HHS Public Access
Author manuscript
Environ Pollut. Author manuscript; available in PMC 2017 July 01.

Published in final edited form as:
Environ Pollut. 2016 July ; 214: 668–679. doi:10.1016/j.envpol.2016.04.085.

Assessment of Indoor and Outdoor PM Species at Schools and Residences in a High-Altitude Ecuadorian Urban Center

Amit U. Raysoni[1], **Rodrigo X. Armijos**[2,3,4,*], **M. Margaret Weigel**[2,3,4], **Teresa Montoya**[5], **Patricia Eschanique**[6], **Marcia Racines**[6], and **Wen-Whai Li**[5]

[1]Department of Public Health Sciences, The University of Texas at El Paso, El Paso, TX, 79968, USA

[2]Department of Environmental Health, School of Public Health, Indiana University, Bloomington, IN 47405, USA

[3]Proyecto Prometeo, Secretaria de Education Superior, Ciencia y Tecnologia (SENESCYT), Quito, Ecuador

[4]Centro de Biomedicina, Universidad Central del Ecuador, Quito, Ecuador

[5]Department of Civil Engineering, The University of Texas at El Paso, El Paso, TX, 79968, USA

[6]Facultad de Medicina, Universidad Central del Ecuador, Quito, Ecuador

Abstract

An air monitoring campaign to assess children's environmental exposures in schools and residences, both indoors and outdoors, was conducted in 2010 in three low-income neighborhoods in Z1(north), Z2(central), and Z3(southeast) zones of Quito, Ecuador - a major urban center of 2.2 million inhabitants situated 2850 meters above sea level in a narrow mountainous basin. Z1 zone, located in northern Quito, historically experienced emissions from quarries and moderate traffic. Z2 zone was influenced by heavy traffic in contrast to Z3 zone which experienced low traffic densities. Weekly averages of PM samples were collected at schools (one in each zone) and residences (Z1=47, Z2=45, and Z3=41) every month, over a twelve-month period at the three zones. Indoor $PM_{2.5}$ concentrations ranged from 10.6±4.9 μg/m^3 (Z1 school) to 29.0±30.5 μg/m^3 (Z1 residences) and outdoor $PM_{2.5}$ concentrations varied from 10.9±3.2 μg/m^3 (Z1 school) to 14.3±10.1 μg/m^3 (Z2 residences), across the three zones. The lowest values for $PM_{10-2.5}$ for indoor and outdoor microenvironments were recorded at Z2 school, 5.7±2.8 μg/m^3 and 7.9±2.2 μg/m^3, respectively. Outdoor school PM concentrations exhibited stronger associations with corresponding indoor values making them robust proxies for indoor exposures in naturally ventilated Quito public schools. Correlation analysis between the school and residential PM size fractions and the various pollutant and meteorological parameters from central ambient monitoring (CAM) sites suggested varying degrees of temporal relationship. Strong positive correlation was

*Corresponding Author: Department of Environmental Health, School of Public Health, Indiana University, 1025 E 7th St, Bloomington, IN 47405, USA; ; Email: rarmijos@iu.edu; Tel: +1 812 856 4731

observed for outdoor $PM_{2.5}$ at Z2 school and its corresponding CAM site (r=0.77) suggesting common traffic related emissions. Spatial heterogeneity in $PM_{2.5}$ concentrations between CAM network and sampled sites was assessed using Coefficient of Divergence (COD) analysis. COD values were lower when CAM sites were paired with outdoor measurements (< 0.2) and higher when CAM and indoor values were compared (> 0.2), suggesting that CAM network in Quito may not represent actual indoor exposures.

Keywords

Air pollution; particulate matter; public schools; residences; indoor; outdoor; Quito

1.0 INTRODUCTION

The world is becoming increasingly urbanized. It is estimated that by 2050, 2.5 billion people or two-thirds of the planetary population will live in urban areas with most of the increase projected to occur in low- and middle-income countries (LMICs) (UN, 2014; UNDP 2010). In Latin America, urbanites already comprise nearly eight of every ten residents (UN, 2014). Many heavily populated urban centers, especially those in the Andean region of Latin America and certain Asian countries, are situated at high altitude (> 2500 m). The number of high-altitude residents who reside in densely packed, heavily polluted urban centers is steadily rising due to increasing rural-to-urban migration and industrialization (UNEP, 2005).

Urbanization can lead to the deterioration of air quality, smog formation, and pollution-related cardiorespiratory and other adverse health effects (Romieu et al., 2012). This situation is particularly problematic in many urban centers in Latin America (Bell et al., 2006) and other LMICs where air quality is exacerbated by congested traffic, weak vehicular emission regulations, poorly maintained roads, older vehicle fleets, reliance on gasoline and diesel fuels with a high sulfur content, and mountainous topography promoting temperature inversions and pollutant entrapment (Armijos et al., 2015; WHO, 2014; HEI 2010; Bogo et al., 2003; Brachtl et al., 2009; Gee and Sollars, 1998; Wang et al., 2003). In addition, the oxygen content of the air in high-altitude urban centers is much lower than that of the sea level. This results in less efficient combustion and greater pollutant release (Armijos et al., 2015).

It is estimated that 600 million urban inhabitants worldwide are currently exposed to high levels of particulate matter (PM) and other air pollutants (Han and Naeher, 2006). The effects of PM emissions on the respiratory and other health outcomes of children have been the focus of many studies during the past two decades (Gehring et al., 2013; Rice et al., 2015; Laborde et al., 2015; Wang et al., 2015). Children appear to be more vulnerable than adults to the adverse health effects of PM and other air pollutants because of their smaller airways and lung size, increased baseline ventilation rates, propensity to mouth breathe, and greater time spent running, jumping, and other aerobic play activities which expose them to greater pollutant loads penetrating deeper into lung tissues (Wright and Brunst, 2013; Bateson and Schwartz, 2008). It is indicative that higher exposure to urban air pollutants is associated with increased blood markers of oxidative stress, systemic inflammation, and endothelial dysfunction in children (Wu et al., 2015; Calderon-Garciduenas et al., 2009; Kelishadi et al., 2014; Poursafa et al., 2011).

Prior studies have documented the adverse impact of traffic-related air pollutants on cardiovascular health in adults (Adar et al., 2013; Araujo 2011; Hoffman et al., 2007; Kunzli et al., 2010). Emerging evidence suggests that close residential proximity to traffic promotes arterial remodeling in children. Iannuzi et al. (2010) reported that Italian schoolchildren living 30–300 meters from a major roadway had increased arterial stiffness. More recently, Armijos and colleagues (2015) reported that long-term exposure to traffic-related pollutants for residents living in close proximity (<100 meters) to highly trafficked roadways promotes ultrasound-detectable arterial remodeling measured, as evident in the increased carotid intima-media thickness (cIMT), in healthy schoolchildren living in Quito, Ecuador. However, this research work analyzed only the contribution of residential traffic exposure indicators to cIMT (i.e. residential distance to traffic, distance-weighted traffic density), rather than PM measured at homes and school environments. Furthermore, the analysis of environmental variables suggested that naturally ventilated homes might have allowed for free passage of traffic-related pollutants into interior residential spaces (Armijos et al., 2015). Previous investigations in the Quito Metropolitan District (QMD) have also documented an association between high carbon monoxide levels and elevated carboxyhemoglobin (COHb) levels in a cohort of school children (Estrella et al., 2005). Brachtl and colleagues (2009) studied the spatial and temporal variations in polycyclic aromatic hydrocarbons (PAHs) at near roadway sites and recorded a three to six fold increase of PAHs concentrations than that measured at low-traffic residential sites.

Thus, in order to better understand the environmental health indicators that best capture the cardiorespiratory and other health effects of traffic-related PM emissions in urban environments, we conducted assessments of PM pollution in multiple microenvironments, i.e., indoors and outdoors at subject homes and schools. We were also interested in comparing our microenvironmental measurements at these sites with those at nearby central ambient monitoring (CAM) stations since CAM-derived exposure estimates may not accurately reflect the actual exposures of children (Raysoni, et al., 2011, 2013). Another aim of our study was to compare the gradient in pollutant concentrations in urban Quito neighborhoods with varying traffic densities.

2.0 STUDY DESIGN AND METHODS

2.1 The City of Quito

The present study was conducted in Quito, the capital city of Ecuador. The city is located in a long narrow high-altitude valley at 2850 meters in the Guayllabamba river basin between the eastern and western chains of the Andes Mountains at approximately 0°13′23″ S and 78° 30′ 45″W. Oxygen levels in this high-altitude city are 27 percent lower than at sea level resulting in less efficient combustion and greater vehicular emissions. The city experiences around 2,000 hours of sunlight per year. It has a subtropical highland climate characterized by year round spring-like weather. Average temperatures range from a nighttime low of 9.3°C to 18.7°C. The city experiences two seasons: a dry season that lasts from June through

September and a wet season that lasts from October through May. Temperature inversions are common in this city due to strong solar radiation during the day and altitude-enhanced rapid cooling at night.

Traffic air pollution is a major health concern for the city of Quito (UNEP, 2011). An estimated 2.2 million people reside in the QMD, and of these, approximately 25% live in close proximity to heavily travelled arterial roads. Vehicle ownership in the QMD has tripled in the last two decades from 61 vehicles/1000 people (1991) to 187 vehicles/1000 people (2008) with 46% of all PM annual emissions apportioned to traffic (UNEP, 2011). The number of registered motor vehicles was reported to be 410,000 by the end of 2011 and 1,302 industries were classified as having a 'significant environmental impact' (Secretaria de Ambiente, 2011). In addition, affordable fossil fuels (as subsidized by the government) significantly incentivized the use of motor vehicles (Jurado and Southgate, 1999). The sulfur content of diesel (500 ppm) and gasoline (2000 ppm) sold in Quito, the highest of any Latin American country (UNEP, 2011), further impairs the quality of the air people breathe in the city.

2.2 Site Characteristics

The PM monitoring campaign was conducted at schools and residences in three low income neighborhoods located in the north, central, and southeast of the QMD. These zones were selected based on five-year (2003–2007) PM_{10} records collected from the three neighborhood CAM sites (Ecogestion, 2005; CORPAIRE, 2007). Each of the three selected neighborhood zones encompassed a 5-mile radius from its municipal CAM site. Additional data on neighborhood traffic density, traffic patterns, and population characteristics were taken into consideration for the zone selections.

The Cotocollao neighborhood in the northern part of QMD, experiencing medium traffic density, was designated as Z1 zone. El Camal, located in the central part of QMD and experiencing heavy traffic at all times, was designated as zone Z2 whereas the southeastern neighborhood of Los Chillos was selected as Z3 zone. This zone experienced much less traffic compared to the other two zones. We posited that the exposure burden of the study subjects would differ based on the varying levels of traffic densities in each of these three zones. Figure 1 displays the locations of the three selected zones along with major highways, arterial and surface roads. The site characteristics for the schools and residences in the three zones are detailed in Table 1.

One public elementary school with a minimum of 150 students located within a 5-mile radius of neighborhood CAM site was selected in each zone. The three schools, with two or three story buildings, shared common features such as a principal outdoor play area consisting of an inner cement courtyard surrounded by classrooms, each of which had direct access to an outdoor hallway. All rooms in the three schools were naturally ventilated. Indoor samples at Z1 school were collected from two classrooms located on the first and third floors. These classrooms face the inner courtyard where physical education activities took place throughout the school day resulting in heavy foot traffic and resuspension of particles. In the Z2 school, in addition to the classroom, PM samples were collected from a conference room. In the Z3 school, PM sampling was conducted in a computer classroom and the school director's office in addition to a regular classroom.

A set of subject homes was randomly selected for indoor and outdoor PM sampling. A subject home is defined as a non-smoker home located within a 5-mile radius of the neighborhood CAM site. All households relied on natural ventilation and opened their windows on average 7 ± 4 hours per day. A typical house was constructed of cement block, concrete, and steel/iron, and consisted of a kitchen/dining room, living room, two bedrooms, and a bathroom. Bottled liquefied petroleum gas (LPG) was the only cooking fuel used by households. The QMD neighborhoods in which participating subject homes were located included a mix of residential housing and commercial businesses, i.e., small grocery stores ("micromercados"), restaurants, bakeries, street food vendors, gasoline stations, LPG depositories, and furniture/carpentry stores, and other small shops.

2.3 Sampling Plan

Seven day indoor and outdoor PM sampling was conducted at the three schools and several households in the three zones for a 12-month period during 2010. In each zone, the school site and four (indoor and outdoor) residential sites served as controlled sites throughout the year-long monitoring period. Other residential sites varied every month between each zone, and the sampling in some homes was repeated once or twice during the study period. School indoor and outdoor PM sampling was paired. However, due to some unanticipated physical and logistical constraints, it was not always feasible to pair residential indoor and outdoor samplers. Sampling was not performed concurrently at all three exposure zones but rather done at only one zone per week.

The indoor residential sampling was always performed in the living room, and outdoor samples collected within 15 m of subject homes. Indoor samplers were placed at a height of 1.2–1.8 m above ground. Outdoor samplers, both at schools and residences, were placed at a height at least 1.8 meters above the ground (or on rooftops). In addition, hourly air quality data (including $PM_{2.5}$, O_3, NO_2, and CO) and meteorological parameters (including wind direction, wind speed, temperature, pressure, and humidity) were concurrently measured by CAM network stations in the three zones. These datasets were averaged over a seven-day period to correspond with the filter-based measurements. Wind rose data from the QMD Environmental District were used to assess the wind patterns. These were plotted using WRPLOT View™ software (Lakes Environmental Inc, Waterloo, Ontario, Canada).

As Figure 2 shows, Quito experienced low north-easterly winds with speeds ranging from 1 to 2.5 m/s during the study period. The majority of the winds at the Z1 CAM site were from the northeast. The Z2 CAM site experienced winds coming from the northeast and the southwest. The Z3 CAM site experienced winds from all directions with a few strong wind events (5.7–8.8 m/s) which emanated from the southeast. Wind speed, wind direction, and atmospheric temperature stratification are of particular importance in assessing air quality as they inhibit or promote pollutant transportation, mixing, and resuspension.

2.3 Field Samplers

Harvard 5 LPM cascade impactors (Demokritou et al., 2002) were used for PM sampling. The impactors operated with two impaction stages. These were used for the collection of $PM_{10-2.5}$ on polyurethane foam (PUF) and $PM_{2.5}$ on 37 mm diameter, 2 μm pore size PTFE (Polytetrafluoroethylene) filters (Pall Life Sciences, Ann Arbor, MI). MEDO Pumps (Model No.VP0125, Medo USA, Inc., Roselle, IL) were employed to generate a constant air stream of 5 *L*/min into the cascade samplers.

2.4 Gravimetric Analyses

Gravimetric analyses of PM samples were conducted at the University of Texas at El Paso (UTEP) Air Quality Laboratory. Filters and PUFs were conditioned, pre-weighed and stored in petri dishes for a period no greater than 30 days, prior to being placed into the PM samplers. The deployed filter media from each week's sampling period were collected, identified, and stored in Ziploc® bags and transported to the Biomedical Research Center laboratory at the Central University of Ecuador medical school for storage until transport to UTEP for post-weighing.

All PM samples were pre- and post- conditioned to room temperature (25 °C) and humidity (30%) for at least twenty-four hours before and after the deployment. Mass concentrations for $PM_{2.5}$ filters and $PM_{10-2.5}$ pufs were determined with a Mettler MX5 microbalance (Mettler-Toledo) having a precision of 1 μg. The accuracy of the microbalance was checked with a certified mass prior to each weighing session. Static effects were eliminated using a static neutralizing bar (MEB Shockless Static Neutralizing Bar, SIMCO, Hartfield, PA, USA).

For each weighing session, laboratory blank filters were also weighed. The average of three consecutive weight measurements was used as the final weight of each sampling media. If the consecutive measurements were not within 10 μg, then the media was re-weighed. The difference in mass was recorded for each sample utilizing the net weight of the media (before-and-after). Mass concentrations were reported as micrograms of PM per cubic meter of air (μg/m^3). The gravimetric analysis used in the current study has been previously described in detail elsewhere (Raysoni et al., 2011).

2.5 Quality Assurance and Quality Control (QA/QC) of Environmental Data

In order to minimize contamination of the PM samples during transportation and field work, strict quality assurance procedures were adopted (U.S. EPA, 2001). Field blanks and collocated samples were collected during the course of the study. A total of 36 field blanks for both the PM size fractions ($PM_{2.5}$ and $PM_{10-2.5}$) were collected. Collocated samples, one in each microenvironment per zone, were collected every month for a total of six paired duplicates per month. However, the field staff faced several logistical challenges during the course of the study. Five field blanks for $PM_{2.5}$ were eliminated due to filters getting wet during storms and hailstorms or possible tampering during inclement weather events. Prolonged blackouts lasting for a couple of days to weeks, children tampering with the instrumentation setup, and school personnel interrupting the measurement procedures led to the elimination of four sets of paired duplicate samples.

The limit of detection (LOD) was determined as three times the standard deviation of the field blanks. Precision was estimated as the root-mean squared difference between the collocated samplers divided by the square root of 2. Completeness was calculated as the number of samples collected divided by the target number of samples. The LOD was 5.2 μg/m^3 and 1.9 μg/m^3 for $PM_{2.5}$ and $PM_{10-2.5}$, respectively. The relative precision for $PM_{2.5}$ and $PM_{10-2.5}$ was 15.1% and 16.7%. For $PM_{2.5}$, 345 samples were collected out of the targeted 396 samples and the total number of collected samples for $PM_{10-2.5}$ was 354 out of 396 samples. The percentage of valid samples analyzed despite the aforementioned field sampling challenges was well above the acceptable value of 75% (Li et al. 2011).

2.6 Statistical Data Analysis

Descriptive statistics were analyzed using IBM SPSS (version 22) and Microsoft Excel 2007. Statistical significance was defined as $p < 0.05$. Box-plots were used to characterize PM species concentrations across the various school and residential sites (both indoors and outdoors), and CAM sites (ambient). Site-specific relationships between various pollutants and inter-pollutant correlations at each site were investigated using Spearman's Rho correlations. The spatial variability of $PM_{2.5}$ across schools and at two CAM sites (Z1 and Z2) was assessed using coefficients of divergence (COD). The COD provides a measure of uniformity between simultaneously sampled sites and is defined as

$$COD_{j,k}=\sqrt{\frac{1}{p}\sum_{i-1}^{p}\left[\frac{x_{i,j}-x_{i,k}}{x_{i,j}+x_{i,k}}\right]^2}$$

where $x_{i,j}$ is the i^{th} concentration measured at site j over the sampling period, j & k are two different sites, and p is the number of observations (Pinto et al., 2004). The COD provides a measure of uniformity between simultaneously sampled sites. A low COD value (≤ 0.2) indicates similar pollutant concentrations between two sites whereas a COD value approaching unity suggests significant difference in the absolute concentrations and subsequent spatial heterogeneity between the sites (Pinto et al., 2004; Li et al., 2011).

RESULTS AND DISCUSSION

3.1 Indoor and Outdoor Pollutant Concentrations

The descriptive statistics and the spatial contrast between indoor, outdoor, and ambient PM ($PM_{2.5}$, $PM_{10-2.5}$, and PM_{10}), concentrations at the schools, residences, and CAM sites are displayed, respectively, in Table 2 and Figure 3. Table 2 also shows the summary statistics for the indoor/outdoor (I/O) concentration ratios for all paired indoor-outdoor samples. Ambient $PM_{2.5}$ data were available only for CAM sites in zones Z1 and Z2. PM_{10} was monitored every 6^{th} day at the CAM sites. Therefore, seven day averages for comparison with school and residential data was not feasible.

3.1.1 $PM_{2.5}$ Concentrations—The mean school indoor $PM_{2.5}$ values, as expected, were 10.6 ± 4.9 μg/m^3, 14.7 ± 15.6 μg/m^3, and 10.8 ± 8.9 μg/m^3 at Z1 (north), Z2 (central), Z3 (southeast) zones, respectively. The school outdoor mean averages were also consistent across the three zones: Z2 (13.2 ± 3.5 μg/m^3), Z3 (13.0 ± 8.7 μg/m^3), and Z1 (10.9 ± 3.2 μg/m^3). The residential indoor concentrations were almost twice the magnitude of the indoor concentrations at the corresponding schools with the highest mean recorded at Z1 (29.0 ± 30.5 μg/m^3), followed by Z2 (20.8 ± 10.4 μg/m^3) and Z3 (19.3 ± 14.6 μg/m^3). The outdoor residential $PM_{2.5}$ values mirrored the same pattern as the outdoor school concentrations: Z1 (12.5 ± 14.6 μg/m^3), Z2 (14.3 ± 10.1 μg/m^3), Z3 (13.5 ± 7.2 μg/m^3). The variation between residential and school concentrations was smaller for outdoors than for the indoor microenvironments.

Residential indoor sources of $PM_{2.5}$ included candle burning, cooking, cleaning activities, and resuspension of dust from foot traffic. These indoor sources along with outdoor traffic emissions that infiltrated indoors resulted in concentrations that defied the pre-defined exposure patterns based on varying traffic densities. Neither cooking nor smoking activities occurred in the school classrooms. Slightly elevated $PM_{2.5}$ outdoor concentrations in the Z3 compared to Z1 may have been due to northerly winds transporting vehicular emissions from the urbanized zone.

During the study period, mean $PM_{2.5}$ values at the Z1 CAM site (11.8 ± 1.1 μg/m^3) were usually similar or lower than those measured at the respective school and residences. However, the Z2 CAM site recorded higher values (18.0 ± 3.5 μg/m^3) compared to those measured at the school and residence in the same zone. The Z1 school and CAM sites were located in close proximity of each other (i.e., one-tenth of a mile) and residences were located in all directions surrounding the monitor. In contrast, the Z2 CAM site was located in the north and the corresponding school and residences were respectively located to the south and east of the monitor. This CAM site was located downwind of the center of QMD and was impacted by heavy traffic emissions from the city center thereby resulting in elevated $PM_{2.5}$ concentrations.

3.1.2 $PM_{10–2.5}$ Concentrations—The indoor $PM_{10–2.5}$ concentrations observed were different from the expected at both the school and residence sites across the three zones. The Z1 school had the highest indoor mean value (16.1 ± 11.6 μg/m^3) followed by the Z3 (9.1± 6.0 μg/m^3), and Z2 schools (5.73 ± 2.8 μg/m^3). For indoor residential sites, both the Z1 site (16.4 ± 10.2 μg/m^3) and the Z3 sites (16.9 ± 21.1 μg/m^3) had similar mean levels compared to the Z2 site (12.4 ± 7.9 μg/m^3). The Z3 school recorded the highest outdoor mean (11.2 ± 5.4 μg/m^3), and the Z2 school the lowest (7.9 ± 2.2 μg/m^3). A similar pattern was observed at the outdoor residential sites, with Z3 and Z1 having the highest mean levels, 11.6 ± 6.6 μg/m^3 and 11.6 ± 3.1 μg/m^3, respectively, followed by Z2, 8.2 ± 2.4 μg/m^3. $PM_{10–2.5}$ concentrations at the residential sites and the schools were almost similar. Coarse particles were relatively uniform throughout the three zones and were due to widespread dust resuspension and soil erosion associated with agricultural activities and unpaved roads rather than combustion emission sources. This resulted in consistent outdoor $PM_{10–2.5}$ levels. In addition, the Z1 and Z3 sites were located on the outskirts of the QMD and were subjected to greater exposure from unpaved and semi-paved roads, quarry mining (especially close to Z1), a concrete plant (Z3), and agricultural activities (Z3).

3.1.3 PM_{10} Concentrations—Measured fine ($PM_{2.5}$) and coarse ($PM_{10–2.5}$) fractions of PM were combined to obtain the PM_{10} concentrations to reflect a combination of emission from combustion sources (e.g. traffic emissions) as well as geological material from dust resuspension. For indoor school PM_{10} concentrations, the highest recorded mean was at Z1 site (26.7 ± 15.9 μg/m^3) followed by almost similar means at Z2 (20.4 ± 16.0 μg/m^3) and Z3 (19.6 ± 13.6 μg/m^3) site. The recorded mean levels for indoor residences were: Z1 (45.3 ± 34.8 μg/m^3), Z2 (33.1 ± 14.5 μg/m^3), and Z3 (36.4 ± 34.0 μg/m^3). The mean outdoor school PM_{10} concentrations (in μg/m^3) were: Z1 (20.7 ± 3.7), Z2 (21.1 ± 4.6), and Z3 (24.2 ± 13.1). The mean outdoor residential PM_{10} concentrations (in μg/m^3) at the three zones were 24.5 ± 7.3 at Z1, 21.1 ± 7.3 at Z2, and 25.1 ± 13.2 at Z3. These values suggest that indoor PM source contributions may vary from site to site. However, outdoor sources and corresponding PM values are somehow ubiquitous in a much larger urban area. The Z1 zone was impacted not only by the ubiquitous traffic emissions resulting in elevated $PM_{2.5}$ levels but also by the significant geological dust emissions from the northern part where quarries are located. This resulted in high PM_{10} values observed both at the school and residences in this zone.

3.2 Indoor/Outdoor Ratios at Schools

Indoor/Outdoor (I/O) ratios for pollutants such as PM are crucial for understanding the real exposures of the schoolchildren in this study. The I/O ratios were computed for the school sites in the three neighborhood zones as both indoor and outdoor PM data were collected concurrently. It was not possible to do so at the residential sites because the collection of indoor and outdoor PM samples was not always paired.

The PM species collected at the school sites demonstrated a range of I/O ratios. These ratios are dependent on a variety of factors such as building and material characteristics and ventilation patterns, indoor sources of air pollution, occupancy rates, and building envelope tightness (Blondeau et al., 2005; Massey et al., 2012). The schools in this study were all naturally ventilated for thermal comfort which resulted in efficient penetration of outdoor particles through structure spaces and openings (Massey et al., 2012). Classroom doors and windows were opened, as needed, to permit the circulation of air. The I/O ratio boxplots for the three PM species are displayed in Figure 4.

As expected, the mean I/O ratios for $PM_{2.5}$ at the three schools were close to unity: Z1 (1.0 ± 0.59), Z2 (1.23 ± 1.15) and Z3 (1.06 ± 0.61). As per our knowledge, no major sources of indoor $PM_{2.5}$ were documented by the field staff during the study. Thus, most of the indoor $PM_{2.5}$ could be attributed to outdoor sources. These values are in line with prior findings from a U.S.-Mexico border community where median I/O ratios in two naturally ventilated Ciudad Juarez, Mexico, schools were 0.91 and 0.86 (Raysoni, et al. 2011). Other studies carried out in naturally ventilated schools have reported mean I/O ratios ranging from 0.69–0.88 in Thailand (Tippayawong et al., 2009) to 1–2 in Greece (Diapouli et al., 2008). The Greek study was conducted during winter months where doors and windows were probably kept closed to reduce cold drafts. Another study conducted at a naturally ventilated school from Chennai, India reported finding mean $PM_{2.5}$ I/O ratios as 1.44 ± 0.67 (Chithra and Nagendra, 2012). In the Quito schools, the relatively steady ambient day time temperatures permitted the classroom windows and doors, for ventilation practices, to be kept open throughout the year resulting in I/O ratios of ~ 1.

Correlations between indoor and outdoor $PM_{2.5}$ concentrations at schools in the Z2 (r = 0.57) and Z3 (r = 0.54) neighborhood zones were moderate but statistically significant (p = 0.03) suggesting common sources of $PM_{2.5}$. The findings from the present and another school study (Goyal and Khare, 2009) confirm that unless smoking is present, which is highly unlikely in a school setting, indoor sources are not significant contributors to $PM_{2.5}$. Thus, $PM_{2.5}$ indoor concentrations most likely reflect outdoor infiltration from traffic and other point and area sources.

The highest mean coarse PM I/O ratio for schools was observed at Z1 zone (1.71 ± 1.38), followed by Z2 (0.76 ± 0.46), and Z3 (0.87 ± 0.61). $PM_{10-2.5}$ usually have lower penetration efficiencies and are removed via gravitational settling (Hinds 1999). Room occupancy rates also have an influence on the re-suspension of previously deposited particles (Blondeau et al., 2005; Branis et al., 2009). It is possible that the conference room (Z2), computer classroom (Z3), and director's office (Z3) might have experienced less student traffic/ activity than the school in Z1, resulting in low I/O ratios. These findings are consistent with results from two Ciudad Juarez (Mexico) schools with median I/O ratios of 0.8 (classroom with infrequent cleaning and more students in relation to room dimensions) and 0.67 (library) resulting in particle resuspension (Raysoni, et al. 2011). In addition, Branis et al. (2009) reported that $PM_{10-2.5}$ was positively associated with the number of exercising pupils at a school gymnasium, suggesting human activity as the main source.

The mean PM_{10} I/O ratios reported for the school sites reflected a pattern similar to the $PM_{10-2.5}$ ratios. The mean PM_{10} I/O values were: 1.27 ± 0.85 (Z1), 1.03 ± 0.73 (Z2), and 0.96 ± 0.48 (Z3), suggesting that the location of the sampler may have affected the I/O ratios. Specifically, the Z1 school values appeared to reflect greater student activity throughout the day compared to those collected at the Z2 and Z3 school sites which experienced less student activity, and subsequently, resulted in less resuspension of particles. Similar to our findings, Diapouli et al. (2008) identified PM_{10} I/O ratios that ranged between 1and 2 in classrooms. They also reported PM_{10} I/O ratios of 2.5 for a school gymnasium, 1.1 for a computer classroom, and 0.53 for a library. In another study, monthly average PM_{10} I/O ratios, recorded at a school in New Delhi, India, ranged from 2 to 5 on weekdays and from 1 to 1.5 during weekends suggesting the important influence building envelope and foot traffic have on indoor PM_{10} measurements (Goyal and Khare, 2009). Similarly, high PM_{10} I/O ratios, 2.52 ± 2.71, were reported in naturally ventilated school in South India by Chithra and Nagendra (2012). The findings from the current study strongly suggest that indoor PM_{10} concentrations, in the absence of any indoor sources, are likely a combination of outdoor PM_{10} (infiltrating indoors) and occupancy rates (causing re-suspension).

3.3 Correlation Analysis between PM Species and Meteorological Parameters

Bivariate analyses were performed to investigate the temporal relationship between the indoor and outdoor PM metrics at schools and residences and the corresponding pollutant and meteorological parameters from CAM sites in each zone (Table 3).

3.3.1 School Indoors and CAM Sites—No statistically significant correlations were observed between school indoor $PM_{2.5}$ and pollutant parameters except SO_2 at the Z2 site (r = 0.66). However, Sarnat and associates (2000) have demonstrated that indoor microenvironments lack sulfur sources; therefore, sulfur concentrations were strongly associated with outdoor levels and sulfur compounds were used to estimate $PM_{2.5}$ of outdoor origin. Our study also showed that the majority of the indoor $PM_{2.5}$ in schools is likely attributable to outdoor infiltration due to the lack of indoor sources. For $PM_{10-2.5}$ and PM_{10}, significant correlations were observed at the Z1 school with temperature (r = 0.69 and r = 0.62) and solar radiation (r = 0.60 and r = 0.59). This result may be explained by the high traffic, outdoor activities, and/or fugitive dust emissions associated with the warm and dry weather conditions.

3.3 2 School Outdoors and CAM Sites—A strong, significant positive correlation was observed for outdoor $PM_{2.5}$ at the Z2 school and its corresponding CAMS site (r = 0.77), suggesting common traffic-related emissions. Significant positive correlations also were identified between outdoor $PM_{10-2.5}$ and both wind velocity (r = 0.65) and solar radiation (r = 0.65) while relative humidity (r = −0.82) and precipitation (r = −0.75) were negatively correlated in Z2. In Z3, outdoor $PM_{10-2.5}$ was positively correlated with wind velocity (r = 0.82) and solar radiation (r = 0.67) but negatively correlated with relative humidity (r = −0.76). Resuspension of $PM_{10-2.5}$ would be expected with increasing wind velocity whereas high humidity and precipitation lead to settlement. For PM_{10}, significant positive correlations were observed with wind velocity (r = 0.71) and solar radiation (r = 0.66) in Z3. In addition, a significant positive correlation was found between PM_{10} and CAMS $PM_{2.5}$ (r = 0.93) at Z2 which was an indication of consistent fraction of $PM_{2.5}$ in PM_{10}.

3.3.3 Residential Indoors and CAM Sites—Significant correlations were found between indoor $PM_{2.5}$ and temperature (r = −0.73), relative humidity (r = 0.59), and precipitation (r = 0.71) at Z2. These relationships were similar to those reported for a study conducted in Chennai, South India where PM concentrations were associated with mild winter temperatures (20° C), high humidity (80–90%), high pressure, and low wind speeds (Srimuruganandam and Nagendra, 2011). Significant negative correlations were observed between $PM_{10-2.5}$ and relative humidity (r = −0.68). This could be attributed to the efficient scavenging of the PM species by precipitation as the wet deposition provides the main PM sink (Jacob and Winner, 2009). Robust correlations between $PM_{2.5}$ and NO_2 (r = 0.73) and PM_{10} and NO_2 (r = 0.69) were observed in Z2 suggesting possible common sources of emissions.

3.3.4 Residential Outdoors and CAM Sites—No significant correlations were observed between the meteorological parameters and residential outdoor $PM_{2.5}$, except for relative humidity (r = 0.72) at Z1. A significant positive correlation was also observed at Z2 between $PM_{2.5}$ and NO_2 (r = 0.68). This might have been due to the close proximity of the residences and CAM sites and a common traffic source for the pollutants. Also, at Z2, $PM_{10-2.5}$ was positively associated with wind velocity (r = 0.62), temperature (r = 0.73), and solar radiation (r=0.66) and negatively correlated with relative humidity (r = −0.70). This could be the result of higher dust emissions linked to increasing wind speeds and higher dust precipitation scavenging with increased relative humidity.

Caution should be exercised while interpreting these results because the correlations were based on seven day averages. In addition, terrain elevation has an important effect on local temperature and pressure conditions. The CAM stations at Z1, Z2, and Z3 were located at respective elevations of 2,800, 2,840, and 2,453 m above sea level. Many of the homes sit on the bank (2,500 – 2,900 m) of the north-south trending Quito valley and may, therefore, be spared from being entrapped in the frequently occurring diurnal radiation inversion layer. Perhaps, some of the idiosyncratic observations in the correlation analysis could be attributed to this factor.

3.4 Spatial Contrast between Schools and Residences and Their Corresponding CAM sites

Spatial variation of $PM_{2.5}$ between the various sampled sites and corresponding CAM locations was elucidated by employing Coefficient of Divergence (COD) statistics. The COD values between the indoor and outdoor school and residences in Z1 and Z2 and the two respective CAM sites are presented in Table 4. The Z3 school and residences were not included in the analysis because the Z3 CAM site did not measure $PM_{2.5}$ concentrations. In addition, a PM_{10} COD analysis was not performed for the three zones because measurements at the respective CAM sites for this PM species were conducted only a few times per month. This resulted in a limited number of available data points. For any pollutant metric, COD analysis can only be conducted between simultaneously sampled sites (Pinto et al., 2004). At the Z1 and Z2 CAM sites, $PM_{2.5}$ was measured on an hourly basis. In order to conduct the COD analysis, this ambient data set from the two CAM sites was averaged over a seven day period to match with the sampled data from the schools and residences.

Spatial heterogeneity between the Z1 CAM and indoor microenvironment sites was confirmed at both the school (0.23) and residences (0.39). The COD values identified between the CAM site and the school (0.14) and residential outdoor microenvironment (0.16) accentuate a degree of spatial homogeneity, suggesting that ambient measurements from the CAM site in Z1 may be more representative of true exposure levels. At Z2, a low COD value (0.19) between the CAM site and the school outdoor microenvironment underscores similar $PM_{2.5}$ concentrations at these two sites. However, spatial non-uniformity is obvious between the Z2 CAM site and the indoor microenvironment at the same school (COD value = 0.30). In this zone, the COD values were 0.24 (between indoor residences and CAM site) and 0.23 (between outdoor residences and CAM site). These results suggest that $PM_{2.5}$ concentrations at residential and school indoor microenvironments in both Z1 and Z2 zones exhibited moderate spatial heterogeneity when paired with ambient data from corresponding CAM site. These results could be attributed to potential indoor sources of $PM_{2.5}$ and a high degree of infiltration from outdoor to indoors. COD values obtained in this study are comparable to those reported in literature. For example, Pinto and associates (2004) reported $PM_{2.5}$ COD values ranging from 0.06 to 0.24 for metropolitan areas in central and eastern United States and 0.07–0.48 for areas in western part of the country. $PM_{2.5}$ COD values ranging from 0.099 to 0.225 were reported in six southern California cities impacted by varying sources of anthropogenic emissions (Wongphatarakul et al.,

1998) while Kim and colleagues (2005) reported mean COD values of 0.22 for $PM_{2.5}$ concentrations at an urban site in St. Louis, Missouri.

3.5 CONCLUSIONS AND RECOMMENDATIONS

This study characterized different PM species, indoors and outdoors, over a 12 month period at schools and residences in three zones impacted by varying traffic densities in the Quito Metropolitan District, Ecuador. To the best of our knowledge, this is the first study to focus on PM_{10}, $PM_{10–2.5}$, and $PM_{2.5}$ measurements collected once a month as weekly averages across three low income neighborhoods impacted with varying levels of traffic densities. It also is the first to do so in a heavily populated high-altitude urban center.

Indoor and outdoor PM concentrations and the magnitude of differences across the three neighborhood zones were strongly dependent on the size fraction and microenvironment. For $PM_{2.5}$, higher exposures occurred in the Z2 neighborhood due to its location in the highly urbanized QMD center and high traffic emissions. The high $PM_{10–2.5}$ concentrations observed in the Z1 and Z3 neighborhoods appeared to be the result of an elevated resuspension of particles from unpaved surfaces, soil and geological erosion, quarries, and less due to traffic emissions. Within zones, concentration levels at the residential sites were consistently higher than that concurrently observed at schools. The difference in $PM_{2.5}$ between school and residential sites was greater for the indoor microenvironment than for $PM_{10–2.5}$. The I/O ratios for the PM species were close to unity at the schools and were attributable to minimal indoor sources, natural ventilation, and relatively uniform annual weather patterns. At the residential sites, higher I/O ratios indicated the presence of PM sources such as cooking, resuspension of particles due to foot traffic, cleaning, and other fugitive sources. Spatial heterogeneity for $PM_{2.5}$ was observed between the indoor microenvironment at schools and residences and the respective CAM sites in Z1 and Z2 zones. This suggests that measurements made at these central ambient monitoring locations were not representative of PM spatial variation in Quito, especially in indoor microenvironments. Furthermore, traffic-related concentration gradient was not identifiable due to the frequently occurring temperature inversions that entrap pollution in a narrow valley with rapid rising banks.

The findings from this unique study add to the body of air quality literature. They contribute to the limited data on indoor and outdoor air pollution in elementary schools and residences located in major urban centers especially ones located at high-altitude. The results provide important insights into the PM exposure of school-aged children not only at their residences but also their schools. Most previous studies have examined only the exposures of children at either the residence or school. This study addresses that limitation. The study findings also highlight the impact of outdoor air in naturally ventilated homes and schools in many developing countries where air conditioning and heating is minimal or nil. In addition, the study findings would aid environmental and public health program planners and policy makers in implementing more effective traffic management as well as behavioral changes (e.g., reducing natural ventilation practices in homes and schools during peak traffic hours) for a rapidly growing urban center such as Quito to reduce the air pollution exposures of children and other population groups.

Acknowledgments

This work was funded by a grant, NIEHS #1R21ESO16637-01A1, from the National Institute of Environmental Health Sciences (NIEHS) to Rodrigo Armijos, Principal Investigator. Its contents are solely the responsibility of the authors and do not necessarily represent the official views of the NIEHS or NIH (National Institutes of Health). The authors thank the students, parents, school principals, and teachers at the three schools for participating in this study. This work would not have been possible without the dedication and perseverance of the field staff involved with the deployment and sampling of instruments at the three schools and residences and technical assistance provided by the QMD Environmental Secretariat.

References

Adar SD, Sheppard L, Vedal SL, Polak JF, Sampson PD, Diez Roux AV, Budoff M, Jacobs DR Jr, Barr RG, Watson K, Kaufman JD. Fine particulate air pollution and the progression of carotid intima-media thickness: a prospective cohort study from the multi-ethnic study of atherosclerosis and air pollution. PLoS Medicine. 2013; 10(4):e1001430. [PubMed: 23637576]

Araujo JA. Particulate air pollution, systemic oxidative stress, inflammation, and atherosclerosis. Air Quality, Atmosphere and Health. 2011; 4(1):79–93.

Armijos RX, Weigel MM, Myers OB, Li WW, Racines M, Berwick M. Residential exposure to urban traffic is associated with increased carotid intima-media thickness in children. Journal of Environmental and Public Health. 2015; 2015:713540. [PubMed: 25685160]

Bateson TF, Schwartz J. Children's response to air pollutants. Journal of Toxicology and Environmental Health Part A: Current Issues. 2008; 71(3):238–243.

Bell ML, Davis DL, Gouveia N, Borja-Aburto VH, Cifuentes LA. The avoidable health effects of air pollution in three Latin American cities: Santiago, Sao Paulo, and Mexico City. Atmospheric Environment. 2006; 100:431–440.

Blondeau P, Iordache V, Poupard O, Gennin D, Allard F. Relationship between outdoor and indoor air quality in eight French schools. Indoor Air. 2005; 15:2–12. [PubMed: 15660564]

Bogo H, Otero P, Castro P, Ozafran MJ, Kreiner A, Calvo EJ, Negri RM. Study of atmospheric particulate matter in Buenos Aires city. Atmospheric Environment. 2003; 37(8):1135–1147.

Brachtl MV, Durant JL, Perez CP, Oviedo J, Sempertegui F, Naumova EN, Griffiths JK. Spatial and temporal variations and mobile source emissions of polycyclic aromatic hydrocarbons in Quito, Ecuador. Environmental Pollution. 2009; 157(2):528–536. [PubMed: 19004535]

Braniš M, Šafránek J, Hytychová A. Exposure of children to airborne particulate matter of different size fractions during indoor physical education at schools. Building and Environment. 2009; 44:1246–1252.

Calderón-Garcidueñas L, Macías-Parra M, Hoffmann HJ, Valencia-Salazar G, Henríquez-Roldán C, Osnaya N, Monte OC, Barragán-Mejía G, Villarreal-Calderon R, Romero L, Granada-Macías M, Torres-Jardón R, Medina-Cortina H, Maronpot RR. Immunotoxicity and environment: immunodysregulation and systemic inflammation in children. Toxicologic Pathology. 2009; 37(2): 161–169. [PubMed: 19171930]

Chithra VS, Shiva Nagendra SM. Indoor air quality investigations in a naturally ventilated school building located close to an urban roadway in Chennai, India. Building and Environment. 2012; 54:159–167.

Corporacion para el Mejoramiento del Aire de Quito (CORPAIRE). Inventario de Emisiones Atmosferico de Quito. 2007. Accessed at: http://190.152.144.74/paginas/articulos.html

Demokritou P, Gupta T, Ferguson S, Koutrakis P. Development and laboratory performance evaluation of a personal cascade impactor. Journal of the Air & Waste Management Association. 2002; 52(10):1230–1237. [PubMed: 12418733]

Diapouli E, Chaloulakou A, Mihalopoulos N, Spyrellis N. Indoor and outdoor PM mass and number concentrations at schools in the Athens area. Environmental Monitoring Assessment. 2008; 136:13–20. [PubMed: 17458512]

Ecogestion. Desarrollo de un Instrumento para la Gestion de la Calidad del Aire en Quito: Sintesis de Proyecto y Lecciones Aprendidas. 2005. Accessed at: www.iadb.org/sds/doc/LeccionesAprendidasQuito.pdf

Estrella B, Estrella R, Oviedo J, Narvaez X, Reyes MT, Gutierrez M, Naumova EN. Acute respiratory diseases and carboxyhemoglobin status in school children of Quito, Ecuador. Environmental Health Perspectives. 2005; 113(5):607–611. [PubMed: 15866771]

Gee IL, Sollars CJ. Ambient air levels of Volatile Organic Compound in Latin American and Asian Cities. Chemosphere. 1998; 36(11):2497–2506.

Gehring U, Gruzieva O, Agius RM, Beelen R, Custovic A, Cyrys J, Eeftens M, Flexeder C, Fuertes E, Heinrich J, Hoffmann B, de Jongste JC, Kerkhof M, Klümper C, Korek M, Mölter A, Schultz ES, Simpson A, Sugiri D, Svartengren M, von Berg A, Wijga AH, Pershagen G, Brunekreef B. Air pollution exposure and lung function in children: the ESCAPE project. Environmental Health Perspectives. 2013; 121(11–12):1357–64. DOI: 10.1289/ehp.1306770 [PubMed: 24076757]

Goyal R, Khare M. Indoor-outdoor concentrations of RSPM in classroom of a naturally ventilated school building near an urban traffic roadway. Atmospheric Environment. 2009; 43:6026–6038.

Han X, Naeher LP. A review of traffic-related air pollution exposure assessment studies in the developing world. Environment International. 2006; 32(1):106–120. [PubMed: 16005066]

Health Effects Institute (HEI). HEI Strategic Plan for Understanding the Health Effects of Air Pollution 2010–2015. Health Effects Institute; Boston, Mass, USA: 2010.

Hinds, WC. Aerosol Technology: Properties, behavior & measurement of airborne particles. New York: John Wiley & Sons, Inc; 1999.

Hoffmann B, Moebus S, Möhlenkamp S, Stang A, Lehmann N, Dragano N, Schmermund A, Memmesheimer M, Mann K, Erbel R, Jöckel KH. Heinz Nixdorf Recall Study Investigative Group. Residential exposure to traffic is associated with coronary atherosclerosis. Circulation. 2007; 116(5):489–496. [PubMed: 17638927]

Iannuzi A, Verga MC, Renis M. Air pollution and carotid arterial stiffness in children. Cardiology in the Young. 2010; 20(2):186–190. [PubMed: 20219153]

Jacob DJ, Winner DA. Effect of climate change on air quality. Atmospheric Environment. 2009; 43:51–63.

Jurado J, Southgate D. Dealing with air pollution in Latin America: The case of Quito, Ecuador. Environment and Development Economics. 1999; 4(3):375–388.

Kelishadi R, Hashemi M, Javanmard SH, Mansourian M, Afshani M, Poursafa P, Sadeghian B, Fakhri M. Effect of particulate air pollution and passive smoking on surrogate biomarkers on endothelial dysfunction in healthy children. Pediatrics and International Child Health. 2014; 34(3):165–169.

Kim E, Hopke PK, Pinto JP, Wilson WE. Spatial variability of fine particle mass, components, and source contributions during the regional air pollution study in St. Louis. Environmental Science & Technology. 2005; 39:4172–4179. [PubMed: 15984797]

Kunzli N, Jerrett M, Garcia-Esteban R, Basagana X, Beckermann B, Gilliland F, Medina M, Peters J, Hodies HN, Mack WJ. Ambient air pollution and the progression of atherosclerosis in adults. PLos One. 2010; 5(2):e9096. [PubMed: 20161713]

Laborde A, Tomasina F, Bianchi F, Bruné MN, Buka I, Comba P, Corra L, Cori L, Duffert CM, Harari R, Iavarone I, McDiarmid MA, Gray KA, Sly PD, Soares A, Suk WA, Landrigan PJ. Children's health in Latin America: the influence of environmental exposures. Environmental Health Perspectives. 2015; 123(3):201–209. DOI: 10.1289/ehp.1408292 [PubMed: 25499717]

Li, WW.; Sarnat, JA.; Raysoni, AU.; Sarnat, SE.; Stock, TH.; Holguin, F.; Greenwald, R.; Olvera, HA.; Johnson, BA. Characterization of traffic related air pollution in elementary schools and its impact on asthmatic children in El Paso, Texas. Mickey Leland National Urban Air Toxics Research Center; Houston, Texas: 2011. p. 246NUATRC Report Number 20

Massey DA, Kulshrestha A, Masih J, Taneja A. Seasonal trends of PM_{10}, $PM_{5.0}$, $PM_{2.5}$, & $PM_{1.0}$ in indoor and outdoor environments of residential homes located in North-Central India. Building and Environment. 2012; 47:223–231.

Pinto JP, Lefohn AS, Shadwick DS. Spatial variability of $PM_{2.5}$ in urban areas in the United States. Journal of the Air & Waste Management Association. 2004; 54:440–449. [PubMed: 15115373]

Poursafa P, Kelishadi R, Amini A, Amin MM, Lahijanzadeh M, Modaresi M. Association of air pollution and hematologic parameters in children and adolescents. Jornal de Pediatria. 2011; 87(4): 350–356. [PubMed: 21842113]

Raysoni AU, Sarnat JA, Sarnat SE, Garcia JH, Holguin F, Luevano SF, Li WW. Binational school-based monitoring of traffic-related air pollutants in El Paso, Texas (USA) and Ciudad Juarez, Chihuahua (Mexico). Environmental Pollution. 2011; 159:2476–2486. [PubMed: 21778001]

Raysoni AU, Stock TH, Sarnat JA, Sosa MT, Sarnat SE, Holguin F, Greenwald R, Johnson B, Li WW. Characterization of traffic-related air pollutant metrics at four schools in El Paso, Texas, USA: Implications for exposure assessment and siting schools in urban areas. Atmospheric Environment. 2013; 80:140–151.

Rice MB, Rifas-Shiman SL, Litonjua AA, Oken E, Gillman MW, Kloog I, Luttmann-Gibson H, Zanobetti A, Coull BA, Schwartz J, Koutrakis P, Mittleman MA, Gold DR. Lifetime exposure to ambient pollution and lung function in children. American Journal of Respiratory and Critical Care Medicine. 2015; doi: 10.1164/rccm.201506-1058OC

Romieu I, Gouveia N, Cifuentes LA, de Leon AP, Junger W, Vera J, Strappa V, Hurtado-Díaz M, Miranda-Soberanis V, Rojas-Bracho L, Carbajal-Arroyo L, Tzintzun-Cervantes G. HEI Health Review Committee. Multicity study of air pollution and mortality in Latin America (the ESCALA study). Res Rep Health Effects Institute. 2012; 10(171):5–86.

Sarnat JA, Koutrakis P, Suh HH. Assessing the relationship between personal particulate and gaseous exposures of senior citizens living in Baltimore, MD. Journal of the Air & Waste Management Association. 2000; 50:1184–1198. [PubMed: 10939211]

Secretaria de Ambiente. Informe de la Calidad del Aire de Quito. 2011. Accessed at: http://www.quitoambiente.gob.ec

Srimuruganandam B, Nagendra SMS. Characteristics of particulate matter and heterogeneous traffic in the urban area of India. Atmospheric Environment. 2011; 45(18):3091–3102.

Tippayawong N, Khuntong P, Nitatwichit C, Khunatorn Y, Tantakitti C. Indoor/outdoor relationships of size-resolved particle concentrations in naturally ventilated school environments. Building and Environment. 2009; 44(1):188–197.

United Nations (UN). World's population increasingly urban with more than half living in urban areas, 10 July 2014. New York: World Urbanization Prospects by UN DESA's Population Division; 2014. Accessed at: http://www.un.org/en/development/desa/news/population/world-urbanization-prospects-2014.html

United Nations Development Programme (UNDP). Human Development Report. 20th Anniversary Edition. The Real Wealth of Nations: Pathways to Human Development. 2010. Accessed at: http://hdr.undp.org/en/media/HDR_2010_EN_Complete_reprint.pdf

United Nations Environmental Programme (UNEP). Environment Climate Change Outlook: ECCO Metropolitan District of Quito, Regional Office for Latin America and the Caribbean. Division of Early Warning and Assessment; 2011.

Hassan R, Scholes R, Ash N. United Nations Environmental Programme (UNEP). Mountain Systems. Ecosystems and Human Well-being: Current State and Trends, Volume 1 Millennium Ecosystem Assessment. 2005; Chapter 24

United States Environmental Protection Agency (U.S.EPA). EPA Requirements for Quality Assurance Project Plans. 2001. EPA/240/B-01/003

Wang M, Gehring U, Hoek G, Keuken M, Jonkers S, Beelen R, Eeftens M, Postma DS, Brunekreef B. Air pollution and lung function in Dutch children: A comparison of exposure estimates and associations based on land use regression and dispersion exposure modeling approaches. Environmental Health Perspectives. 2015; 123(8):847–851. DOI: 10.1289/ehp.1408541 [PubMed: 25839747]

Wang G, Wang H, Yu Y, Gao S, Feng J, Gao S, Wang L. Chemical characterization of water-soluble components of PM_{10} and $PM_{2.5}$ atmospheric aerosols in five locations of Nanjing, China. Atmospheric Environment. 2003; 37:2893–2902.

Wongphatarakul V, Friedlander SK, Pinto JP. A comparative study of $PM_{2.5}$ ambient aerosol chemical databases. Environmental Science & Technology. 1998; 32(24):3926–3934.

World Health Organization (WHO). Public health, environmental and social determinants of health (PHE). Ambient (outdoor) air pollution in cities database. 2014. Available: http://www.who.int/phe/health_topics/outdoorair/databases/cities/en/

Wright RJ, Brunst KJ. Programming of respiratory health in childhood: influence of outdoor air pollution. Current Opinion in Pediatrics. 2013; 25(2):232–239. [PubMed: 23422354]

Wu S, Yang D, Wei H, Wang B, Huang J, Li H, Shima M, Deng F, Guo X. Association of chemical constituents and pollution sources of ambient fine particulate air pollution and biomarkers of oxidative stress associated with atherosclerosis: A panel study among young adults in Beijing, China. Chemosphere. 2015; 135:347–53. [PubMed: 25981523]

Author Manuscript
Author Manuscript
Author Manuscript

HIGHLIGHTS

- Z2(central) zone recorded elevated $PM_{2.5}$ levels compared to Z3(southeast) and Z1(north) zones.
- Major $PM_{10–2.5}$ sources are quarries, unpaved roads, agricultural activities, and soil erosion.
- The three study neighborhoods were impacted by varying traffic densities.
- Indoor-outdoor relationships for PM species were investigated at three schools.
- Central ambient monitoring sites may not be a good surrogate for understanding children's exposure in various exposure settings.

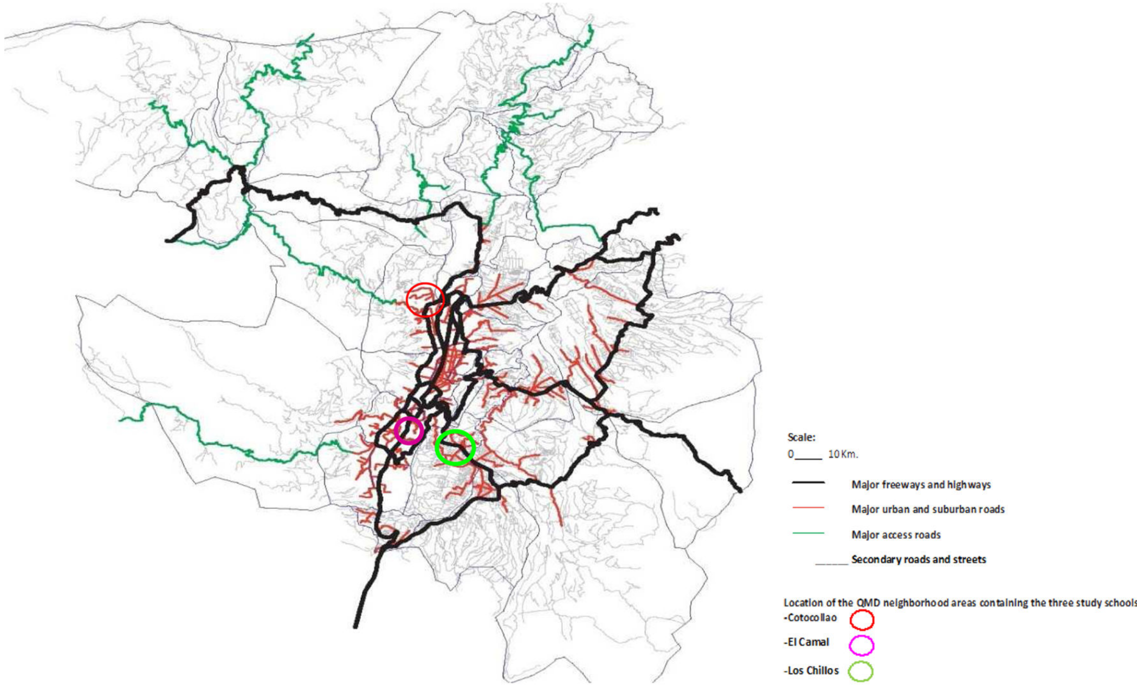
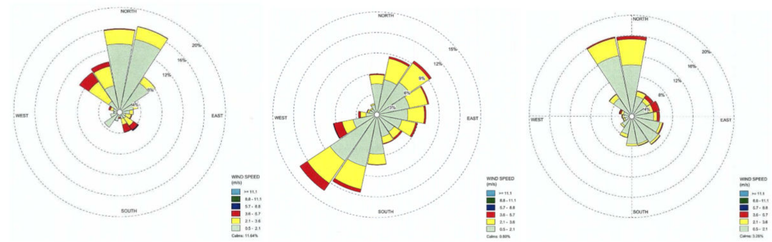


Figure 1.

Map of the study area including the three zones (Z1 = Cotocollao, Z2 = El Camal, and Z3 = Los Chillos) and major roadways in Quito, Metropolitan Area

Source: Armijos et al. (2015); used with permisssion

Author Manuscript
Author Manuscript
Author Manuscript



Cotocollao (Z1) El Camal (Z2) Los Chillos (Z3)

Figure 2.

Wind roses for the three CAMS sites for the study period (January 01 – December 31, 2010)

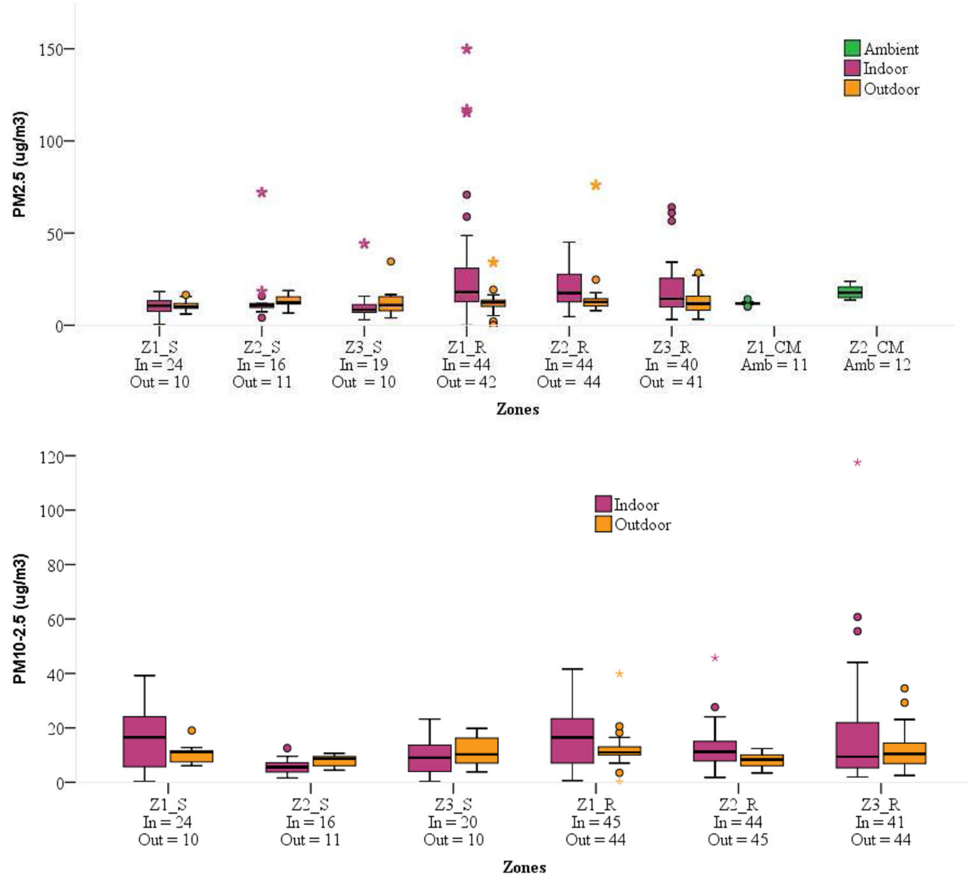


Figure 3a

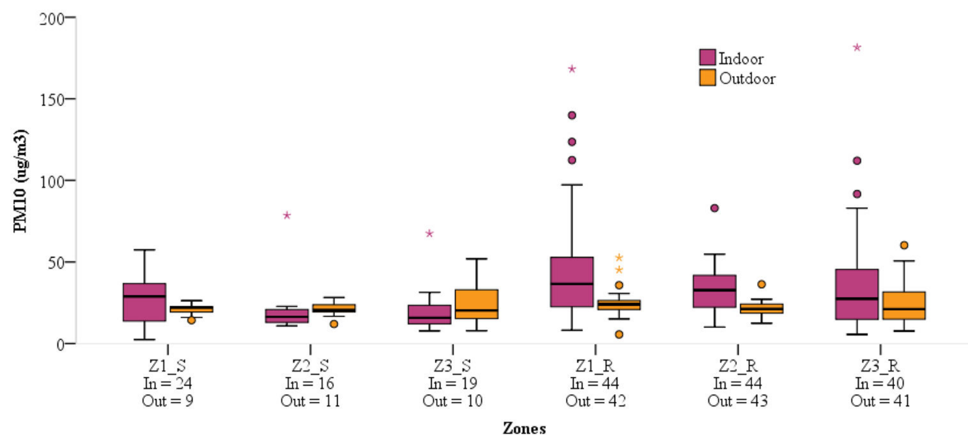


Figure 3b

Figure 3.

Boxplots for seven day indoor, outdoor and ambient PM species at schools, residences and CAMS sites (S = Schools, R = Residences, CM = CAM sites, In = Indoors, Out = Outdoors)

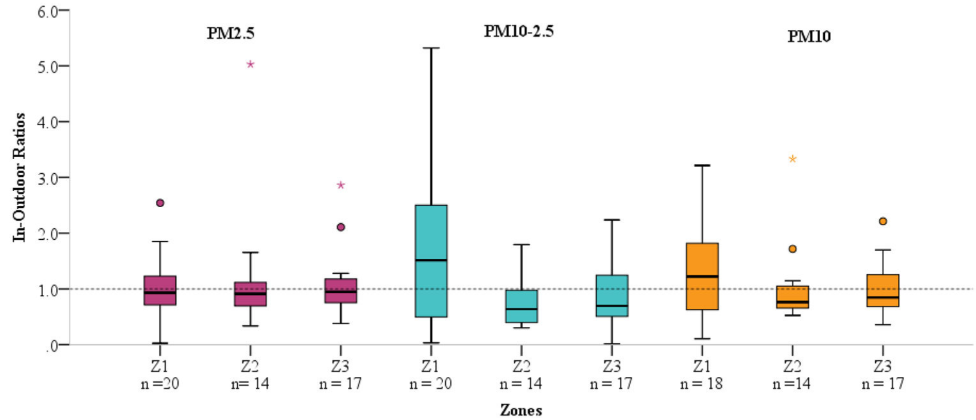


Figure 4.
In-Outdoor Ratio Boxplots for the PM Species at schools in the three zones (n = number of valid samples)

Table 1

Site Characteristics of the schools and residences in the three zones

Zones	Traffic Density	Site Characteristics	Historic PM Range (2004–2007 data)		School Sampling Sites			Number of Houses Sampled
			$PM_{2.5}$ (μg/m^3)	PM_{10} (μg/m^3)	Indoor1	Indoor 2	Outdoor	
Zone 1 (Cotocollao)	Medium	Northern part of the city Air quality influenced by emissions from quarries and vehicle exhausts (private and public transportation)	20.0	34.1	Classroom (1st floor)	Classroom (3rd floor)	School Terrace (2nd floor)	47
Zone 2 (El Camal)	High	Center of the city Area crisscrossed by several main avenues and bus routes Influenced by heavy traffic (both private and public)	26.7	77.0	Classroom (1st floor)	Students Conference Room (1st floor)	School Terrace (1st floor)	45
Zone 3 (Los Chillos)	Low	Southeast part of the city Low traffic density. Textile industry located east of the zone. Major emitter of NO_X, PM, SO_2, CO. School located downwind/upwind of Panamerican Highway (E35).	13.1	24.4	Computer room 2nd floor (Jan–Jun); Classroom 1st floor (Jul–Dec)	Director's Office 2nd floor	Outside classroom 2nd floor (Jan–Jun); Outside classroom 1st floor (Jul–Dec)	41

Table 2

Summary statistics of indoor, outdoor, and CAM PM species concentrations and corresponding indoor – and outdoor (I/O) ratios in the three zones.

PM Species	Zones	Indoor						Outdoor & Ambient Concentrations						I/0 Ratios							
		N	Mean	Median	SD	Min	Max	N	Mean	Median	SD	Min	Max	N	Mean	Median	SD	Min	Max	Spearman's R	p-value
$PM_{2.5}$	Schools																				
	Z1	24	10.6	10.6	4.9	0.4	18.3	10	10.9	10.1	3.2	6.1	16.5	20	1.0	0.93	0.59	0.02	2.54	−0.193	0.414
	Z2	16	14.7	11.0	15.6	4.1	72.1	11	13.2	12.2	3.5	6.6	18.9	14	1.23	0.91	1.15	0.34	5.03	**0.572**	**0.033**
	Z3	19	10.8	8.4	8.8	3.0	44.2	10	13.0	11.0	8.7	4.0	34.6	17	1.06	0.95	0.61	0.38	2.86	**0.537**	**0.026**
	Homes																				
	Z1	44	29.0	18.1	30.5	4.4	149.8	42	12.5	12.4	4.6	2.1	34.2								
	Z2	44	20.8	17.5	10.4	4.7	45.1	44	14.3	12.6	10.1	8.0	76.1								
	Z3	40	19.3	14.3	14.6	3.1	64.0	41	13.5	11.6	7.2	3.2	28.4								
	Central Monitoring Sites																				
	CM1							11	11.8	11.9	1.1	10.1	14.2								
	CM2							12	18.0	1.7	3.5	13.7	23.8								
$PM_{10-2.5}$	Schools																				
	Z1	24	16.1	16.5	11.6	0.4	39.2	10	10.5	11.0	3.8	6.1	19.0	20	1.71	1.51	1.38	0.03	5.32	0.109	0.648
	Z2	16	5.7	5.6	2.8	1.6	12.6	11	7.9	8.7	2.2	4.5	10.7	14	0.76	0.64	0.46	0.30	1.80	0.046	0.875
	Z3	20	9.1	9.1	6.0	0.1	23.2	10	11.2	10.2	5.4	3.8	19.8	17	0.87	0.69	0.61	0.02	2.24	0.461	0.063
	Homes																				
	Z1	45	16.4	16.5	10.2	0.6	41.6	44	11.6	11.0	3.1	0.1	20.6								
	Z2	44	12.4	11.2	7.9	1.8	45.7	45	8.2	8.3	2.4	3.4	12.4								
	Z3	41	16.9	9.4	21.1	2.0	117.5	44	11.6	10.5	6.6	2.5	34.5								
PM_{10}	Schools																				
	Z1	24	26.7	28.9	15.9	2.4	57.4	9	20.7	22.1	3.7	14.3	26.4	18	1.27	1.22	0.85	0.11	3.21	−0.216	0.390
	Z2	16	20.4	16.4	16.0	10.8	78.5	11	21.1	20.6	4.6	11.9	28.3	14	1.03	0.76	0.73	0.52	3.33	0.428	0.127
	Z3	19	19.6	15.7	13.6	7.7	67.5	10	24.2	20.3	13.1	7.8	51.9	17	0.96	0.84	0.48	0.36	2.21	**0.542**	**0.025**
	Homes																				
	Z1	44	45.3	36.5	34.8	8.1	168.3	42	24.5	24.1	7.3	5.6	52.7								
	Z2	44	33.1	32.7	14.5	10.0	83.0	43	21.1	21.2	4.5	12.4	36.3								

PM Species	Zones	Indoor						Outdoor & Ambient Concentrations						I/0 Ratios							
		N	Mean	Median	SD	Min	Max	N	Mean	Median	SD	Min	Max	N	Mean	Median	SD	Min	Max	Spearman's R	p-value
	Z3	40	36.4	27.5	34.0	5.6	181.5	41	25.1	21.1	13.2	7.6	60.3								

Table 3

Correlation coefficients for indoor and outdoor PM species (schools & residences) and corresponding pollutant and meteorological data from the CAM sites in each zone

Microenvironment	PM Species	Zones	Central Monitors									
			WV	T	RH	SR	P	$PM_{2.5}$	CO	NO2	SO_2	O_3
Schools												
Indoor	$PM_{2.5}$	Z1	−0.04	0.25	0.12	0.34	0.06	0.54	0.42	0.0	0.13	−0.31
		Z2	−0.21	−0.01	−0.12	0.11	−0.02	0.55	0.18	0.06	**0.66**	−0.13
		Z3	0.47	−0.02	−0.41	0.55	−0.37	-	-	-	-	0.45
	$PM_{10-2.5}$	Z1	0.15	**0.69**	−0.24	**0.60**	−0.50	0.36	0.42	−0.49	0.05	−0.33
		Z2	0.38	0.35	−0.18	0.45	−0.43	−0.46	−0.10	−0.60	−0.26	−0.05
		Z3	0.39	0.31	−0.30	0.12	0.24	-	-	-	-	0.29
	PM_{10}	Z1	0.12	**0.62**	−0.11	**0.59**	−0.37	0.46	0.46	−0.47	0.08	−0.30
		Z2	0.09	0.34	−0.34	0.25	−0.46	0.14	0.05	−0.40	0.50	−0.24
		Z3	0.58	0.15	−0.47	0.44	−0.16	-	-	-	-	0.40
Outdoor	$PM_{2.5}$	Z1	0.10	0.13	−0.20	0.41	−0.28	0.53	0.40	−0.22	−0.30	0.17
		Z2	0.08	−0.45	0.05	0.12	0.17	***0.77***	0.24	0.52	0.15	0.11
		Z3	0.62	0.28	−0.54	0.58	−0.22	-	-	-	-	0.46
	$PM_{10-2.5}$	Z1	0.26	0.08	−0.22	0.39	−0.43	0.25	−0.12	−0.22	−0.13	−0.03
		Z2	**0.65**	0.55	***−0.82***	**0.65**	***−0.75***	0.54	−0.46	−0.13	−0.01	0.60
		Z3	***0.82***	0.52	**−0.76**	**0.67**	−0.28	-	-	-	-	0.54
	PM_{10}	Z1	0.36	0.40	−0.55	0.60	−0.52	0.24	0.33	−0.36	0.55	−0.24
		Z2	0.30	0.05	−0.42	0.42	−0.23	***0.93***	0.02	0.25	0.21	0.35
		Z3	***0.71***	0.41	−0.61	**0.66**	−0.26	-	-	-	-	0.39
Residences												
Indoor	$PM_{2.5}$	Z1	−0.12	−0.07	0.47	−0.38	0.38	−0.15	−0.22	−0.16	0.44	−0.01
		Z2	−0.51	***−0.73***	**0.59**	−0.49	**0.71**	0.28	0.45	***0.73***	0.08	−0.19
		Z3	0.43	0.30	−0.60	0.31	−0.30	-	-	-	-	−0.07
	$PM_{10-2.5}$	Z1	−0.35	0.13	−0.19	−0.33	−0.19	−0.33	−0.17	−0.07	0.57	−0.35
		Z2	0.31	0.34	**−0.68**	0.50	−0.20	0.33	−0.29	0.04	−0.12	0.54
		Z3	0.30	0.38	−0.52	0.21	−0.15	-	-	-	-	−0.12

Microenvironment	PM Species	Zones	Central Monitors									
			WV	T	RH	SR	P	$PM_{2.5}$	CO	NO2	SO_2	O_3
	PM_{10}	Z1	−0.33	−0.15	0.33	−0.52	0.20	−0.21	−0.36	0.02	0.47	−0.11
		Z2	−0.40	−0.54	0.21	−0.36	0.53	0.46	0.34	**0.69**	0.18	−0.06
		Z3	0.35	0.41	−0.58	0.24	−0.17	-	-	-	-	−0.07
Outdoor	$PM_{2.5}$	Z1	−0.29	−0.28	***0.72***	0.01	0.39	0.10	−0.14	−0.11	−0.26	0.03
		Z2	0.08	−0.57	0.22	0.01	0.42	0.52	0.33	**0.68**	−0.12	0.24
		Z3	−0.05	−0.32	−0.24	−0.25	−0.03	-	-	-	-	0.69
	$PM_{10-2.5}$	Z1	0.48	0.29	−0.43	***0.76***	***−0.81***	0.32	0.27	−0.47	−0.40	0.05
		Z2	**0.62**	***0.73***	**−0.70**	**0.66**	−0.57	0.13	−0.05	−0.32	0.05	**0.58**
		Z3	0.58	0.32	***−0.82***	0.36	−0.12	-	-	-	-	0.62
	PM_{10}	Z1	0.22	−0.18	0.06	0.41	−0.25	0.27	−0.06	−0.14	−0.48	0.26
		Z2	0.57	0.41	**−0.63**	**0.62**	−0.44	0.42	−0.04	0.00	0.09	0.56
		Z3	0.34	−0.04	**−0.68**	0.10	−0.11	-	-	-	-	0.71

Significant Spearman's correlation coefficients at α = 0.05 level are shown in bold; Significant correlations at the α = 0.01 level are shown in bold and italicized.

WV: Wind velocity, T: Temperature, RH: Relative Humidity, SR: Solar Radiation, P = Precipitation

Table 4

Coefficient of Divergence (COD) values for $PM_{2.5}$ at schools and residences and corresponding CAM sites in Zone 1 and Zone 2

CAM Sites	Schools		Residences	
	Indoor	**Outdoor**	**Indoor**	**Outdoor**
Z1	**0.23**	0.14	**0.39**	0.16
Z2	**0.30**	0.19	**0.24**	**0.23**

Author Manuscript
Author Manuscript
Author Manuscript