Monitoring, Source Identification and Health Risks of Air Toxics in Albuquerque, New Mexico, U.S.A.

Ilias G. Kavouras[1,2]*, David W. DuBois[1,3], George Nikolich[1], Vic Etyemezian[1]

[1] *Division of Atmospheric Sciences, Desert Research Institute, Las Vegas, Nevada, USA*
[2] *Department of Environmental and Occupational Health, Fay W. Boozman College of Public Health, University of Arkansas for Medical Sciences, Little Rock, Arkansas, USA*
[3] *Department of Plant and Environmental Sciences, New Mexico State University, Las Cruces, New Mexico, USA*

ABSTRACT

Air toxics were monitored in three locations in Albuquerque, New Mexico, U.S.A representing different land use configurations to determine the air toxics concentration gradient and factors controlling their ambient levels and to assess potential health impacts. The total polycyclic aromatic hydrocarbons (PAHs), heavy metals and volatile organic compounds (VOCs) concentrations at all sites ranged from 18.0 to 323.8 ng/m³, from 5.7 to 122.0 ng/m³ and from 0.1 to 18.7 ppbv, respectively. Sources of air toxics were determined using coupled positive matrix factorization and linear regression analysis. For PAHs, a mixture of emissions from traffic, oil residues and wood burning was identified. The comparison of PAHs relative distributions in Albuquerque to those emitted from three types of woodstoves showed a good correlation with emissions from non-catalytic woodstoves. The values for toluene/benzene, xylene/benzene and xylene/toluene ratios in Albuquerque indicated that traffic was the major source of aromatic VOCs. In addition, regression analysis of air toxics to daily $PM_{2.5}$ source contributions showed that the vast majority of aromatic VOCs and PAHs were strongly associated with the secondary NO_3^- source category, providing additional evidence of the significant role of traffic emissions of air toxics and $PM_{2.5}$ in Albuquerque. A small fraction of PAHs and heavy metals was associated with road dust, indicating that mechanically released dust particles may be contaminated by oil residues and vehicle exhausts. The annual hazard quotient values for benzene, toluene, xylene and methylene chloride were typical for urban communities dominated by traffic emissions and similar to those estimated for other urban areas in the U.S. The annualized cancer risks for benzene and methylene chloride were lower than 1-in-a-million.

Keywords: Air toxics; Hazardous air pollutants; Health effects/risks; PAHs; Toxic air pollutants; Toxic metals; Urban aerosols.

INTRODUCTION

There are 188 hazardous air pollutants (HAPs), or air toxics, that require specific attention, long-term monitoring, health risk assessment and, eventually, regulation under the Clean Air Act and its Amendments (CAAA), because they have been associated with a wide variety of adverse health effects, including cancer, neurological effects, reproductive and developmental effects (Delfino, 2002; Leikauf, 2002; Windham *et al.*, 2006). These air toxics are emitted from multiple sources, including major stationary, area and mobile sources (Ozkaynak *et al.*, 2008; Logue *et al.*, 2009). The implementation of CAAA led to the development of the Integrated Urban Air Toxics Strategy (IUATS) and the

Air Toxics Program (ATP) which included the detailed investigation of a subset of 33 HAPs including aromatic hydrocarbons (e.g., benzene and its derivatives), halocarbons, heavy metals (e.g., As, Hg, Ni), polycyclic aromatic hydrocarbons (PAHs) and diesel particles in urban areas (USEPA, 2000; Kyle *et al.*, 2001). In addition, the National Air Toxics Assessment (NATA) program has been developed to monitor ambient concentrations, evaluate the impact of sources nationwide and assess the health risks.

The Albuquerque, New Mexico (NM), U.S.A. greater metropolitan area has almost 800,000 residents (Fig. 1). It is located by the Rio Grande River and the intersection of Interstate-40 and Interstate-25 highways. Urban growth and recent improvements in the transportation network have contributed to Albuquerque's development of industry. The major activities include manufacturing (electronic equipment, semiconductors, missile guidance systems, surgical appliances, transportation equipment and parts), printing and publishing and food processing. Major regional point sources are two gas-fired power plants and a coal fired

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* Corresponding author.

 Tel.: +1-501-526-6627; Fax: +1-501-526-6750
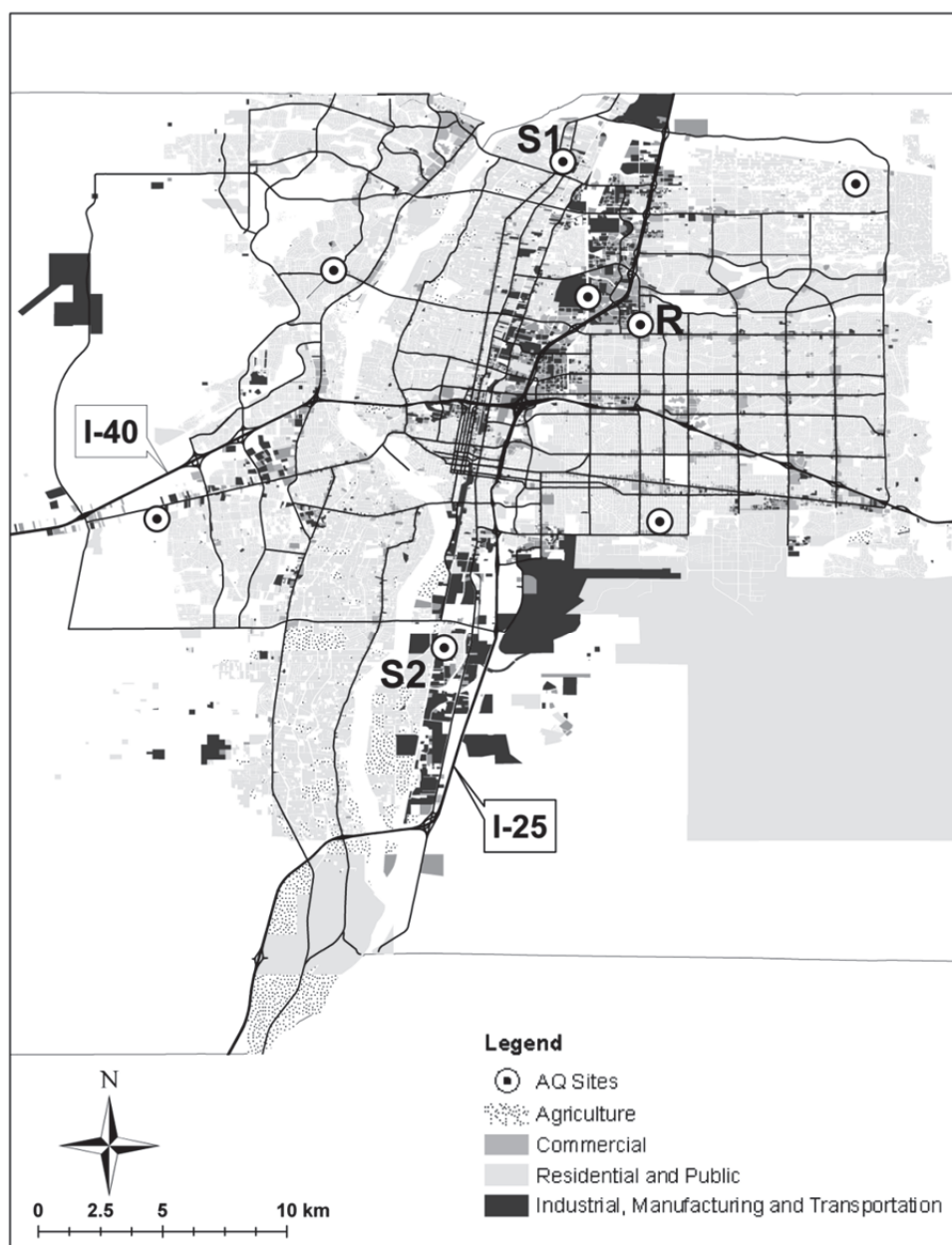 *E-mail address:* ikavouras@uams.edu

Fig. 1. The study region, land use and the locations of air toxics monitoring sites in Albuquerque, NM. (Monitoring sites: R: Reference, S1 and S2: Satellites sites 1 and 2).

cement plant. The coal-fired Four Corners Power Plant is one of the largest fossil-fuel steam-driven power plants in the world. There is also a power plant to the west, between Grants and Gallup, New Mexico and oil refineries located outside of Albuquerque. Most of these activities are controlled through the Maximum Achievable Control Technology standards. The 2002 NATA analysis showed that the lifetime cumulative cancer risk (30 in a million), neurological hazard index (4.6×10^{-2}) and respiratory hazard index (3.1) in Bernalillo County, New Mexico were among the highest estimated in the US (higher than the 75th percentile among all counties (USEPA, 2002a). The Pilot City Monitoring Program (PCMP) in 2001–2002 showed that the cancer risk and toxicity due to inhalation, calculated using EPA's

Prioritized Chronic Dose-Response values for screening risk assessments to the local population in the greater Albuquerque area was up to 12.44 in a million for benzene, especially for those residing in the vicinity of industrial/commercial/ warehousing activities (USEPA, 2002b).

We carried out a community-scale monitoring program in order to characterize air toxics concentrations and assess their effects on living standards and human health in and around the city of Albuquerque, NM. The objectives of this study were: (*i*) measure ambient concentration levels of air toxics within specific community regions in the city of Albuquerque; (*ii*) identify and evaluate the impact of air toxics sources; and (*iii*) assess adverse health impacts from exposure using risk assessment models.

MATERIALS AND METHODS

Sampling

A map of the Albuquerque area, showing major interstate highways, land use and the local air quality monitoring sites is presented in Fig. 1. The reference site (R) was located at the existing 2ZM site (Del Norte High School; Air Quality System (AQS) ID: 350010023), northeast of the intersection of I-25 and I-40. O_3, PM_{10}, $PM_{2.5}$ CO, NO_x and meteorological parameters were monitored at this site by AQD. In addition, this site has been used in a previous pilot-scale air toxics study (USEPA, 2002b). A major source at this site is the high density traffic from the nearby intersection of San Mateo and Montgomery. There were also a number of area sources such as filling stations, dry cleaners and automotive repair shops. The satellite sites included: (i) S1: 2ZH (North Valley; AQS ID: 350011013) located northwest of the reference site in the North Valley community. One of the major sources in the area is the Intel computer chip manufacturing plant which was about 3.2 km to the northwest. Other area sources included filling stations and emissions from sand and gravel plants to the southeast. O_3, CO and PM_{10} and $PM_{2.5}$ were measured at 2ZH; and (ii) S2: 2ZV (South Valley, AQS ID: 350010029) was located south-southwest of the reference site in the South Valley community, which represented a diversity in land use characterized by a mix of urban and rural population densities as well as industrial and commercial expansion. Urban growth began spreading into the South Valley from the north and along its major streets. Industrial activities included a power station, the City sewage treatment plant, asphalt plants and tanks farms, auto dismantlers, brick manufacturing, a chicken farm, and a meat packing plant. O_3, PM_{10}, and meteorological parameters were monitored continuously at 2ZV, which has been used in prior pilot-scale air toxics studies (USEPA, 2002b).

The schedule and frequency of sampling of volatile organic compounds (VOCs), polycyclic aromatic hydrocarbons (PAHs) and heavy metals is presented in Table 1. 24-hr integrated VOCs were measured using the Method TO-15 by the New Mexico State Laboratory Division (SLD) (USEPA, 1999a). Ambient samples were collected using CS1200 samplers equipped with TM1000 Start/stop timers and 6-L SUMMA canisters and analyzed, using a preconcentrator (Model 7100, Entech Instruments Inc, USA) and a gas chromatography (Carlo Erba GC-8060, CE Elantech Inc, USA) equipped with a mass spectrometer (Model MD-800, Fisons, Inc. United Kingdom). In addition, two intensive monitoring periods (IMP) during which hourly measurements of VOCs at the reference site were measured in February

and June 2008, using a SRI gas chromatography system (Model TO-14, SRI Instruments, USA). A multi-component certified calibration standard was obtained (Item No. 0102AZ00004ZCL, Scott Specialty Gases, USA). PAHs and heavy metals were measured at the reference and satellite sites on a 1-in-6 day frequency. Samples were collected by Environmental Health Department of the City of Albuquerque staff and analyzed by ERG Inc. using the TO-13A and ERG standard protocols (US EPA, 1999b; ERG, 2005). A portion of the quartz filters (1-cm^2) was also analyzed for elemental and organic carbon content (EC/OC) at Desert Research Institute's Environmental Analysis Facility (Chow *et al.*, 2001).

Source Apportionment

The Positive Matrix factorization (PMF) receptor model was used to identify the major sources of $PM_{2.5}$ in the study area and to estimate the contributions of major sources and source types to the $PM_{2.5}$ mass. The concentrations of $PM_{2.5}$ mass and chemical species measured at the NCore site in Albuquerque (reference site in our study) for the study period were retrieved from US EPA's AQS. Sampling was done in 1 every 3 days frequency using a four channel speciation sampler. Elements (from Na to U) were measured by spectroscopy (X-ray fluorescence, PIXE, ICP, ICP-MS); water-soluble ions (sulfate, nitrate, chloride, ammonium, sodium, potassium, calcium and magnesium) by ion chromatography and colorimetry; and elemental and organic carbon (EC and OC, respectively) by thermal optical reflectance method. PMF is a least squares formulation of factor analysis with built-in non-negativity constraints (Paatero and Tapper, 1994; Paatero, 1997). Through a solution that minimizes the objective function (Q in Eq. (1)) based on measurements uncertainties.

$$Q = \sum_{i}^{n}\sum_{k=1}^{m}\left[\frac{x_{ik} - \sum_{k=1}^{n}\left(g_{ik}\, x f_{kj}\right)}{\sigma_{ij}}\right] \tag{1}$$

where x_{ik} and σ_{ik} are the concentration and associated uncertainty of *k*-species in *i*-sample, g_{ik} is the contribution of the *j*-factor to particle mass in *i*-sample and f_{kj} is the mass fraction of *j*-species on *k*-factor, the source contribution (G(*nxp*)) and the source profile matrices (F(*pxm*)) where *p* is the number of sources are computed from the concentrations of *m*-aerosol species for *n*-sampling days (X(*nxm*)). The optimum number of factors (sources) and the F$_{peak}$ value (that defines the rotation) were selected on trial and error analysis of the solutions using a set of statistical tools and by comparison of the source profiles with previous source

Table 1. Sampling period and frequencies of air toxics in Albuquerque, New Mexico.

Compounds	Period	Location and frequency
VOCs (24-hr canister)	September 2007–August 2008 and December 2008–March 2009	Reference and satellite sites; 1-in 6 days
VOCs (1-hr online GC)	February and June 2008	Reference site; Every hour
PAHs	December 2008–March 2009	Reference and satellite sites; 1-in 6 days
Heavy metals	December 2008–March 2009	Reference and satellite sites; 1-in 6 days

profiles (Paatero *et al.*, 2002, 2005). Missing concentration data are replaced by the geometric mean of the measured concentrations while missing uncertainties are substituted by four times the geometric mean of measured uncertainties. In this effort, we run the PMF model in the robust method with $\alpha = 4.0$ using the error model "–12" (that uses observed values). We also added an additional modeling uncertainty of 25% to reduce modeling errors. We initially ran the model for 3 to 20 factors. The configuration includes 20 runs with random seed per each run. All base run results were converged which implied that the model had found the local minima. For more than 6 factors, the Q value did not change significantly, indicating that most of the variation was explained by five to six factors. The solutions for that six or more factors were rejected because they had excessive rotational freedom. The four-factor model was rejected because two factors, namely biomass burning and traffic emissions, were merged into a single factor. For the five-factor model, more than 90% of the residuals for individual chemical species were normally distributed between –3 and +3, indicating good agreements of the fit to the observed values. The bootstrap configuration included 200 runs, minimum r value of 0.75 and suggested block of 6 with a random seeding. The interquartile ranges for individual compounds were ± 20%.

Five-factor models with F_{peak} values lower than –1.5 or higher than 1.5 did not converge into a solution. Finally, a five factor model with a rotation with $F_{peak} = -0.5$ was selected based on the profiles of sources and the agreement between the calculated and estimated mass and elementals concentrations. The contributions of the retained factors to the individual PM$_{2.5}$ components were computed using a least-squares multivariate linear regression model (Song *et al.*, 2001)

$$X_{ik} = a + \sum_{k=1}^{p} \left(f_k \cdot \beta_k \cdot g_{ik} \right) \tag{2}$$

where β_k is the regression coefficient or scaling constant to convert the factor profiles and contributions to µg of compounds/µg of PM$_{2.5}$ and µg/m^3, respectively. The intercept, α, was attributed to other PM$_{2.5}$ sources. Negative regression coefficients indicated that there were too many factors. The coefficient of variation of the root mean square error, CV(RMSE) was used to evaluate the residuals between measured and predicted PM$_{2.5}$ and individual chemical species values. It was defined as the RMSE normalized to the mean of the observed values

$$CV\left(RMSE\right) = \frac{RMSE}{\overline{X_{measured}}} = \frac{\sqrt{\sum_{i=1}^{n} \left(X_{predicted,i} - X_{measured,i} \right)^2}}{n \cdot \overline{X_{measured}}} \tag{3}$$

with RMSE being defined as the sample standard deviation of the differences between predicted values and observed values, n is the number of measurements and $\overline{X_{measured}}$ is the average X-component concentration. PMF source

appointment results were compared with the air toxics data to investigate the relationship between the aerosol sources and ambient concentrations of the toxic components using equations 2 and 3. Due to their reactive nature, using PMF directly (or any statistical source apportionment model) on air toxics concentrations would not likely to yield useful results. Therefore, we used only the traditional aerosol chemical characterizations (metals, ions, EC/OC) to perform the brunt of the source apportionment. Results of the source apportionment were then combined with ambient air toxics concentration information to determine relationships between source contributions and air toxics measured at the sites (Vargas *et al.*, 2014).

Risk Assessment

The USEPA's Total Risk Integrated Methodology – Risk (TRIM.Risk) model was used to calculate human health cancer risks and chronic hazards by combining exposure estimates with toxicity values and using non-probabilistic exposure-response values (Efroymson and Murphy, 2001; ICF Consulting 2005) (USEPA, 2005). The exposure modeling was done using the Hazardous Air Pollutant Exposure Model (HAPEM5) based on measured ambient concentrations. The model uses the U.S. Census Bureau as the primary source of most population demographic data, which is organized in predefined tracts divided into a set of cohorts based on gender and age. Because measurements were obtained in three locations, each tract was assigned to the nearest monitoring site. The risk assessment was conducted for specific air toxics for which more than 50% of valid measurements were obtained during a year. These included: benzene (SAROAD Code: 45201), methylene chloride (SAROAD Code: 43802), Toluene (SAROAD Code: 45202) and xylenes (SAROAD Code: 45102). For inhalation risk assessments using non-probabilistic exposure-response (i.e., toxicity) values, TRIM.Risk combines toxicity values with human inhalation exposure estimates obtained from HAPEM and population-specific information to derive annual hazard quotient (AHQ) and annual cancer risk (ACR). The USEPA's prioritized Chronic Dose-Response Values for non-cancer were used to estimate the AHQ and ACR based on the exposure estimates. The annual hazard quotient (AHQ) and annual cancer risk (ACR) were computed as follows:

$$AHQ = ([X])/RfC \tag{4}$$
and
$$ACR = ([X] \cdot URE)/70 \tag{5}$$

where [X] is the mean annual concentration of X compound, RfC is the reference dose-response concentration for non-cancer effects of compound X and URE is the chronic dose-response concentration for cancer of compound X for a lifetime of 70 years.

RESULTS AND DISCUSSION

Concentration Levels

Table 2 shows the minimum, maximum, mean and standard deviation of 24-hr VOCs, PAHs and heavy metals

Table 2. The minimum, maximum, mean (and standard deviation) concentrations of 24-hr VOCs (in ppbv), PAHs (in ng/m^3) and heavy metals (in ng/m^3) at Del Norte (R), North Valley (S1) and South Valley (S2) sites.

Compound	Del Norte (R)	North Valley (S1)	South Valley (S2)
VOCs			
Benzene	0.1–1.4; 0.3 (0.2)	0.1–1.1; 0.3 (0.2)	0.1–1.7; 0.5 (0.4)
1,3-Butadiene	0.1–2.3; 0.3 (0.4)	0.1–1.7; 0.2 (0.2)	0.1–2.1; 0.4 (0.5)
Carbon tetrachloride	0.1–0.1; 0.1 (0.1)	0.1–0.1; 0.1 (0.1)	0.1–0.1; 0.1 (0.1)
Chloromethane	0.1–15.3; 1.1 (2.2)	0.4–5.1; 1.1 (0.8)	0.1–2.7; 0.7 (0.5)
Ethylbenzene	0.1–0.7; 0.1 (0.1)	0.1–0.5; 0.1 (0.1)	0.1–1; 0.2 (0.2)
Trichlorofluoromethane	0.1–0.4; 0.1 (0.1)	0.1–0.3; 0.2 (0.1)	0.1–0.5; 0.2 (0.1)
Dichlorodifluoromethane	0.1–0.9; 0.4 (0.1)	0.1–0.6; 0.4 (0.1)	0.1–0.9; 0.4 (0.1)
Trichlorotrifluoroethane	0.1–0.6; 0.1 (0.1)	0.1–0.2; 0.1 (0.1)	0.1–0.6; 0.1 (0.1)
Methylene Chloride	0.1–2.0; 0.5 (0.4)	0.1–6.4; 0.5 (0.9)	0.1–2.1; 0.3 (0.3)
Toluene	0.1–3.9; 0.7 (0.5)	0.1–2.8; 0.8 (0.5)	0.1–5.6; 1.2 (1.2)
1,2,4-Trimethylbenzene	0.1–0.5; 0.1 (0.1)	0.1–0.4; 0.1 (0.1)	0.1–0.8; 0.2 (0.2)
m/p-Xylenes	0.1–1.6; 0.3 (0.2)	0.1–1.3; 0.3 (0.2)	0.1–2.9; 0.6 (0.6)
o-Xylene	0.1–0.6; 0.1 (0.1)	0.1–0.5; 0.1 (0.1)	0.1–1.0; 0.2 (0.2)
PAHs			
Naphthalene	29.6–169; 106.5 (70.8)	13.6–234; 121.4 (76.9)	13.2–186; 106.7 (55)
Acenaphthylene	0.2–3.8; 1.6 (2.0)	0.1–14.8; 4.8 (5.4)	0.1–5.1; 1.4 (1.5)
Acenaphthene	0.9–1.8; 1.4 (0.5)	0.4–11.5; 2.4 (2.7)	0.5–6; 2.7 (1.5)
Fluorene	1.2–4.5; 2.8 (1.7)	0.7–13.2; 4.6 (3.6)	0.8–6.1; 3.5 (1.5)
Phenanthrene	2.5–10.9; 7.2 (4.3)	1.7–26.4; 11.0 (8.2)	1.5–15.4; 7.6 (3.7)
Anthracene	0.1–1.3; 0.6 (0.6)	0.1–20.6; 2.8 (5.3)	0.1–2; 0.7 (0.5)
Retene	0.1–6.6; 2.5 (3.6)	0.3–16.5; 3.3 (4.1)	0.1–6.6; 2.0 (2.2)
9-Fluorenone	0.3–3; 1.7 (1.3)	0.4–6.8; 2.7 (2.0)	0.3–3.5; 1.8 (0.9)
Fluoranthene	0.8–3.3; 2.0 (1.3)	0.5–10.1; 3.3 (2.8)	0.4–4.4; 2.2 (1.2)
Pyrene	0.5–2.9; 1.6 (1.2)	0.3–8.9; 2.8 (2.5)	0.3–3.6; 1.8 (1.1)
Benzo[a]anthracene	*bdl*–0.7; 0.3 (0.4)	0.1–2.4; 0.7 (0.7)	*bdl*–1.6; 0.4 (0.4)
Chrysene	0.1–1.1; 0.5 (0.5)	*bdl*–3.0; 0.8 (0.9)	0.1–2.4; 0.6 (0.6)
Cyclopenta[cd]pyrene	*bdl*–0.3; 0.1 (0.2)	0.1–1.7; 0.5 (0.5)	*bdl*–0.6; 0.2 (0.2)
Benzo[b]fluoranthene	0.1–1.4; 0.6 (0.7)	0.1–4; 1.1 (1.2)	0.1–2.4; 0.7 (0.7)
Benzo[k]fluoranthene	*bdl*–0.4; 0.2 (0.2)	0.1–4.2; 1.2 (1.3)	*bdl*–0.6; 0.2 (0.2)
Benzo[e]pyrene	0.1–0.6; 0.3 (0.3)	*bdl*–1.2; 0.3 (0.4)	0.1–1.1; 0.3 (0.3)
Benzo[a]pyrene	0.1–0.7; 0.3 (0.4)	*bdl*–1.8; 0.5 (0.5)	*bdl*–1.2; 0.3 (0.3)
Perylene	*bdl*–0.1; 0.1 (0.1)	*bdl*–2.5; 0.7 (0.8)	*bdl*–0.2; 0.1 (0.1)
Benzo[g,h,i]perylene	0.1–0.7; 0.4 (0.3)	*bdl*–0.4; 0.1 (0.1)	0.1–0.9; 0.3 (0.3)
Indeno[1,2,3-cd]pyrene	0.1–0.8; 0.3 (0.4)	0.1–1.9; 0.6 (0.6)	0.1–1; 0.4 (0.3)
Dibenz[a,h]anthracene	not detected	0.1–0.3; 0.1 (0.1)	*bdl*–0.2; 0.1 (0.1)
Coronene	0.1–0.4; 0.2 (0.2)	*bdl*–0.7; 0.2 (0.2)	*bdl*–0.4; 0.2 (0.1)
Heavy metals			
Sb	0.3–2.0: 0.9 (0.5)	0.1–1.7: 0.6 (0.5)	0.1–1.66: 0.55 (0.4)
As	0.2–1.9: 0.5 (0.4)	0.1–5.8: 0.9 (1.2)	0.2–4.69: 1.15 (1.1)
Be	*bdl*–0.1: 0.1 (0.0)	*bdl*–0.1: 0.1 (0.1)	*bdl*–0.1: 0.1 (0.1)
Cd	0.1–0.2: 0.1 (0.1)	0.1–0.3: 0.1 (0.1)	0.1–0.6: 0.2 (0.1)
Cr	0.9–2.0: 1.4 (0.3)	0.9–2.8: 1.4 (0.5)	0.7–3.5: 1.6 (0.7)
Co	0.1–0.4: 0.2 (0.1)	0.1–0.5: 0.2 (0.1)	0.1–1.4: 0.43 (0.3)
Pb	1.0–4.7: 2.7 (1.2)	0.7–10.9: 2.9 (2.3)	0.9–16.3: 6.16 (3.8)
Mn	3.0–28.1: 14.2 (7.7)	3.2–42.3: 20.1 (11.7)	3.0–100: 31.3 (23.8)
Hg	*bdl*–0.9: 0.1 (0.2)	*bdl*–0.4: 0.1 (0.1)	*bdl*–0.8: 0.1 (0.2)
Ni	0.4–1.6: 0.8 (0.3)	0.3–1.6: 0.8 (0.3)	0.3–3.7: 1.3 (0.8)
Se	0.1–0.5: 0.1 (0.1)	0.1–0.6: 0.1 (0.1)	0.1–0.5: 0.2 (0.1)

bdl: below detection limit.

concentrations. Figs. 2(a)–2(c) presents the comparison of the mean concentrations for the measured air toxics with the levels of air toxics at 50 sites around the U.S. and the USEPA's RfC and URE values (when available) (USEPA, 2008). Note that for PAHs and heavy metals, only wintertime concentrations were measured in this study as compared to annual averages in the NATTS/UATP network.

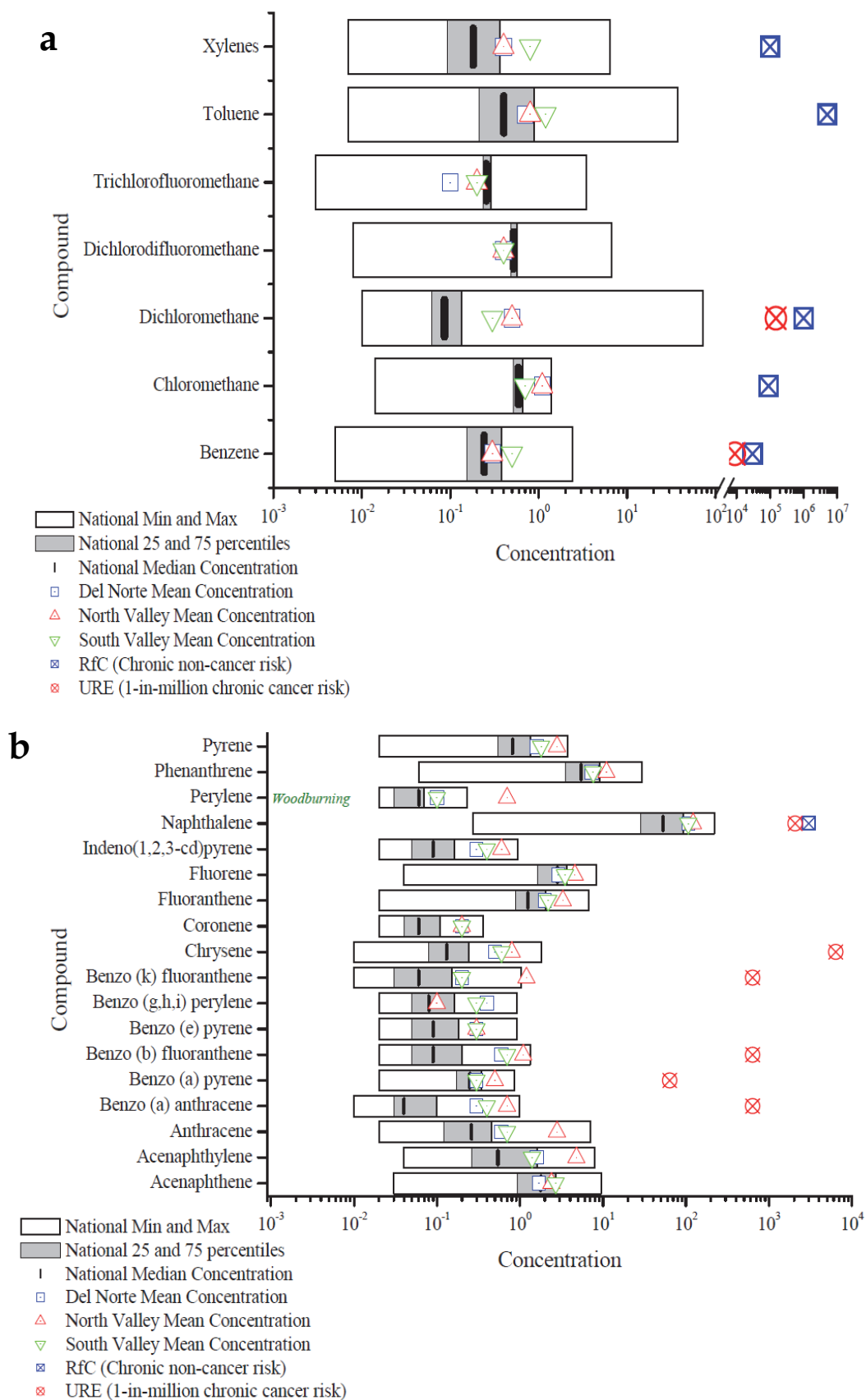The total concentrations of the VOCs measured in the

Fig. 2. Comparison of mean concentrations of VOCs (ppbv) (a), PAHs (ng/m³) (b) and heavy metals (ng/m³) (c) measured at the three sites with those measured at NAATS/UATP sites and the RfC (non-cancer chronic inhalation) and URE (cancer chronic inhalation) levels.
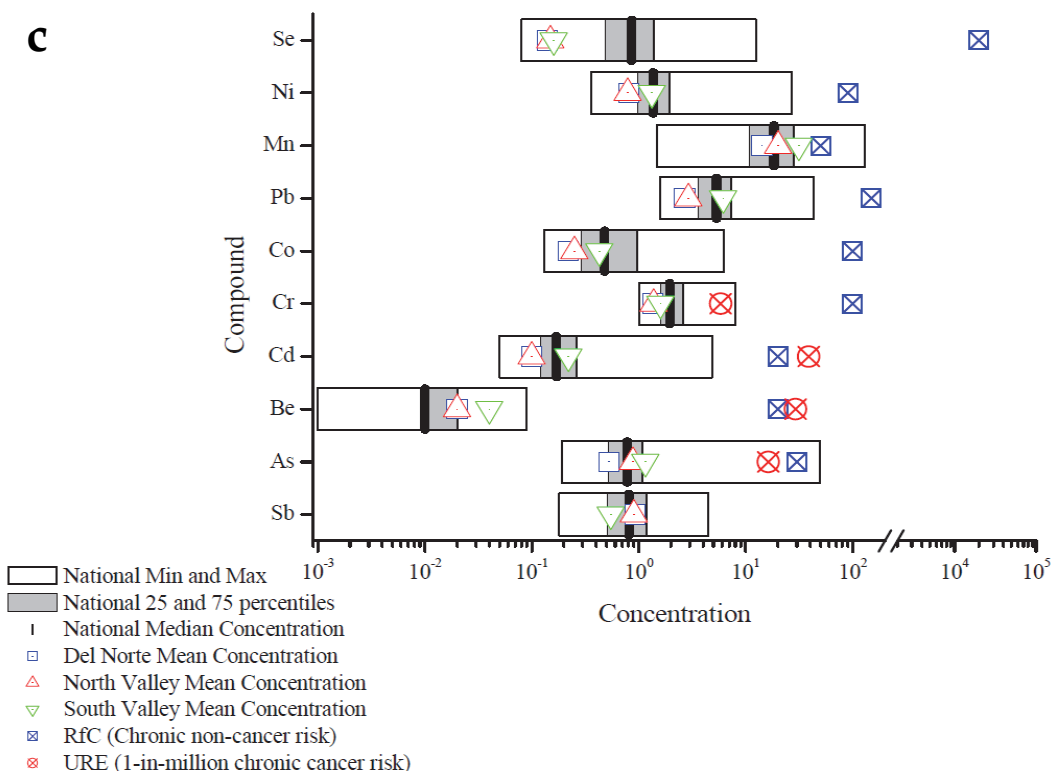
Fig. 2. *(continued).*

collected samples were from 0.1 to 18.7 ppbv at Del Norte (R), from 0.1 to 8.7 ppbv at North Valley (S1) and from 0.1 to 15.4 ppbv at South Valley (S2). Toluene was the major component of unsaturated hydrocarbons, while chloromethane and methylene chloride dominated the fraction of chlorinated hydrocarbons. The levels of 24-hr VOCs in Albuquerque were comparable to those measured nationally and were at least five orders of magnitude lower than the USEPA's RfC and URE threshold values for non-cancer and 1-in-million cancer outcomes (Fig. 2(a)).

The collected total (gas + particulate) PAHs identified in the analyzed samples had total concentrations from 40.0 to 181.1 ng/m^3 at Del Norte (R), from 34.9 to 323.8 ng/m^3 at North Valley (S1) and from 18.0 to 228.6 ng/m^3 at South Valley (S2). For all PAHs, ambient levels measured in South Valley were up to three times higher than those measured at Del Norte, while the differences between North Valley and Del Norte were more significant (more than five times for anthracene and cyclopenta[cd]pyrene). Naphthalene was the dominant PAH, representing more than 70% of total PAHs concentrations. The lowest concentrations were measured for coronene and dibenz[a,h]anthracene. For PAHs, the concentrations measured at the three locations were within the range of those measured elsewhere in the U.S. and significantly lower than threshold values for chronic non-cancer and cancer health effects (Fig. 2(b)). Elevated concentrations of perylene, anthracene, acenaphthylene and benzo(k)fluoranthene at the North Valley site are probably associated with increased emissions from wood burning in the winter (perylene is a tracer of wood burning and is not

emitted from fossil fuel combustion).

The heavy metals measured in the collected samples had total concentrations from 6.1 to 39.8 ng/m^3 at Del Norte (R), from 6.0 to 52.8 ng/m^3 at North Valley (S1) and from 5.7 to 122.0 ng/m^3 at South Valley (S2). Ambient levels of heavy metals measured in South Valley were up to three times higher than those measured at Del Norte, while comparable concentrations were measured between North Valley and Del Norte. Mn was the dominant metal, representing more than 60% of total metal concentrations. The lowest concentrations were measured for Hg and Be. The ambient concentrations measured in this study were comparable (or lower) to those measured at NAATTS/ UATP sites and lower than the threshold values for chronic non-cancer and cancer health effects (Fig. 2(c)). Note that heavy metals were only measured during the winter.

PM$_{2.5}$ and Air Toxics Source Attribution

Air toxics are ubiquitous pollutants of the atmosphere and originate primarily from combustion of contemporary and fossil organic material. Table 3 shows a comparison of toluene/benzene, (*m/p*)-xylene/benzene and (*m/p*)-xylene/ toluene ratios in Albuquerque and other major urban areas in the U.S., including data from highway tunnels. In our study, the mean toluene/benzene ratios were 2.02 ± 0.13 at Del Norte (R), 1.88 ± 0.10 at North Valley (S1), and 2.49 ± 0.14 at South Valley (S2). These values were comparable to those estimated for most of the other urban areas and highway tunnels (Zielinska *et al.*, 1994a, b; Gertler *et al.*, 1996; Baldasano *et al.*, 1998; Fraser *et al.*, 1998; Rogak *et al.*,

Table 3. VOCs and PAHs concentration diagnostic ratios measured in Albuquerque, New Mexico, compared to source ratios and other areas.

	T/B	mpXl/B	mpX/T	Fl/(Fl + Py)	BaA/(BaA + CT)	BeP/(BeP + BaP)	IP/(IP + BgP)
This study							
Del Norte (R)	2.02 ± 0.13	1.12 ± 0.07	0.53 ± 0.02	0.57 ± 0.01	0.32 ± 0.03	0.56 ± 0.01	0.47 ± 0.01
North Valley (S1)	1.88 ± 0.10	0.88 ± 0.05	0.46 ± 0.01	0.56 ± 0.01	0.36 ± 0.02	0.49 ± 0.02	0.51 ± 0.01
South Valley (S2)	2.49 ± 0.14	1.34 ± 0.06	0.53 ± 0.02	0.56 ± 0.01	0.34 ± 0.02	0.53 ± 0.02	0.50 ± 0.01
Sources							
Highway tunnels[a]	1.2–1.7	0.8–2.5	0.52–2.1				
Diesel engines[b]				0.60–0.70	0.38–0.64	0.29–0.40	0.35–0.37
Gasoline engines[b]				0.40	0.43	0.60–0.80	0.18
Wood combustion[c]				0.74	0.56	0.48	
Crude oil[d]				0.18 ± 0.06	0.16 ± 0.12	0.87 ± 0.10	
Used motor oil[d]				0.36 ± 0.08	0.50	0.64 ± 0.10	0.25 ± 0.05
ETS[e]				0.46 ± 0.03	0.17 ± 0.03	0.34 ± 0.13	0.65 ± 0.13
Ambient air							
U.S. Cities[f]	0.7–2.7	0.3–1.3	0.2–1.6				
Urban aerosol[g]				0.54 ± 0.06	0.34 ± 0.02	0.70 ± 0.02	0.30 ± 0.02
Suburban aerosol[g]				0.54 ± 0.17	0.31 ± 0.09	0.73 ± 0.15	0.59 ± 0.27
Rural aerosol[g]				0.75 ± 0.11	0.20 ± 0.09	0.79 ± 0.09	0.61 ± 0.04
Urban and wood burning[c]				0.57 ± 0.08	0.41 ± 0.01	0.41 ± 0.08	0.47 ± 0.02

[a] Gertler *et al.*, 1996.
[b] Rogge *et al.*, 1993a.
[c] Tsapakis *et al.*, 2002.
[d] Sicre *et al.*, 1987.
[e] Kavouras *et al.*, 1998.
[f] Zielinska and Fujita, 1994a.
[g] Kavouras and Stephanou, 2002.

1998). The (*m/p*)-xylene/benzene and (*m/p*)-xylene/toluene ratios varied from 0.88 ± 0.05 to 1.34 ± 0.06 and from 0.46 ± 0.01 to 0.53 ± 0.02, respectively, and fall within the range of those computed for the other urban areas, providing evidence of the substantial contribution of automobile emissions on aromatic VOCs levels in all monitoring locations within Albuquerque.

Emissions from gasoline and diesel-powered vehicle exhaust, fugitive sources, unburnt oil residues, environmental tobacco smoke (ETS), wood burning and industrial sources are the major sources of PAHs (Rogge *et al.*, 1993a, b, c, 1994, 1997; Kavouras *et al.*, 1998). PAH concentration diagnostic ratios (characteristic of anthropogenic emissions) are used to reconcile their presence in the atmosphere with potential emission sources. The PAHs concentration diagnostic ratios included: (i) Fluoranthene to (Fluoranthene and Pyrene) [Fl/(Fl + Py)]; (ii) Benzo[a]anthracene to (Benzo[a]anthracene and Chrysene/Triphenylene (BaA/(BaA + CT)]; (iii) Benzo[e]pyrene to (Benzo[e]pyrene and Benzo[a]pyrene) [BeP/(BaP + BeP)] and; (iv) Indeno[1,2,3-cd]pyrene to (Indeno[1,2,3-cd]pyrene and Benzo[ghi]perylene) [IP/(IP + BgP)].

Typical values for emissions sources, including the values measured in this study and in other urban, suburban, and rural locations are presented in Table 3. Note that values obtained from previously published research were computed for particulate PAHs, while total PAHs (particulate and gas phase) were measured in this study. For PAHs from naphthalene to anthracene, more than 90% were in the gas phase; more than 50% of PAHs from fluorene to chrysene were usually in the gas phase; and more than 90% of heavier PAHs (from benzo[e]pyrene to coronene) were associated with particulate matter. The mean [Fl/(Fl + Py)] ratio was similar for the three sites (from 0.56 ± 0.01 to 0.57 ± 0.01) and comparable to those computed from emissions from diesel and gasoline vehicles. In addition, [BaA/(BaA + CT)] mean values ranged from 0.32 ± 0.06 to 0.36 ± 0.02 and were similar to those calculated for diesel engines. For these two diagnostic ratios, the values were comparable to those measured in other urban and suburban locations, underscoring the contribution of traffic-related activities. The mean [BeP/(BeP + BaP)] obtained in our study values (from 0.49 ± 0.02 to 0.56 ± 0.02) compared with the corresponding values for used motor oil residues and wood combustion. The mean [IP/(IP + BgP)] values (from 0.479 ± 0.01 to 0.51 ± 0.01) were higher than those measured for cars and diesel emissions, but comparable to those observed in suburban locations and in Temuco, Chile (Tsapakis, *et al.*, 2002). The later urban area was characterized by a significant contribution of wood burning emission to $PM_{2.5}$ and PAHs concentrations. The analysis of concentration diagnostic ratios in Albuquerque indicated a mixed origin for contributions from traffic and wood burning.

Fig. 3(a)–3(b) show the relative distribution of emissions factors for three different types of woodstoves and concentrations of PAHs at the three sites, respectively. Emission factors were retrieved from USEPA AP-42 Chapter 1.10 Residential Wood Stoves (US EPA, 1996). Catalytic and non-catalytic woodstoves decrease emissions using a ceramic catalyst coated with a noble metal and by mixing the smoke plume with fresh, preheated makeup air that enhances further combustion, respectively. The total emission factor for PAHs presented in Fig. 3(a) were 201.4 g/ton for conventional woodstoves, 122.3 g/ton for non-catalytic woodstoves and 100.9 g/ton for catalytic woodstoves. A comparison among the three woodstove types showed that there were significant qualitative differences for acenaphthylene (about 50% of conventional woodstove emissions and 30% for catalytic woodstove emissions), benzo(a)anthracene (5 and 10% for conventional and catalytic woodstoves emissions, respectively, and negligible amounts for non-catalytic woodstoves) and phenanthrene (about 35% for non-catalytic woodstoves and less than 25% for the other two types). The relative distribution of PAHs measured at the three sites (without naphthalene) showed that phenanthrene (more than 30%) was the dominant PAH followed by fluorine (more than 10%). The relative distribution profiles for all three sites were comparable among each other and similar to that observed for non-catalytic woodstoves.

Fig. 4 shows the profiles of the five retained factors. The factors were attributed to specific types and/or sources of fine aerosol based on the associations of specific chemical species with each factor. The estimated and measured mass and elemental concentrations were in an acceptable agreement (R = 0.83; %RSME = 17%), indicating that the five factors explained most of the variability of measured concentrations. The contributions of the five factors to $PM_{2.5}$ mass for the January 2007–February 2009, the two IMPs (February and June 2008) and the Air Toxics (December 2008– February 2009) periods are depicted in Fig. 5.

The strong correlations of the first factor with EC and OC indicated the contribution of primary particulate emissions from automobiles (Fig. 4(a)). The EC/OC ratio values (0.15 ± 0.01) were lower than those determined for traffic-dominated urban aerosol, indicating the possible contribution of smoke aerosol that generally contains less EC than OC. Furthermore, factor loadings of S, SO_4^{2-}, Pb, Al, Ni and V were also associated with motor vehicle emissions. Automobile emissions were responsible for 5.9% during the January 2007 to February 2009 period, 3.9% in February 2008, 11.9% in June 2008, and 7.9% during the December 2008 to February 2009 period of $PM_{2.5}$ mass, respectively (Fig. 5).

Road dust was identified as a source of $PM_{2.5}$ in Albuquerque. Most of the measured concentrations of crustal elements Al, Si, Ca, Fe and Ti were associated with this source (Fig. 4(b)). In addition, fractions of organic carbon, elemental carbon, and other elements, such as Br, As, Cu, Zn and K, were also associated with this factor because of the resuspension of urban soil containing these elements. The mean ratios for Al/Si, Al/Ca and K/Fe were 0.30 ± 0.01, 0.61 ± 0.03, and 0.98 ± 0.15, respectively. These ratios were somewhat lower, but comparable, to those estimated for samples collected at the Interagency Monitoring of Protected Visibility Environments (IMPROVE) sites in the western United States (Al/Si: 0.31 to 0.43, K/Fe: 0.67 to 0.78, Al/Ca: 1.4 to 1.7) when soil dust was the major component of particulate matter. Water-soluble ions (Na^+,
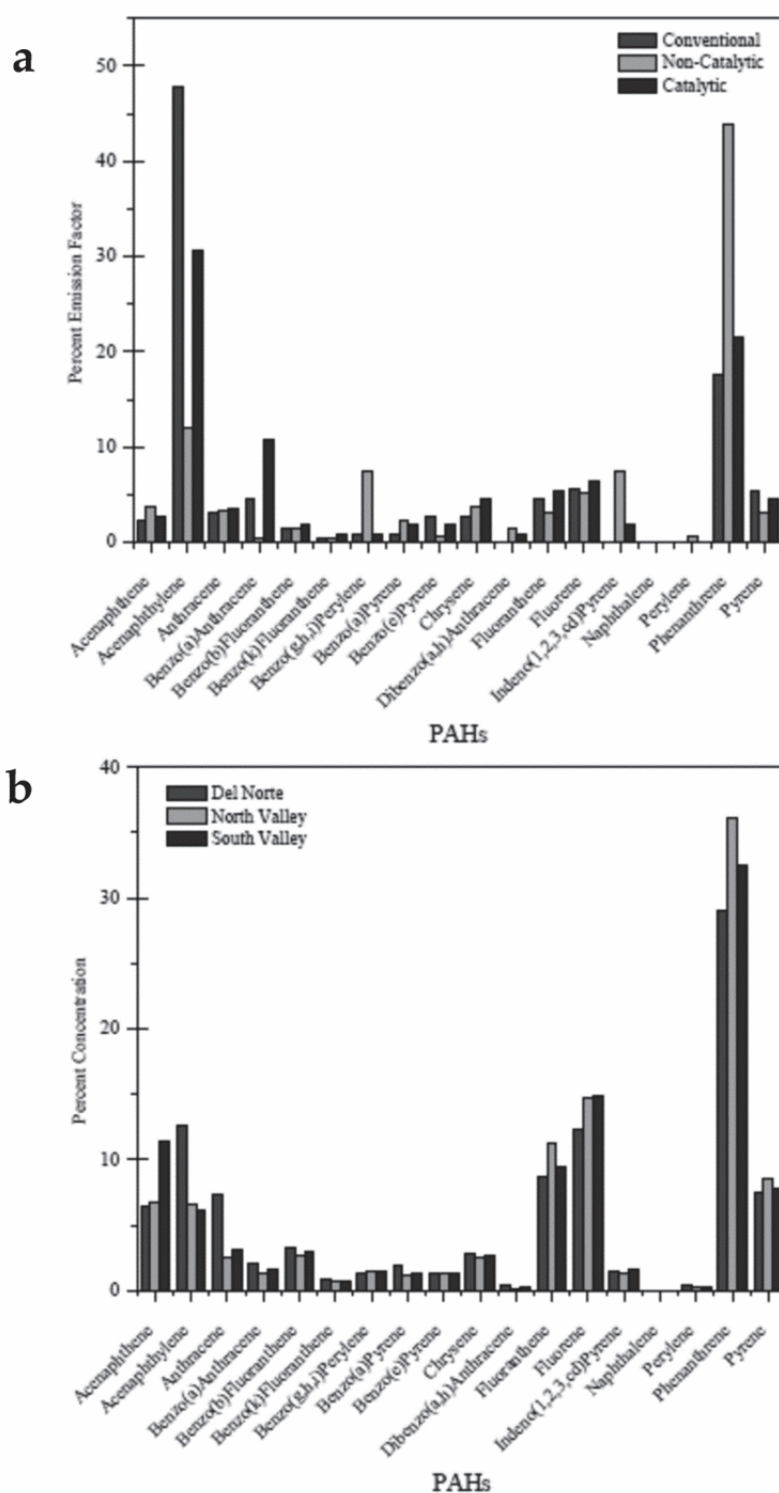
Fig. 3. Relative distribution of PAHs emitted from different types of woodstoves (a) and measured in Albuquerque, NM (b).

K^+, and SO_4^{2-}) may be attributed to road-sanding material (Kuhns *et al.*, 2003). The contribution of road dust to $PM_{2.5}$ mass represented 10.7% for the January 2007 to February 2009 period. During the two IMPs, road dust accounted for 4.9% in February 2008 and 15.3% in June 2008 (Fig. 5). This is due to dry conditions in the spring and summer that favor the resuspension of dust by vehicles. Mineral particles

in the summer may be associated with elevated emissions from unpaved roads.

The third factor, biomass burning, includes a mixture of secondary aerosols, including some from traffic emissions (Fig. 4(c)). Biomass burning includes wildfires, prescribed burning and agricultural fires, as well as wood burning. CO, CO_2, NO_x, light hydrocarbons, CH_3Cl, CH_3Br, PAHs,
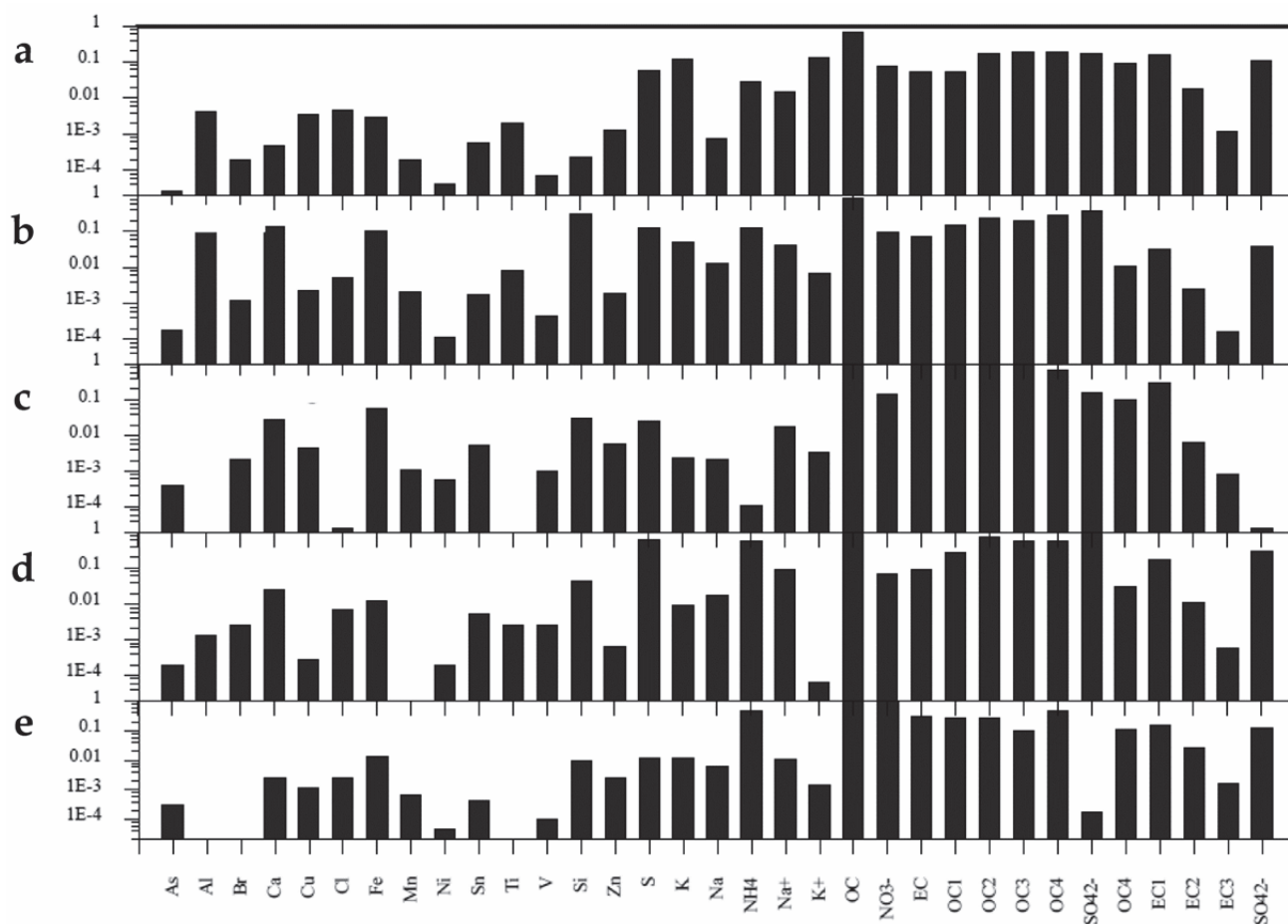
Fig. 4. Source composition profiles for the five retained $PM_{2.5}$ factors at Albuquerque, NM.
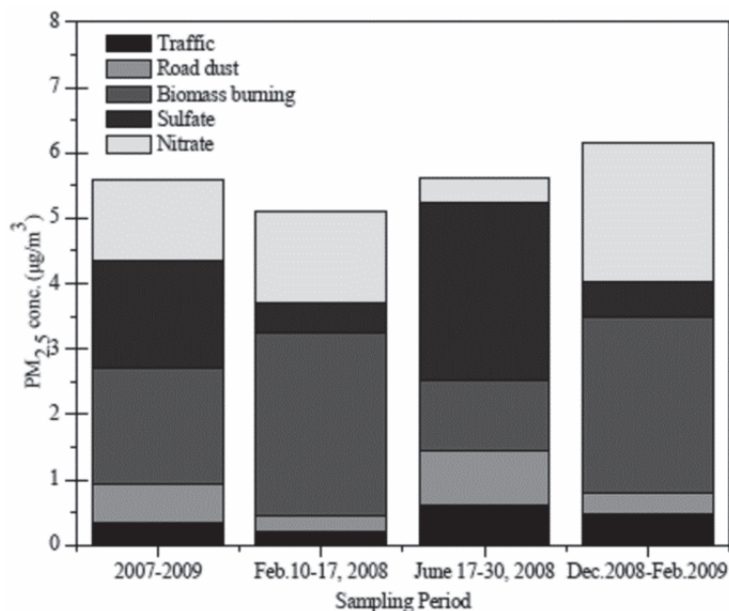

Fig. 5. Modeled mean contributions of sources to $PM_{2.5}$ mass based on chemical speciation data collected at Albuquerque, NM.

formaldehyde, formic and acetic acids, methanol, R-hydroperoxides, NH_3, HCN, acetonitrile, nitric acid, peroxyacetylnitrates (PAN), SO_2 and particulate matter have been reportedly directly emitted during the flaming, smoldering and pyrolysis stages. In addition to primary emissions, secondary organic aerosols (SOA) are formed

through the condensation of hot vapors and the gas-to-particle conversion of gas-phase precursors to low volatility compounds. The highest contributions of these sources were identified during the January to February 2009 period. This source contributed 31.9% during the January 2007 to February 2009 period and 43.5% of $PM_{2.5}$ mass during the December 2008 to February 2009 period. During the two IMPs, this source accounted for 54.8% and 59.1% (Fig. 5). Assuming that domestic wood burning is the dominant source in winter and wildfires/prescribed burning are observed in summer, it is confirmed that emissions from wood burning for domestic heating make a significant contribution to particulate matter concentration in the Albuquerque area.

A separate factor for secondary SO_4^{2-} aerosol was resolved. This source showed strong associations with elemental S, NH_4^+, EC, and OC (Fig. 4(d)). Furthermore, a small fraction of Al, Ni and V was associated with this factor. The contribution of this source to $PM_{2.5}$ was ~29.2% for the January 2007 to February 2009 period and 8.9% for the December 2008 to February 2009 period. The contribution of secondary sulfate was 9.5% in February 2008 and 48.4% in June 2008, indicating an important seasonal variation (Fig. 5). Sulfur (S) was in the form of SO_4^{2-} with a SO_4^{2-}-to-sulfur ratio of 3.00 ± 0.03. The mean NH_4^+/SO_4^{2-} and $NH_4^+/(NO_3^- + 2 SO_4^2)$ molar ratios for the two periods were 2.77 ± 0.15 and 0.77 ± 0.01, respectively, suggesting that $SO4^{2-}$ aerosols were mostly in the form of $(NH_4)_2SO_4$ in the winter and $(NH_4)HSO_4$ in the summer, while NO_3^- appeared to be partially neutralized by the NH_3 (Malm *et al.*, 2004).

Finally, the fifth factor was attributed to particulate NO_3^- with high loadings of NH_4^+ and SO_4^{2-} and minor loadings of the volatile fractions of organic carbon (OC1) and elemental carbon (Fig. 4(e)). NO_3^- appeared to be a major component of fine particulate matter in the winter, whereas only minimal amounts were observed in the spring, fall and summer. For the entire measurement period, NO_3^- accounted for 22.3% of $PM_{2.5}$. The contribution of particulate NO_3^- to $PM_{2.5}$ mass during February 2008 and December 2008 to February 2009 were 26.9% and 34.4%, but only 6.6% in June 2008 (Fig. 5).

The associations between air toxics concentrations and $PM_{2.5}$ source contributions during the December 2008 to February 2009 period were examined using regression analysis. Table 4 shows the contributions of $PM_{2.5}$ sources to air toxics concentrations, estimated and measured concentrations, and the %RSME based on the regression of air toxics concentrations against the $PM_{2.5}$ source contributions (all contributions are included). We previously applied the same approach to apportion the contributions of individual VOCs sources on ozone concentrations in Pacific Northwest (Vargas *et al.*, 2014). PAHs are present in both emissions and ambient air in the gas and particulate phases. A fraction of them are directly emitted from tailpipes in the particulate phase, which was in agreement with the good correlations with $PM_{2.5}$ mass and direct traffic emissions in this study. The very good correlations between PAHs and the nitrate source, a source that is closely

associated with traffic emissions of NO and NO_2 may be due to emissions of volatile PAHs such as naphthalene and phenanthrene in the gas phase and partition between gas and particulate phase. It is further corroborated by the good correlations of aromatic hydrocarbons (benzene, toluene, xylenes and ethylbenzene) with $PM_{2.5}$, traffic contribution and nitrate contribution. The significant role of traffic was also identified by the moderate loadings of aromatic hydrocarbons and heavy metals with road dust.

The strong associations between benzene alkylated derivatives and the biomass burning/SOA source indicated they played an important role of SOA formation in the region as they react with OH radicals from 10 to 50 times faster than benzene. Finally, sulfate source, a regional source, was only correlated to mercury and trichlorotrifluoromethane, contaminants with more regional and global characteristics. The largest quantities of PAHs were associated with the nitrate source, followed by biomass burning and direct particulate emissions from traffic sources. This indicated that nitrate in Albuquerque is largely associated with local traffic emissions as compared to other regional sources and that gas-to-particle conversion of hot PAHs vapors is the dominant pathway for the accumulation of PAHs in the particulate phase. A fraction of PAHs were also associated with road dust, suggesting that road dust was contaminated by vehicle exhausts and oil residues. Road dust was also responsible for less than 20% of heavy metals concentrations. Biomass burning appeared to be a major contributor to air toxics, especially volatile (form naphthalene to pyrene) PAHs. Aromatic hydrocarbons were associated with both traffic and biomass burning emissions. In general, good agreements between estimated and measured concentrations were observed. The %RSME varied from 0 to 55% with the highest values being computed for chlorinated hydrocarbons.

Risk Assessment

Exposures to toluene and methylene chloride were associated with very low AHQ values. The mean AHQ values for benzene and mixed xylenes were 0.0430 and 0.0275, respectively, however, these values are typical in urban communities dominated by traffic emissions and similar to those estimated for other urban areas in the U.S. Figs. 6(a)–(d) presents the cumulative frequencies of AHQ values for the air toxics. Because exposure concentrations are not spatially uniform, these plots show that about 70 to 80% of the population residing in Albuquerque may be exposed to concentrations of benzene, toluene and xylenes with AHQ below the average for the entire population, while the remaining 20%–30% may be exposed to concentrations that are above the local average but are below the level of health concerns. The similarities between these compounds are probably because of the large contribution of traffic to their levels. For methylene chloride, about 60% of the population will be exposed to levels with AHQ higher than the average. The annualized cancer risks for the two air toxics are low, indicating that for a lifetime of 70 years, the cumulative cancer risk for exposures to the four air toxics would be less than 1-in-a-million. These estimates were in the same range with those obtained in NATTS sites.

Table 4. Contributions of PM$_{2.5}$ sources to concentrations of heavy metals (in ng/m^3), PAHs (in ng/m^3) and VOCs (in ppbv) during the December 2008–February 2009 period based on chemical speciation data collected at Del Norte (R).

|	Traffic	Road dust	Biomass burning/SOA	Sulfate	Nitrate	Estimated	Computed	%RSME
Antimony	0.08 ± 0.02	0.19 ± 0.03	0.93 ± 0.11	-	-	0.99 ± 0.16	1.1 ± 0.18	−1
Arsenic	0.08 ± 0.1	0.08 ± 0.12	-	-	0.43 ± 0.2	0.59 ± 0.42	0.67 ± 0.19	−22
Cadmium	0.01 ± 0.01	0.02 ± 0.01	0.03 ± 0.05	-	0.02 ± 0.02	0.07 ± 0.09	0.11 ± 0.01	−7
Chromium	-	0.12 ± 0.08	0.44 ± 0.31	-	0.08 ± 0.11	0.64 ± 0.5	1.46 ± 0.12	−2
Cobalt	-	0.06 ± 0.02	-	-	0.03 ± 0.03	0.09 ± 0.05	0.22 ± 0.03	−12
Lead	-	0.62 ± 0.17	-	-	0.8 ± 0.25	2.68 ± 0.41	2.92 ± 0.39	−4
Manganese	-	5.27 ± 1.34	-	-	1.55 ± 1.96	6.82 ± 3.3	13.96 ± 2.61	−18
Nickel	-	0.21 ± 0.05	0.28 ± 0.18	0.1 ± 0.06	-	0.59 ± 0.29	0.84 ± 0.12	−4
Selenium	-	0.01 ± 0.02	-	0.03 ± 0.03	-	0.08 ± 0.08	0.14 ± 0.03	−5
Naphthalene	3.05 ± 7.2	13.94 ± 13.37	53.51 ± 70.22	3.66 ± 15.06	31.74 ± 21.25	105.9 ± 127.1	109.73 ± 16.7	−3
Acenaphthylene	0.7 ± 0.08	0.2 ± 0.1	-	-	1.07 ± 0.19	1.97 ± 0.37	1.81 ± 0.66	−24
Acenaphthene	0.1 ± 0.07	0.04 ± 0.13	0.37 ± 0.67	0.19 ± 0.14	0.34 ± 0.2	1.04 ± 1.21	1.3 ± 0.15	−6
Fluorene	0.34 ± 0.15	0.25 ± 0.28	0.73 ± 1.46	0.31 ± 0.31	0.97 ± 0.44	2.6 ± 2.64	2.99 ± 0.45	−3
9-Fluorenone	0.26 ± 0.09	0.13 ± 0.18	0.54 ± 0.92	0.24 ± 0.2	0.73 ± 0.28	1.9 ± 1.67	1.76 ± 0.33	−4
Phenanthrene	1.13 ± 0.42	0.59 ± 0.78	0.71 ± 4.09	0.89 ± 0.88	2.32 ± 1.24	5.64 ± 7.41	7.22 ± 1.19	−7
Anthracene	0.11 ± 0.04	0.08 ± 0.05	-	-	0.34 ± 0.09	0.53 ± 0.18	0.66 ± 0.15	−8
Retene	0.63 ± 0.13	-	-	-	2.43 ± 0.3	2.36 ± 0.43	3.06 ± 0.96	−6
Pyrene	0.23 ± 0.09	0.18 ± 0.14	0.63 ± 0.72	-	0.41 ± 0.27	1.45 ± 1.22	1.85 ± 0.29	−3
Fluoranthene	0.3 ± 0.13	0.29 ± 0.25	0.55 ± 1.32	0.09 ± 0.28	0.54 ± 0.4	1.77 ± 2.38	2.25 ± 0.35	−3
Benzo[a]anthracene	0.09 ± 0.03	-	-	-	0.16 ± 0.08	0.31 ± 0.11	0.37 ± 0.1	−21
Chrysene	0.15 ± 0.01	0.07 ± 0.02	-	-	0.25 ± 0.03	0.5 ± 0.06	0.6 ± 0.15	−1
Benzo[b]fluoranthene	0.13 ± 0.04	0.08 ± 0.07	0.18 ± 0.34	-	0.34 ± 0.13	0.73 ± 0.58	0.74 ± 0.17	−4
Benzo[k]fluoranthene	0.05 ± 0.01	0.03 ± 0.01	-	-	0.09 ± 0.02	0.17 ± 0.04	0.21 ± 0.05	−7
Benzo[e]pyrene	0.05 ± 0.02	0.05 ± 0.02	-	-	0.16 ± 0.04	0.29 ± 0.08	0.35 ± 0.07	−5
Benzo[a]pyrene	0.08 ± 0.03	0.05 ± 0.05	0.04 ± 0.27	0.01 ± 0.06	0.2 ± 0.08	0.37 ± 0.49	0.35 ± 0.1	−3
Benzo[ghi]perylene	0.04 ± 0.02	0.03 ± 0.03	0.15 ± 0.17	-	0.15 ± 0.06	0.37 ± 0.28	0.4 ± 0.07	−7
Indeno[123,cd]pyrene	0.05 ± 0.03	0.05 ± 0.04	0.06 ± 0.23	-	0.19 ± 0.09	0.35 ± 0.39	0.4 ± 0.08	−10
Coronene	0.02 ± 0.02	0.01 ± 0.04	0.12 ± 0.22	0.01 ± 0.05	0.07 ± 0.07	0.22 ± 0.4	0.2 ± 0.04	−11
Benzene	0.12 ± 0.08	0.02 ± 0.03	0.29 ± 0.16	0.01 ± 0.03	0.1 ± 0.05	0.54 ± 0.35	0.48 ± 0.07	−2
Chloromethane	0.28 ± 0.11	0.28 ± 0.04	-	-	-	1.29 ± 0.15	0.81 ± 0.11	−55
Methylene Chloride	0.22 ± 0.05	-	-	-	-	0.49 ± 0.05	0.5 ± 0.12	−32
Toluene	0.01 ± 0.04	0.01 ± 0.06	1.23 ± 0.25	0.13 ± 0.07	0.21 ± 0.1	1.59 ± 0.52	1.03 ± 0.15	0
m/p-Xylenes	0.05 ± 0.13	0.01 ± 0.05	0.49 ± 0.26	0.06 ± 0.06	0.09 ± 0.08	0.69 ± 0.58	0.6 ± 0.08	−4
o-Xylene	0.03 ± 0.01	0.01 ± 0.01	0.19 ± 0.02	-	-	0.19 ± 0.04	0.21 ± 0.03	0
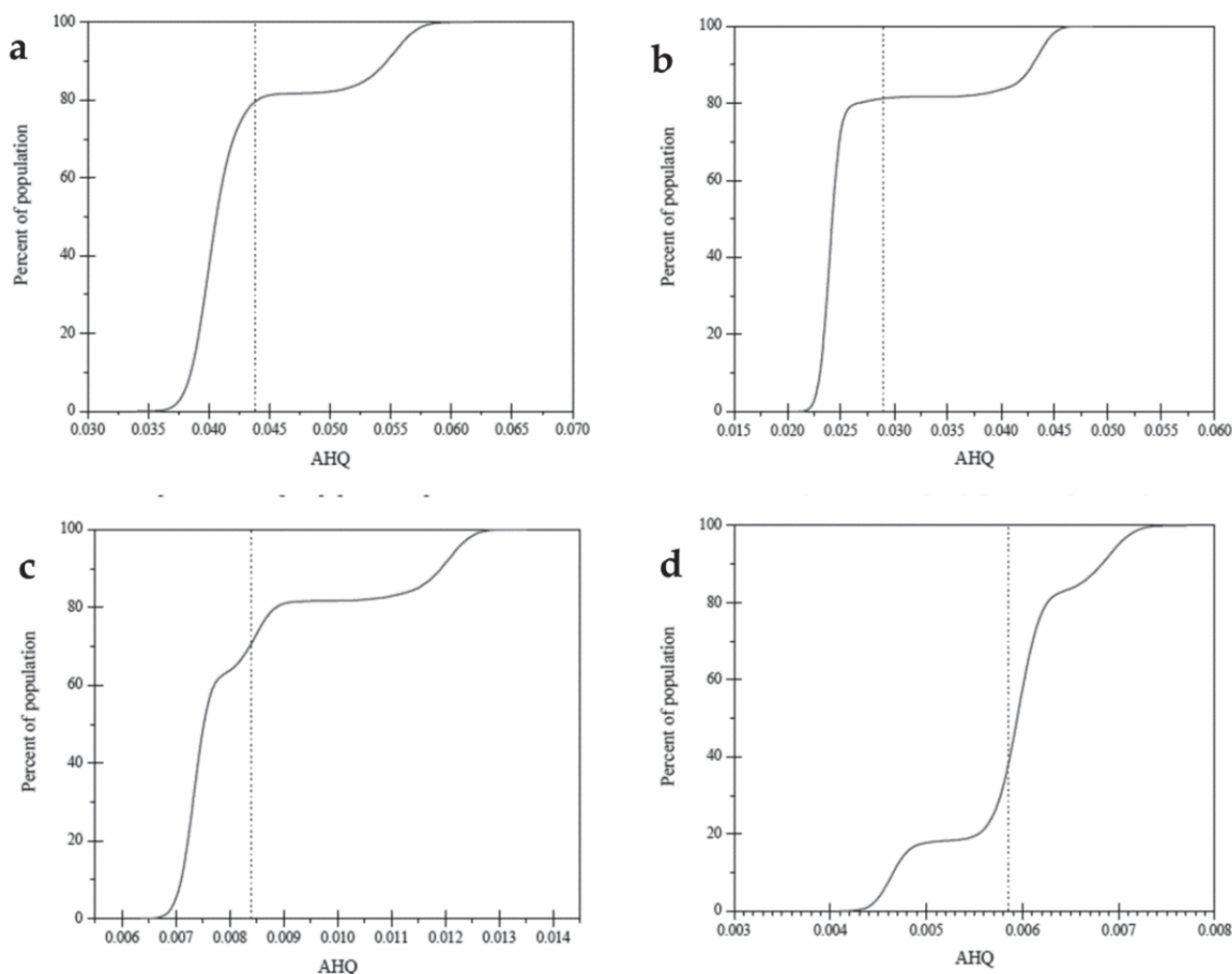
Fig. 6. Cumulative frequencies of AHQ for population exposed to benzene (a), toluene (b), xylenes (c) and methylene chloride (d).

CONCLUSIONS

The concentrations levels, spatiotemporal trends, sources and health risks of air toxics (VOCs, PAHs and heavy metal) in the City of Albuquerque/Bernalillo County were examined. Samples were collected in three locations from September 2007 to August 2008 (only VOCs) and from December 2008 to March 2009 (VOCs, PAHs and heavy metals). Analysis of spatial and temporal trends showed that (1) higher VOCs concentrations were measured during the cold period; (2) lowest VOCs concentrations were measured on weekends as compared to weekdays; (3) there was a strong spatial pattern for VOC. For PAHs and heavy metals, a rather uniform spatial variation was observed. The mean values of the PAHs diagnostic ratios were indicative of a mixed origin from combustion of fossil fuels and biomass burning.

Analysis of chemical speciation data using PMF identified the contribution of five sources to $PM_{2.5}$ particle mass, namely, secondary NO_3^-, secondary SO_4^{2-}, primary emissions from traffic, road dust and secondary organic aerosol-biomass burning. Biomass burning and particulate NO_3^- accounted for most of $PM_{2.5}$ mass during the winter, while SO_4^{2-} was mostly present in the summer. Primary particulate emissions and road dust contributed about 25% of $PM_{2.5}$ mass. Analysis of air toxics concentrations using the source contributions for $PM_{2.5}$ mass indicated that traffic and biomass burning are the major source of VOCs and PAHs, while some quantities of PAHs and heavy metals were also associated with the resuspension of contaminated road dust. The estimated total inhalation non-cancer and cancer risk for benzene, toluene, xylenes and methylene chloride were similar to those estimated for other urban areas in the U.S. and did not indicate significant health risks.

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