PubMed: 11929723]
- NJDEP. Camden Waterfront South Air Toxics Pilot Project. Department of Environmental Protection, Division of Air Quality; New Jersey: 2005.
- NJDHSS. Asthma in New Jersey: August 2006. Department of Health and Senior Services; New Jersey: 2006.

- NJDHSS. NJ State Cancer Registry (NJSCR) Reports. 2008. Available:.. Department of Health and Senior Services, New Jersey <http://nj.gov/health/ces/reports.shtml> (accessed 5.01.08.)
- NJDOT. Roadway Information and Traffic Counts. 2007. Available:.. Department of Transportation, State of New Jersey <http://www.state.nj.us/transportation/refdata/roadway/traffic.shtm> (accessed 29.05.07.)
- Ohura T, Amagai T, Fusaya M. Regional assessment of ambient volatile organic compounds in an industrial harbor area, Shizuoka, Japan. *Atmospheric Environment*. 2006; 40:238–248.
- Smith LA, Stock TH, Chung KC, Mukerjee S, Liao XL, Stallings C, Afshar M. Spatial analysis of volatile organic compounds from a community-based air toxics monitoring network in Deer Park, Texas, USA. *Environmental Monitoring and Assessment*. 2007; 128:369–379. [PubMed: 17057989]
- Spicer CW, Buxton BE, Holdren MW, Smith DL, Kelly TJ, Rust SW, Pate AD, Sverdrup GM, Chuang JC. Variability of hazardous air pollutants in an urban area. *Atmospheric Environment*. 1996; 30:3443–3456.
- Touma JS, Isakov V, Ching J, Seigneur C. Air quality modeling of hazardous pollutants: current status and future directions. *Journal of the Air & Waste Management Association*. 2006; 56:547–558. [PubMed: 16739790]
- USEPA. Guidelines for Carcinogen Risk Assessment Risk Assessment Forum. 2005. Available at:.. U.S. Environmental Protection Agency, Washington, DC <http://www.epa.gov/cancerguidelines/>
- USEPA. Integrated Risk Information System (IRIS). 2007. Available:.. U.S. Environmental Protection Agency, Washington, DC <http://www.epa.gov/IRIS/> (accessed 4.12.07.)
- USEPA. 2002 National-Scale Air Toxics Assessment (NATA). 2009. <http://www.epa.gov/nata2002/> (accessed 23.12.09.)
- Wallace L, Nelson W, Ziegenfuss R, Pellizzari E, Michael L, Whitmore R, Zelon H, Hartwell T, Perritt R, Westerdaahl D. The Los Angeles TEAM Study: personal exposures, indoor-outdoor air concentrations, and breath concentrations of 25 volatile organic compounds. *Journal of Exposure Analysis and Environmental Epidemiology* April. 1991; 1:157–192.
- Weisel CP, Zhang JF, Turpin BJ, Morandi MT, Colome S, Stock TH, Spektor DM, Korn L, Winer A, Alimokhtari S, Kwon J, Mohan K, Harrington R, Giovanetti R, Cui W, Afshar M, Maberti S, Shendell D. Relationships of Indoor, Outdoor, and Personal Air (RIOPA): Part I. Collection Methods and Descriptive Analyses. *Health Effects Institute Number*. 2005; 130
- Wichmann FA, Müller A, Busi LE, Cianni N, Massolo L, Schlink U, Porta A, Sly PD. Increased asthma and respiratory symptoms in children exposed to petrochemical pollution. *Journal of Allergy and Clinical Immunology*. 2009; 123:632–638. [PubMed: 19111332]
- Wu X, Fan Z, Ohman-Strickland P. Time-location patterns of a population living in an air pollution hotspot. *Journal of Environmental and Public Health*. 2010; 2010:1155/2010/625461
- Zhu X, Fan Z, Wu X, Meng Q, Wang SW, Georgopoulos P, Tang X, Ohman-Strickland P, Zhang J, Held J, Liou PJ. Spatial variation of volatile organic compounds in a “Hot spot” in New Jersey. *Atmospheric Environment*. 2008; 42:7329–7338. [PubMed: 21603123]
- Zhu X, Fan Z, Wu X, Jung KH, Ohman-Strickland P, Bonanno LJ, Liou P. Ambient concentrations and personal exposure to polycyclic aromatic hydrocarbons (PAH) in an urban community with mixed sources of air pollution. *Journal of Exposure Science and Environmental Epidemiology*. 2011; 21:437–449. [PubMed: 21364704]

Fig. 1.

Comparison of personal and outdoor concentrations of toluene between (a) the Waterfront South neighborhood in Camden, NJ, (b) the Coopwood/Davis neighborhood in Camden, NJ, (c) Elizabeth in NJ (Weisel et al., 2005), and (d) Baltimore in MD (Buckley et al., 2005).

Table 1

Descriptive summary for ambient air concentrations ($\mu\text{g m}^{-3}$).^a

Compound	Overall			By season		p-value ^c	By day-of-the-week		p-value ^d
	N ^b	Geomean	95th%	Summer	Winter		Weekday	Weekend	
WFS									
MTBE	99	1.44 ± 2.94	5.57	1.60 ± 2.54	1.25 ± 3.48	0.35	1.32 ± 3.26	1.61 ± 2.53	0.34
Benzene	92	1.35 ± 2.42	8.64	1.55 ± 2.70	1.15 ± 2.03	0.03	1.39 ± 2.62	1.30 ± 2.18	0.56
Hexane	87	2.83 ± 7.57	65.1	8.33 ± 8.61	0.89 ± 2.69	<0.01	2.17 ± 3.95	3.93 ± 13.4	0.10
Toluene	99	2.48 ± 2.88	15.8	2.29 ± 3.09	2.77 ± 2.60	0.45	2.88 ± 3.08	2.04 ± 2.56	0.05
Ethylbenzene	99	0.40 ± 2.24	1.65	0.36 ± 2.28	0.47 ± 2.16	0.11	0.43 ± 2.28	0.36 ± 2.19	0.20
<i>m/p</i> -xylene	99	1.24 ± 2.24	4.27	1.11 ± 2.21	1.45 ± 2.25	0.12	1.39 ± 2.34	1.07 ± 2.08	0.07
<i>o</i> -xylene	99	0.44 ± 2.18	1.63	0.40 ± 2.03	0.50 ± 2.36	0.27	0.49 ± 2.19	0.39 ± 2.14	0.08
Styrene ^e	99	0.12 ± 2.03	0.32	0.11 ± 2.20	0.13 ± 1.77	N/A	0.12 ± 2.05	0.11 ± 2.00	N/A
Chloroform	62	0.13 ± 2.13	0.33	0.20 ± 2.07	0.10 ± 1.86	<0.01	0.13 ± 1.91	0.13 ± 2.39	0.75
Carbon tetrachloride	62	0.52 ± 1.22	0.68	0.56 ± 1.19	0.49 ± 1.21	0.01	0.52 ± 1.22	0.53 ± 1.22	0.81
CDS									
MTBE	86	1.48 ± 2.69	7.54	1.55 ± 2.77	1.39 ± 2.61	0.35	1.41 ± 2.92	1.57 ± 2.43	0.35
Benzene	80	1.46 ± 2.82	16.0	1.73 ± 3.46	1.20 ± 2.01	0.07	1.50 ± 2.66	1.40 ± 3.06	0.87
Hexane	75	2.69 ± 7.40	1109	7.21 ± 10.2	0.98 ± 2.16	<0.01	1.79 ± 3.22	4.39 ± 13.8	0.05
Toluene	86	1.67 ± 2.41	5.85	1.32 ± 2.35	2.28 ± 2.29	0.01	1.79 ± 2.47	1.54 ± 2.35	0.50
Ethylbenzene	86	0.33 ± 2.03	1.16	0.26 ± 1.96	0.45 ± 1.90	<0.01	0.34 ± 2.00	0.32 ± 2.09	0.85
<i>m/p</i> -xylene	86	0.90 ± 2.21	3.18	0.75 ± 2.14	1.14 ± 2.19	0.01	0.92 ± 2.24	0.87 ± 2.21	0.74
<i>o</i> -xylene	86	0.31 ± 2.19	0.91	0.27 ± 2.15	0.37 ± 2.19	0.03	0.33 ± 2.10	0.28 ± 2.30	0.30
Styrene ^e	86	0.10 ± 2.08	0.37	0.09 ± 2.27	0.12 ± 1.75	N/A	0.10 ± 1.94	0.10 ± 2.24	N/A
Chloroform	62	0.14 ± 2.24	0.71	0.22 ± 1.98	0.10 ± 2.02	<0.01	0.13 ± 1.90	0.15 ± 2.59	0.63
Carbon tetrachloride	62	0.53 ± 1.21	0.70	0.57 ± 1.18	0.50 ± 1.21	0.01	0.53 ± 1.21	0.53 ± 1.21	0.64

^aData are presented in geometric mean ± geometric standard deviation.^bTotal sample size: There were a few extremely high concentrations of benzene and hexane which may be due to solvent contamination, and were not included in calculation the distribution. Measurements of chloroform and carbon tetrachloride were added in the middle of the study, and thus have smaller sample size.^cComparison between the arithmetic mean summer and winter concentrations, *p*-value of *t*-test based on log-transformed concentrations.

^d Comparison between the arithmetic mean weekday and weekend concentrations, p -value of t -test based on log-transformed concentrations.

^e Comparisons by season and by day-of-the-week were not conducted on styrene concentrations, as only 32% of the styrene concentrations were above detection limit.

Table 2

Descriptive summary for personal air concentrations ($\mu\text{g m}^{-3}$).^a

Compound	Overall			By season		By day-of-the-week		
	N ^b	Geomean	95th%	Summer	Winter	p-value ^c	Weekday	Weekend
<i>WFS</i>								
MTBE	200	2.24 ± 3.21	12.9	2.46 ± 3.28	2.03 ± 3.13	0.71	1.97 ± 3.26	2.61 ± 3.12
Benzene	187	2.24 ± 2.05	7.24	2.20 ± 2.16	2.28 ± 1.96	0.64	2.34 ± 2.12	2.12 ± 1.98
Hexane	181	3.46 ± 6.25	230	6.59 ± 10.1	1.95 ± 2.64	<0.01	2.28 ± 2.68	5.26 ± 10.3
Toluene	200	5.61 ± 2.56	17.4	5.48 ± 2.36	5.75 ± 2.78	0.42	6.26 ± 2.96	4.94 ± 2.05
Ethylbenzene	200	0.88 ± 2.24	3.16	0.82 ± 2.36	0.95 ± 2.11	0.26	1.00 ± 2.27	0.76 ± 2.17
<i>m/p</i> -xylene	200	2.58 ± 2.17	9.36	2.50 ± 2.13	2.68 ± 2.22	0.45	2.90 ± 2.30	2.26 ± 1.98
<i>o</i> -xylene	200	0.83 ± 2.12	2.66	0.82 ± 2.09	0.83 ± 2.16	0.05	0.92 ± 2.21	0.73 ± 1.97
Styrene ^e	200	0.20 ± 2.25	0.78	0.14 ± 2.00	0.29 ± 2.13	N/A	0.22 ± 2.29	0.18 ± 2.18
Chloroform	173	0.55 ± 2.67	3.32	0.68 ± 2.53	0.46 ± 2.72	0.05	0.57 ± 2.79	0.54 ± 2.57
Carbon tetrachloride	173	0.50 ± 1.29	0.71	0.55 ± 1.24	0.45 ± 1.28	<0.01	0.49 ± 1.30	0.50 ± 1.28
<i>CDS</i>								
MTBE	165	2.53 ± 2.96	17.9	2.71 ± 3.84	2.34 ± 2.02	0.62	2.54 ± 2.93	2.51 ± 3.01
Benzene	153	3.10 ± 2.23	11.5	4.30 ± 2.29	2.28 ± 1.91	<0.01	3.17 ± 2.19	3.03 ± 2.29
Hexane	139	4.88 ± 7.53	1184	12.8 ± 13.9	2.34 ± 2.32	<0.01	2.86 ± 2.43	8.27 ± 13.6
Toluene	165	5.75 ± 2.69	24.4	5.47 ± 2.90	6.07 ± 2.47	0.74	6.09 ± 2.58	5.42 ± 2.81
Ethylbenzene	165	0.98 ± 2.66	5.43	0.91 ± 2.71	1.05 ± 2.60	0.22	1.07 ± 2.30	0.89 ± 3.02
<i>m/p</i> -xylene	165	2.67 ± 2.86	14.9	2.52 ± 2.91	2.84 ± 2.81	0.30	2.94 ± 2.48	2.41 ± 3.25
<i>o</i> -xylene	165	0.87 ± 2.72	4.92	0.86 ± 2.79	0.88 ± 2.66	0.36	0.95 ± 2.49	0.80 ± 2.95
Styrene ^e	165	0.21 ± 2.81	1.03	0.14 ± 2.95	0.32 ± 2.20	N/A	0.22 ± 2.62	0.19 ± 3.00
Chloroform	130	0.65 ± 4.18	6.95	0.86 ± 3.75	0.54 ± 4.37	0.04	0.67 ± 4.10	0.63 ± 4.30
Carbon tetrachloride	130	0.45 ± 1.35	0.66	0.49 ± 1.45	0.42 ± 1.25	0.02	0.46 ± 1.30	0.44 ± 1.40

^aData are presented in geometric mean ± geometric standard deviation.

^bTotal sample size: There were a few extremely high concentrations of benzene and hexane which may be due to solvent contamination, and were not included in calculation the distribution. Measurements of chloroform and carbon tetrachloride were added in the middle of the study, and thus have smaller sample size.

^cComparison between the arithmetic mean summer and winter concentrations, *p*-value of *t*-test based on log-transformed concentrations.

^d Comparison between the arithmetic mean weekday and weekend concentrations, *p*-value of *t*-test based on log-transformed concentrations.

^e Comparisons by season and by day-of-the-week were not conducted on styrene concentrations, as only 32% of the styrene concentrations were above detection limit.

Table 3

Regression analysis on the contribution of direct ambient exposure (the exposure that individuals experienced outdoors in the neighborhood) to personal exposure to VOC.^a

Compounds	$P_i = \alpha + \beta A_i \varepsilon$				$P_i = \alpha + \beta(A_i t_{o,i} \%) + \beta^b$					
	N	β_i	SE ^c	p-value	R ²	N	β_i	SE ^c	p-value	R ²
WFS										
MTBE	155	1.00	0.04	<0.001	0.77	109	0.49	0.06	<0.001	0.42
Hexane	130	0.54	0.08	<0.001	0.27	95	0.33	0.06	<0.001	0.26
Chloroform	139	0.53	0.10	<0.001	0.18	96	0.17	0.07	0.018	0.06
Carbon tetrachloride	137	1.03	0.07	<0.001	0.61	94	0.06	0.03	0.019	0.06
Benzene	145	0.55	0.06	<0.001	0.40	105	0.22	0.05	<0.001	0.15
Toluene	155	0.40	0.06	<0.001	0.21	109	0.26	0.05	<0.001	0.19
Ethylbenzene	155	0.50	0.07	<0.001	0.25	108	0.22	0.05	<0.001	0.14
<i>m</i> & <i>p</i> -Xylene	155	0.60	0.07	<0.001	0.33	109	0.28	0.05	<0.001	0.21
<i>o</i> -Xylene	155	0.50	0.06	<0.001	0.28	108	0.23	0.05	<0.001	0.16
CDS										
MTBE	133	0.82	0.07	<0.001	0.51	91	0.40	0.06	<0.001	0.35
Hexane	105	0.68	0.08	<0.001	0.39	73	0.33	0.08	<0.001	0.20
Chloroform	101	0.66	0.16	<0.001	0.15	70	0.36	0.11	0.003	0.13
Carbon tetrachloride	102	1.10	0.07	<0.001	0.71	71	0.04	0.03	0.209	0.02
Benzene	124	0.40	0.05	<0.001	0.30	86	0.20	0.05	<0.001	0.15
Toluene	132	0.62	0.10	<0.001	0.23	92	0.18	0.08	0.034	0.05
Ethylbenzene	134	0.67	0.13	<0.001	0.17	92	0.08	0.10	0.425	0.01
<i>m</i> & <i>p</i> -Xylene	134	0.61	0.12	<0.001	0.16	92	0.10	0.10	0.306	0.01
<i>o</i> -Xylene	134	0.65	0.12	<0.001	0.19	92	0.13	0.09	0.180	0.02

^aProc GLM in SAS is used for the regression analysis.

^bResults are based on the model $\log P_i = \alpha + \beta \log(A_i t_{o,i} \%) + \varepsilon_i$. Samples with $t_{o,i} \% = 0$ are not included.

^cSE: standard error.

Table 4
Cancer and non-cancer risk estimates based on ambient VOC concentrations in the WFS and CDS neighborhoods.

Compound	WFS				CDS					
	N	Mean	SD	Median	90th%	N	Mean	SD	Median	90th%
Cancer Risks ^a										
MTBE	99	0.6	0.7	0.4	1.2	86	0.6	0.8	0.4	1.2
Chloroform	62	4.3	5.9	3.0	8.9	62	4.7	5.8	3.2	5.5
Carbon tetrachloride	62	7.9	1.5	8.2	9.8	62	8.1	1.5	8.2	9.9
Benzene	92	17	23	9.3	50	80	22	37	9.8	33
Ethylbenzene	99	1.4	1.1	1.0	2.3	86	1.1	0.9	0.8	2.9
Cumulative ^b	59	27	16	24	39	59	26	15	22	39
Non-cancer Risks ^c										
Neurological system ^d	106	0.6	2.2	0.1	1.9	92	0.7	2.3	0.1	3.8
Respiratory system ^e	106	0.4	2.1	0.0	0.8	92	0.5	2.1	0.0	2.1
Liver and Kidney	106	0.0	0.0	0.0	0.0	92	0.0	0.0	0.0	0.0
Reproductive/Developmental system	106	0.0	0.0	0.0	0.0	92	0.0	0.0	0.0	0.0
Immune systems	106	0.1	0.4	0.0	0.5	92	0.2	0.8	0.0	0.7

^aUnit in excess cancer risks per million population.

^bCumulative cancer risk was calculated for each study participant by summing compound-specific cancer risk estimates for all of the known, probable, and possible carcinogenic target VOCs, and then the distribution was calculated based on cumulative cancer risks for individual study participants.

^cUnitless.

^dThe risk of neurological effects was related to hexane, benzene, toluene and xylenes.

^eThe risk of respiratory effects was associated with exposures to hexane and toluene.