Spatial variation in ambient benzene concentrations over a city park

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Abstract

Passive diffusive samplers were used to collect ambient benzene for a one week sampling period (April 27 to May 4, 2011) at eleven locations throughout a city park in the Tampa area. Concentrations were determined through gas chromatography with mass spectrometry. Spatial variability within the park and its contribution to uncertainty in health risk estimates were studied. Measured concentrations ranged from 0.23 – 0.34 μg m^{-3}. The relative percent differences for samplers collocated with a regulatory reference monitor and with a duplicate were 3% and 14%, respectively. The spatial variability over the park was small with a coefficient of variation of 11%. The concentration variation due to sampler placement contributes less to uncertainty in health risk estimates than the uncertainty associated with the inhalation unit risk parameter (39% versus 170% relative percent differences over the ranges studied). Result suggest a limit to spatial resolution needed for risk calculations.

Keywords: air pollution, children's exposure, health risk uncertainty, passive sampling

Introduction

Benzene is a known human carcinogen (e.g. it is classified as Group 1 by the International Agency for Research on Cancer), and a substantial contributor to estimated health risks of ambient exposures to toxic air pollutants (McCarthy, O'Brien, Charrier, & Hafner, 2009).

Current regulatory monitoring networks for hazardous air pollutants, including benzene, use large and expensive systems that require electricity, restricting the location and number of measurement sites. This results in low spatial resolution of concentration data and inadequate characterization of environmental benzene exposures, particularly for children; hence, a high level of uncertainty is present in risk assessments (e.g. Möller, Schuetzle, & Autrup, 1994).

Due to continued lung development, higher breathing rates and activity levels, and more time spent outdoors, children are more susceptible to the health effects of air pollution than adults (American Academy of Pediatrics Committee on Environmental Health, 2004). High concentrations of combustion-related pollutants (including benzene) have been associated with higher incidences of acute respiratory infections in children (Romieu, Samet, Smith, & Bruce, 2002). Levels of a benzene metabolite in the urine of school children in Bangkok have been found to be comparable to that of adult street vendors who experienced higher ambient concentrations (Ruchirawat, Settachan, Navasumrit, Tuntawiroon, & Autrup, 2007). Toxicological support documents on benzene carcinogenicity for the U.S. Environmental Protection Agency (USEPA, 1998) suggest that both the type of leukemia and susceptibility differ between children and adults; however, not enough data are available to quantify these differences. Higher resolution data would contribute to the understanding of both exposure and health effect differences.

To understand children's exposures, concentrations of benzene where children spend time, such as parks and school grounds, are needed. In a review, Mejía et al. (2011) found that most studies on air pollutant exposures at school used data from remote monitoring stations or dispersion modeling. Studies that monitored on school grounds often did not indicate the specific location of measurement. The pollutant focus of previous work has primarily been on

NO_2, ozone, SO_2 and particulate matter. Work on spatial variations of benzene in urban areas has investigated impacts of roadway traffic and the relationship between indoor and outdoor concentrations. Thorsson and Eliasson (2006) and Menezes et al. (2009) included measurements of benzene in urban parks, in Sweden and Brazil respectively; they found concentrations in parks to be substantially lower than levels in high traffic areas. Janssen et al. found decreasing outdoor benzene concentrations with distance from the motorway in a study of 24 schools in the Netherlands (Janssen, van Vliet, Aarts, Harssema, & Brunekreef, 2001); indoor concentrations were higher than outdoor concentrations. Godoi et al. (2009) found measured benzene levels to be higher indoors than outdoors in a study of two schools in Brazil. By examining the indoor-outdoor concentration ratio, they concluded that the indoor concentrations were primarily due to outdoor sources. Overall, more measurement data is needed on the spatial variations in benzene concentrations within areas where children may be exposed.

Studies of environmental equity investigate the distribution of environmental exposures, risks, and effects among subpopulations of different race or socioeconomic status (Brown, 1995). In studies of multiple urban areas, disadvantaged groups have been found to disproportionately live in neighborhoods with high traffic density and poor air quality (Houston, Wu, Ong, & Winer, 2004; Pastor Jr, Sadd, & Morello-Frosch, 2002; Wheeler & Ben-Shlomo, 2005). In the Tampa area, studies have found that census tracts and elementary schools with higher proportions of black, Hispanic and low income groups are located in areas with higher levels of NO_2, air toxics, and traffic counts (Chakraborty, 2009; Stuart, Mudhasakul, & Sriwatanapongse, 2009; Stuart & Zeager, 2011). Using neighborhood-scale (block group) census data, Stuart et al. (2009) also found that these same groups disproportionately reside further from regulatory monitors. In a study of southern California, Houston et al. (2004) concluded that the historic

urban sprawl growth pattern resulted in minority and poor neighborhoods located in areas with high traffic density and of more affluent neighborhoods located in suburban areas. Hence, it contributed to a disproportionate burden of health effects on poorer populations. Additionally, urban sprawl is considered a less sustainable form of growth than compact, dense cities, due to increased land use and inefficiency of public transportation (Camagni, Gibelli, & Rigamonti, 2002). As cities work to manage growth, high resolution monitoring can guide approaches for maximizing exposure equality and urban sustainability.

This manuscript presents a pilot study of small-scale variation in benzene concentrations over a city park, and investigates impacts on the uncertainty of cancer risk estimates, particularly for children. We present the methods used for the sampling and analysis of benzene, method evaluation, and data analyses. Results are then discussed on ambient benzene levels, their spatial variation, and the uncertainty in cancer risk estimates introduced by sampler placement.

Figure 1

Methods

A city park in Hillsborough County, Florida was chosen as the field site for this pilot study. The park is located adjacent to an elementary school and shares a structured playground with the school. We chose eleven sampling sites with available standing structures for sampler placement (a utility pole or tree), with the goal of spanning the park with approximately equidistant placement. For mapping and analysis, location coordinates were determined using a Garmin GPS device (with accuracy listed as within 10 m). Figure 1a shows the study area and sampling locations. For evaluation of measurement accuracy, we collocated an additional sampler with the only benzene regulatory monitor for the county, operated by the Environmental Protection Commission (EPC) of Hillsborough County; this site is outside the park.

To characterize ambient benzene, we exposed passive diffusive samplers [RAD 130, Radiello, Ltd., activated charcoal sorbent] for a one week sampling period (April 27 to May 4, 2011) at the sites discussed above. A duplicate sampler was also exposed at site 8 for assessment of measurement precision. Two field blank cartridges accompanied the exposed samplers (one during deployment and the other during retrieval). Each field blank cartridge was uncapped and immediately resealed at one sampling location, as specified by the USEPA (1999b). Upon return to the laboratory, cartridges were stored at 4°C. Each cartridge was submersed for thirty minutes in 2 ml of carbon disulfide [Sigma-Aldrich, 342270-ReagentPlus, low benzene] for extraction. One laboratory blank cartridge was also used for quality control purposes. We quantified benzene via gas chromatography with mass spectrometry (GC/MS) [Varian Saturn 3800-GC, 2000-MS] using a polysiloxane capillary column [Varian CP-Sil 8 CB, 50 m x 0.25 mm x 0.25 μm]. A 1.0 μl aliquot of the extracted solution was injected on the column at a temperature of 240°C, with helium as the carrier gas at a flow rate of 1.2 ml min^{-1}. The total run time was sixty minutes with the following temperature program: start at 35°C for nine minutes, increase at 5°C min^{-1} to 60°C, hold at 60°C for 46 minutes. To quantify unknown concentrations, we diluted benzene stock [Fluka, 12540-Analytical Standard] to create five calibration standards ranging from 0.10-1.75 μg ml^{-1}. We used daily calibration curves, with a daily control chart for quality control. All samples and calibration standards were normalized by addition of 2-fluorotoluene [Sigma-Aldrich, F15323] as an internal standard. The analysis protocol was adapted for our specific equipment from the Radiello sampler manual (Fondazione Salvatore Maugeri, 2006), previous studies (Angiuli et al., 2003; Cocheo, Boaretto, & Sacco, 1996; Godoi et al., 2009), and established standard methods (USEPA, 1999a; USEPA, 1999b).

Quality control procedures and metrics were adapted from USEPA Compendium Method TO-15 (USEPA, 1999a).

We converted to concentrations in air by subtracting the mean blank value and multiplying the blank-adjusted values by a temperature-adjusted sampling rate and the exposure time recorded for each sampler. We used the sampling-period average of hourly temperatures measured at the Tampa International Airport (National Weather Service KTPA station) to adjust the sampling rate (Fondazione Salvatore Maugeri, 2006). The limit of detection was calculated as three times the standard deviation of the blank values, assuming a one week sampling time (10080 minutes).

To characterize spatial variations in benzene concentrations over the study area, we mapped the concentration distribution in ArcGIS, followed by spatial interpolation using kriging. Summary statistics, including the coefficient of variation (*CV*), were used to quantify variation.

To investigate the health risks associated with the concentrations measured here, we estimated excess lifetime cancer risk from inhalation exposure by multiplying the exposure concentration (C_E) by the inhalation unit risk (*IUR*) for benzene. The exposure concentration was calculated as $C_E = C_a t_E f_E d_E / T$ (USEPA, 2009), where C_a is the measured concentration of benzene, t_E is the exposure time (hours day^{-1}), f_E is the exposure frequency (days year^{-1}), d_E is the exposure duration (years), and T is the averaging time (hours). We chose exposure parameters to represent a child who is exposed due to time spent at school, with t_E of 6.5 hours day^{-1}, f_E of 180 days year^{-1}, and and d_E of 6 years (representing kindergarten through 5th grade) (Hillsborough County Public Schools, 2010; 2011). For T, we applied the value for excess lifetime cancer risk, of 70 years (in units of hours) (USEPA, 2009). The inhalation unit risk represents the upper bound excess cancer risk per unit of exposure concentration determined through review of

human and animal studies. USEPA lists a range of $2.2x10^{-6}$ to $7.8x10^{-6}$ per µg /m^3 (USEPA, 2010), while the California Office of Environmental Health Hazard Assessment lists a value of $2.9x10^{-5}$ per µg /m^3 (OEHHA, 2009). For this work, we estimated risks using the minimum and maximum of these values, i.e. $2.2x10^{-6}$ and $2.9x10^{-5}$. These were combined with the range of measured concentrations to estimate a range of risks.

Finally, we investigated the impact of sampler placement (and sampling resolution) on the uncertainty of health risks calculated using the standard methods discussed above. Since actual levels vary over the area represented by a single sample, the choice of sampler location leads to uncertainty in the exposure concentration. When propagated through the risk calculation, proportional uncertainty in the estimated health risk results. Uncertainty in the inhalation unit risk has a similar effect. Hence, we compared the range of measured concentrations (quantified as a relative percent difference) over the study area to that of the inhalation unit risk, in order to assess impacts of sampler placement on health risk assessment.

Table 1

Results and Discussion

Measured benzene concentrations and summary statistics for the study area are shown in Table 1. Results are at the low end of the range in weekly outdoor concentrations (0.3-5.0 µg m^{-3}) observed at schools in the Netherlands (Janssen et al., 2001) and weekly levels (0.3-1.4 µg m^{-3}) for an urban park in Gothenburg, Sweden (Thorsson and Eliasson, 2006; Figure 5 therein). This is consistent with previous work showing low concentrations in urban parks as compared with levels in high traffic areas (Upmanis, Eliasson, & Andersson-Skold, 2001). Overall, the values measured here are near the low end of the general range of benzene measured in U.S. studies, of

approximately 1-10 μg m^{-3}, with peak values up to about 50 μg m^{-3} (Health Effects Institute Air Toxics Review Panel, 2007).

The overall range of estimated excess cancer risk for the range of concentrations measured here is from $5.7x10^{-9}$ to $1.1x10^{-7}$. These risks are quite low when compared with other studies (Tam & Neumann, 2004; Payne-Sturges, Burke, Breysse, Diener-West, & Buckley, 2004; McCarthy et al., 2009), which reported cancer risk due to benzene of greater than 10^{-6} at almost all sites. Here, we considered only the contribution to the lifetime cancer risk due to elementary school exposure, not an exposure duration of an entire lifetime. Further, we note that the concentrations here are for one week of sampling, and hence, may not be representative of longer-term averages or extremes.

Evaluation results indicate a precision from the two collocated duplicate samplers of 14% (as a relative percent difference), which is within the recommended guideline of less than 25% (USEPA, 1999a). A 3% relative percent difference was found between the concentration measured by the reference regulatory monitor and that measured by a collocated passive sampler; this is within the precision of the passive measurement (that ultimately limits the measurable accuracy). We note that the comparison is not direct, since the sampling times are different. The reference monitor takes 24 hour canister samples every six days; the final 13 hours of one sampling period and the full second sampling period of the reference monitor overlapped with the sampling period for this study.

The spatially-interpolated concentration field is provided in Figure 1b. A gradient can be observed; the highest interpolated concentrations are in the northwest corner of the sampling area, which includes the site near the playground structure (site 2). The highest point measurement value was at site 3. The location of the high area is consistent with emissions

expected from the parking lot and vehicular park entrance nearby. Wind data for the Tampa area (at the international airport) do not indicate a dominant wind direction during the study period. However, only 12% of the winds were from the WNW to N quadrant, which may have kept concentrations low overall. It should be noted that, due to the precision of measurement, confidence in real differences is small. Overall, the coefficient of variation for the sampling area was only 11%. Although there is no standard metric for quantifying spatial heterogeneity, a value of 20% has been used to distinguish between a homogeneous and heterogeneous field in previous studies of particulate matter (Blanchard et al., 1999; Wilson, Kingham, Pearce, & Sturman, 2005). The appropriateness of this criteria for a benzene field is unknown, but is explored below.

The range of measured concentration values can be used to explore impacts of sampler placement on uncertainties in the estimated health risks of benzene. The relative percent difference between the maximum and minimum measured concentrations was 39%; this results in an equivalent percent difference in estimated excess health risks (if all other parameters are kept constant). As a comparison, a similar value (44%) can be calculated from the range of risks due to toxic air pollutant levels (driven primarily by formaldehyde and benzene) measured at four sites in a study of intra-urban differences in Pittsburgh (Logue, Small, Stern, Maranche, & Robinson, 2010). Conversely, the range of values for inhalation unit risk listed by the USEPA and California OEHHA is much larger, at 170% (as relative percent difference). (Note that this does not represent the full range of risks, just a range of upper bound risk estimates). Hence, the uncertainties associated with sampler placement in this study are small compared with those from the unit risk parameter; the variation in concentration over the sampling area would need to be much larger to have a similar independent effect. This suggests a threshold to the utility of

increased spatial resolution of measurement that will depend on the purpose for which the concentration measurements are used. However, we note that the effect on overall risk uncertainty is multiplicative, so even small variations in concentrations could lead to large overall uncertainties in risk. Further, there are uncertainties in other parameters in the calculation that are not analyzed here (e.g. personal activity patterns).

Although of limited utility for risk estimation at the very high resolution studied here, monitoring data that characterizes intra-urban variations may be helpful for epidemiological studies, development of better risk assessment values, and city planning. While considering the effects of misclassification of particulate matter exposure on inference from time-series mortality studies, Zeger et al. (2000) found that the largest bias was due to differences between the ambient levels measured by fixed-site monitors and average individual exposures. Spatially-resolved outdoor benzene concentrations may help reduce this bias, particularly given the association between indoor and outdoor concentrations of benzene (Godoi et al., 2009). In combination with personal activity data, higher-resolution concentration data may allow better estimation of individual exposures (Nuckols, Ward, & Jarup, 2004; Whitworth, Symanski, Lai, & Coker, 2011). However, the usefulness of additional increases in spatial resolution may currently be limited both by other uncertainties in risk estimation and the precision of passive measurements. Nonetheless, decreased misclassification of exposure in health effect studies may in turn lower the uncertainty in health risk assessment parameters. This is particularly important for children, as current USEPA risk assessment methods do not differential risk to children for benzene carcinogenicity. Finally, Jensen et al. (2001) suggest that high-resolution concentration data may aid city planners to build sustainability cities and reduce environmental inequity. For

these purposes, characterization of concentrations variations over space may be as important than differences in individual activities and exposures.

Conclusions

Eleven sampling locations were chosen at a city park to investigate spatial variation in ambient benzene using passive samplers. Precision and accuracy were evaluated through co-location. Excess lifetime cancer risks were estimated, and the magnitude of uncertainty due to sampler placement was compared to the uncertainty due to inhalation unit risk. The precision and accuracy were measured at 14% and 3%, respectively. The spatial variation over the park was found to be low (*CV* of 11%), with an overall range, as a relative percent difference, of 39%. Comparison suggests limits to the use of these methods for very high resolution sampling, but appropriateness for study of larger-scale intra-urban concentration variations of benzene in Hillsborough County.

Compared with the range of inhalation unit risk of 170% (as a relative percent difference), these results indicate that one sampler may be sufficient to represent the excess lifetime cancer risk from benzene exposure for the sampling area, using currently recommended methods. Further research is needed to characterize the spatial variation in benzene concentrations and risks over larger intra-urban scales, including neighborhoods. The necessary monitoring resolution will depend on the purpose for which the data will be applied. High-resolution data may lower errors in epidemiological studies resulting from differences between actual and measured concentrations, may improve risk assessment parameters (particularly for less-characterized groups, like children), and may aid city design.

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References

American Academy of Pediatrics Committee on Environmental Health. (2004). Ambient air pollution: Health hazards to children. *Pediatrics, 114*, 1699-1707.

Angiuli, L., Bruno, P., Caputi, M., Caselli, M., de Gennaro, G., & de Rienzo, M. (2003). Radial passive samplers for air quality monitoring in field comparison with a BTEX automatic analyser: preliminary results. *Fresenius Environmental Bulletin, 12*, 1167-1172.

Blanchard, C.L., Carr, E.L., Collins, J.F., Smith, T.B., Lehrman, D.E., & Michaels, H.M. (1999). Spatial representativeness and scales of transport during the 1995 integrated monitoring study in California's San Joaquin valley. *Atmospheric Environment, 33*, 4775-4786.

Brown, P. (1995). Race, class, and environmental health: A review and systematization of the literature. *Environmental Research, 69*, 15-30.

Camagni, R., Gibelli, M.C., & Rigamonti, P. (2002). Urban mobility and urban form: the social and environmental costs of different patterns of urban expansion. *Ecological Economics, 40*, 199-216.

Chakraborty, J. (2009). Automobiles, air toxics, and adverse health risks: Environmental inequities in Tampa Bay, Florida. *Annals of the Association of American Geographers, 99*, 674-697.

Cocheo, V., Boaretto, C., & Sacco, P. (1996). High uptake rate radial diffusive sampler suitable for both solvent and thermal desorption. *American Industrial Hygiene Association Journal, 57*, 897-904.

Fondazione Salvatore Maugeri. (2006, January). *Radiello*. Retrieved September 13, 2010, from http://www.radiello.com/english/Radiello's%20manual%2001-06.pdf.

Godoi, R.H.M., Avigo Jr, D., Campos, V.P., Tavares, T.M., de Marchi, M.R.R., Van Grieken, R., Godoi, A.F.L. (2009). Indoor air quality assessment of elementary schools in Curitiba, Brazil. *Water, Air, & Soil Pollution: Focus, 9*, 171-177.

Health Effects Institute Air Toxics Review Panel. (2007). *Mobile-Source Air Toxics: A Critical Review of the Literature on Exposure and Health Effects.* Boston, MA: Author.

Hillsborough County Public Schools. (2010). *Calendars*. Retrieved June 17, 2011, from http://www.sdhc.k12.fl.us/info/calendars/2010_11impdates.html.

Hillsborough County Public Schools. (2011). *Riverhills Information*. Retrieved June 17, 2011, from http://www.sdhc.k12.fl.us/schools/school_info.asp?site=3621.

Houston, D., Wu, J., Ong, P., & Winer, A. (2004). Structural disparities of urban traffic in southern California: Implications for vehicle-related air pollution exposure in minority and high-poverty neighborhoods. *Journal of Urban Affairs, 26*, 565-592.

Janssen, N.A.H., van Vliet, P.H.N., Aarts, F., Harssema, H., & Brunekreef, B. (2001). Assessment of exposure to traffic related air pollution of children attending schools near motorways. *Atmospheric Environment, 35*, 3875-3884.

Jensen, S.S., Berkowicz, R., Hansen, H.S., Hertel, O. (2001). A Danish decision-support GIS tool for management of urban air quality and human exposures. *Transportation Research Part D, 6*, 229-241.

Logue, J.M., Small, M.J., Stern, D., Maranche, J., & Robinson, A.L. (2010). Spatial variation in ambient air toxics concentrations and health risks between industrial-influenced, urban, and rural sites. *Journal of the Air & Waste Management Association, 60*, 271-286.

McCarthy, M.C., O'Brien, T.E., Charrier, J.G., & Hafner, H.R. (2009). Characterization of the chronic risk and hazard of hazardous air pollutants in the United States using ambient monitoring data. *Environmental Health Perspectives, 117*, 790-796.

Mejía, J.F., Choy, S.L., Mengersen, K., & Morawska, L. (2011). Methodology for assessing exposure and impacts of air pollutants in school children: Data collection, analysis and health effects -- A literature review. *Atmospheric Environment, 45*, 813-823.

Menezes, H.C., Amorium, L.C.A., & Cardeal, Z.L. (2009). Sampling of benzene in environmental and exhaled air by solid-phase microextraction and analysis by gas chromatography-mass spectroscopy. *Analytical and Bioanalytical Chemistry, 395*, 2583-2589.

Möller, L., Schuetzle, D., & Autrup, H. (1994). Future research needs associated with the assessment of potential human health risks from exposure to toxic ambient air pollutants. *Environmental Health Perspectives, 102 (S-4)*, 193-210.

Nuckols, J.R., Ward, M.H., & Jarup, L. (2004). Using geographic information systems for exposure assessment in environmental epidemiology studies. *Environmental Health Perspectives, 112*, 1007-1015.

Office of Environmental Health Hazard Assessment. (2009, May). *Air Toxics Hot Spots Risk Assessment Guidelines Part II: Technical Support Document for Cancer Potency Factors-Appendix B: Chemical-specific summaries of the information used to derive unit risk and*

cancer potency values. Retrieved January 19, 2011, from http://www.oehha.ca.gov/air/hot_spots/2009/AppendixB.pdf.

Pastor Jr, M., Sadd, J.L., & Morello-Frosch, R. (2002). Who's minding the kids? Pollution, public schools, and environmental justice in Los Angeles. *Social Science Quarterly, 83*, 263-280.

Payne-Sturges, D.C., Burke, T.A., Breysse, P., Diener-West, M., & Buckley, T.J. (2004). Personal exposure meets risk assessment: A comparison of measured and modeled exposures and risks in an urban community. *Environmental Health Perspectives, 112*, 589-598.

Romieu, I., Samet, J.M., Smith, K.R., Bruce, N. (2002). Outdoor air pollution and acute respiratory infections among children in developing countries. *Journal of Occupational and Environmental Medicine, 44*, 640-649.

Ruchirawat, M., Settachan, D., Navasumrit, P., Tuntawiroon, J., & Autrup, H. (2007). Assessment of potential cancer risk in children exposed to urban air pollution in Bangkok, Thailand. *Toxicology Letters, 168*, 200-209.

Stuart, A.L., Mudhasakul, S., & Sriwatanapongse, W. (2009). The social distribution of neighborhood-scale air pollution and monitoring protection. *Journal of the Air & Waste Management Association, 59*, 591-602.

Stuart, A.L., & Zeager, M. (2011). An inequality study of ambient nitrogen dioxide and traffic levels near elementary schools in the Tampa area. *Journal of Environmental Management, 92*, 1923-1930.

Tam, B.N., & Neumann, C.M. (2004). A human health assessment of hazardous air pollutants in Portland, OR. *Journal of Environmental Management, 73*, 131-145.

Thorsson, S., & Eliasson, I. (2006). Passive and active sampling of benzene in different urban environments in Gothenburg, Sweden. *Water, Air, and Soil Pollution, 173*, 39-56.

U.S. Environmental Protection Agency. (1998, April). *Carcinogenic Effects of Benzene: An Update.* Retrieved January 6, 2011, from Integrated Risk Information System: http://www.epa .gov/ncea/pdfs/benzenef.pdf.

U.S. Environmental Protection Agency. (1999a, January). *Compendium of Methods for the Determination of Toxic Organic Compounds in Ambient Air: Compendium Method TO-15.* Retrieved August 30, 2010, from http://www.epa.gov/ttnamti1/files/ambient/airtox/to-15r.pdf.

U.S. Environmental Protection Agency. (1999b, January). *Compendium of Methods for the Determination of Toxic Organic Compounds in Ambient Air: Compendium Method TO-17.* Retrieved August 30, 2010, from http://www.epa.gov/ttnamti1/files/ambient/airtox/to-17r.pdf.

U.S. Environmental Protection Agency. (2009, January). *Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual (Part F, Supplemental Guidance for Inhalation Risk Assessment).* Retrieved December 16, 2010, from http://www.epa.gov/oswer/riskassessment/ragsf/pdf/partf_200901_final.pdf.

U.S. Environmental Protection Agency. (2010, December 20). *Benzene (CASRN 71-43-2)*. Retrieved January 6, 2011, from Integrated Risk Information System: http://www.epa.gov/iris/subst/0276.htm.

Upmanis, H., Eliasson, I., & Andersson-Skold, Y. (2001). Case studies of the spatial variation of benzene and toluene concentrations in parks and adjacent built-up areas. *Water, Air, and Soil Pollution, 129*, 61-81.

Wheeler, B.W., & Ben-Shlomo, Y. (2005). Environmental equity, air quality, socioeconomic status, and respiratory health: a linkage analysis of routine data from the Health Survey for England. *Journal of Epidemiology & Community Health, 59*, 948-954.

Whitworth, K.W., Symanski, E., & Coker, A.L. (2008). Childhood lymphohematopoietic cancer incidence and hazardous air pollutants in southeast Texas, 1995-2004. *Environmental Health Perspectives, 116*, 1576-1580.

Whitworth, K.W., Symanski, E., Lai, D., & Coiker, A.L. (2011). Kriged and modeled ambient air levels of benzene in an urban environment: an exposure assessment study. *Environmental Health, 10 (21)*.

Wilson, J.G., Kingham, S., Pearce, J., & Sturman, A.P. (2005). A review of intraurban variations in particulate air pollution: Implications for epidemiological research. *Atmospheric Environment, 39*, 6444-6462.

Zeger, S.L., Thomas, D., Dominici, F., Samet, J.M., Schwartz, J., Dockery, D., Cohen, A. (2000). Exposure measurement error in time-series studies of air pollution: Concepts and consequences. *Environmental Health Perspectives, 108*, 419-426.

Figure Captions

Figure 1 a) The study area with sampling locations (and corresponding sample IDs) and b) a contour plot of interpolated concentrations ($\mu g/m^3$) [generated in ArcGIS]. The white box in a) corresponds to the area plotted in b). Sample ID 12 (not shown) was located at the site of the county reference monitor, outside of the study area. A duplicate sampler was located at site 8.

Tables

Table 1 Benzene concentrations and summary statistics.

Sample ID (or summary statistic)	Concentration (μg m^{-3})
1	0.33
2	0.33
3	0.34
4	0.28
5	0.31
6	0.27
7	0.29
8	0.29
8* (duplicate)	0.34
9	0.29
10	0.23
11	0.31
12	0.26
reference monitor	0.26
mean	0.30
standard deviation	0.03
range	0.23 - 0.34

The limit of detection was 0.18 μg/m^3. Sample ID locations are provided in Figure 1. Summary statistics are based on measurements from sites in the study area (1-11), excluding the site 8 duplicate value. The reference monitor value is the average of two 24-hr sample values (from April 27 and May 3) obtained from the Environmental Protection Commission of Hillsborough Co. from every 6th day monitoring during the study period. Reference-method samples are collected in 6 L stainless steel canisters and analyzed by GC/MS, per Compendium Method TO-15 (USEPA, 1999a). All calculations were performed using full precision in Excel; hence summary values cannot necessarily be reproduced from the concentration values reported to two significant digits.

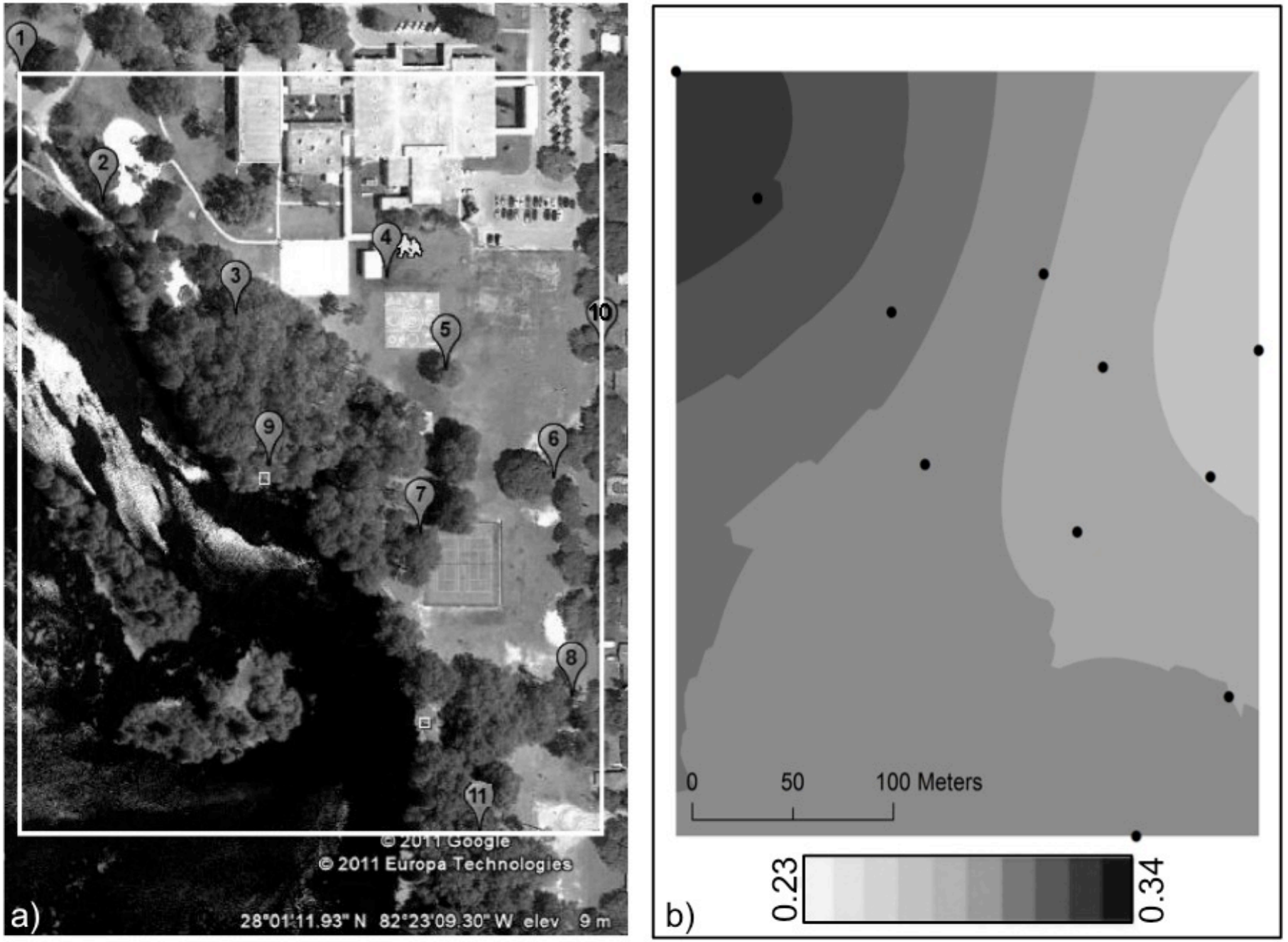

Figure 1. (Caption text provided with the manuscript text)