Spatial and Temporal Variability of Outdoor Coarse Particulate Matter Mass Concentrations Measured with a New Coarse Particle Sampler During the Detroit Exposure and Aerosol Research Study

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Abstract

The Detroit Exposure and Aerosol Research Study (DEARS) provided data to compare outdoor residential coarse particulate matter ($PM_{10-2.5}$) concentrations in six different areas of Detroit with data from a central monitoring site. Daily and seasonal influences on the spatial distribution of $PM_{10-2.5}$ during Summer 2006 and Winter 2007 were investigated using data collected with the newly developed coarse particle exposure monitor (CPEM). These data allowed the representativeness of the community monitoring site to be assessed for the greater Detroit metro area. Multiple CPEMs collocated with a dichotomous sampler determined the precision and accuracy of the CPEM $PM_{10-2.5}$ and $PM_{2.5}$ data.

CPEM $PM_{2.5}$ concentrations agreed well with the dichotomous sampler data. The slope was 0.97 and the R^2 was 0.91. CPEM concentrations had an average 23% negative bias and R^2 of 0.81. The directional nature of the CPEM sampling efficiency due to bluff body effects probably caused the negative CPEM concentration bias.

$PM_{10-2.5}$ was observed to vary spatially and temporally across Detroit, reflecting the seasonal impact of local sources. Summer $PM_{10-2.5}$ was 5 $\mu g/m^3$ higher in the two industrial areas near downtown than the average concentrations in other areas of Detroit. An area impacted by vehicular traffic had concentrations 8 $\mu g/m^3$ higher than the average concentrations in other parts of Detroit in the winter due to the suspected suspension of road salt. $PM_{10-2.5}$ Pearson Correlation Coefficients between monitoring locations varied from 0.03 to 0.76. All summer $PM_{10-2.5}$ correlations were greater than 0.28 and statistically significant (p-value < 0.05). Winter $PM_{10-2.5}$ correlations greater than 0.33 were statistically significant (p-value < 0.05). The $PM_{10-2.5}$ correlations found to be insignificant were associated with the area impacted by mobile sources during the winter. The suspected suspension of road salt from the Southfield Freeway, combined with a very stable atmosphere, caused concentrations to be greater in this area compared to other areas of Detroit. These findings indicated that $PM_{10-2.5}$, although correlated in some instances, varies sufficiently across a complex urban airshed that that a central monitoring site may not adequately represent the population's exposure to $PM_{10-2.5}$.

Introduction

Epidemiological evidence indicates fine ($PM_{2.5}$) and coarse ($PM_{10-2.5}$) particulate matter have an acute effect on morbidity and mortality (Dockery et al., 1993; Brunekreef and Forsberg, 2005). These epidemiology studies frequently use data from one or a few central monitoring stations as a proxy for personal exposure. Ito et al. (2001) noted central monitoring sites are frequently sited for convenience, and not specifically to support epidemiology studies. Therefore, the particulate matter concentration data may not be representative of the exposure of the general population. Williams et al. (2008a) found personal $PM_{10-2.5}$ were not correlated with concentrations measured at a centrally located outdoor site. Wilson et al. (2005) concluded the degree of intra-urban spatial heterogeneity in particulate matter concentrations should be determined before using central monitoring site data as an estimate of personal exposures. This spatial analysis will minimize exposure misclassification and reduce uncertainty in relative risk estimates in longitudinal cohort studies.

Recent personal exposure studies have focused on the effects of $PM_{10-2.5}$ because of persistent concerns about the toxicity of this size fraction combined with a paucity of exposure data compared with the $PM_{2.5}$ fraction. Yeatts et al. (2007) showed that $PM_{10-2.5}$ increments of as little as 1 $\mu g/m^3$ can alter heart rate variability in adults with asthma. This finding suggests that $PM_{10-2.5}$ spatial and temporal variability need to be fully understood if concentration data from a central monitoring site are to be used to represent the exposure of the general population.

Source apportionment methodologies also are being used to link ambient $PM_{2.5}$ concentrations to personal exposure to specific sources (Hopke et al., 2001) and specific species (Landis et al., 2001;Zhao et al., 2006). Again, these techniques rely upon data from one or two central monitoring sites to provide the requisite speciation data. Extension of the methods to $PM_{10-2.5}$ exposure also will require a better understanding of the concentration spatial and temporal variability.

Spatial and temporal variability of particulate matter concentrations have been previously investigated to understand the influence of meteorology and geographical location of the sample collection instruments with respect to local sources. These studies focused on $PM_{2.5}$ and PM_{10}, where $PM_{10-2.5}$ were calculated by difference. Burton et al. (1996) found population density to be a surrogate for local sources. They also found that $PM_{10-2.5}$ spatial variability was determined by wind direction, but not

wind speed. Guerra et al. (2006) found a similar influence of wind direction on PM_{10} and $PM_{2.5}$ concentrations in southeast Kansas. Suh et al. (1997) found one site in Washington D.C. with elevated $PM_{10-2.5}$ compared to the other sites. However, Wilson and Suh (1997) concluded that their analysis was limited by the low measurement precision resulting from calculating low coarse particle concentrations by the difference between PM_{10} and $PM_{2.5}$ concentrations. Chen et al. (2007) used low volume PM_{10} and $PM_{2.5}$ samplers to assess $PM_{10-2.5}$ spatial variability and concluded that local sources affected the strength of the correlation with a centrally located monitor.

This study used a new coarse particulate matter sampler (CPEM) to provide direct measurement of $PM_{10-2.5}$ and $PM_{2.5}$ concentrations. The CPEM sample flow and size were designed for personal exposure assessment, but the instrument was also designed to be sufficiently rugged for monitoring at outdoor and indoor locations. Data collected with the CPEM characterized the $PM_{10-2.5}$ spatial and temporal variability across the Detroit metropolitan airshed during the Detroit Exposure and Aerosol Research Study (DEARS). Specifically, this portion of the study compared outdoor residential concentrations with those from a community monitoring site. Daily and seasonal influences on the spatial distribution of the $PM_{10-2.5}$ were examined. Also, the $PM_{10-2.5}$ and $PM_{2.5}$ concentration data collected at the community monitoring site allowed the CPEM performance to be evaluated against a referee dichotomous sampler.

Methods

CPEM

The CPEM simultaneously provides $PM_{10-2.5}$ and $PM_{2.5}$ concentrations from one compact instrument. The CPEM is a series of three separation stages designed to be inserted into the MSP Model 200 PM_{10} PEM (MSP Corp, Shoreview, MN) (Figure 2). The CPEM operates on a battery powered pump to collect $PM_{10-2.5}$ by impaction on two, sequential 25 mm Teflo filters (Pall Corp., East Hills, NY). The final stage collects the $PM_{2.5}$ on a 37 mm Teflo filter (Pall Corp., East Hills, NY). Teflo filters allow speciation analyses to be conducted on both size fractions. The CPEM's small dimensions (4.7 cm high, 4.2 cm wide), light weight (110 g), low flow (2 Lpm), yet rugged construction allow the system to be deployed as a personal exposure monitor as well as a stationary indoor or outdoor monitor. Flow

measurements measured with a Drycal DC-Lite (BIOS International, Butler NJ) at CPEM deployment and retrieval were averaged to calculate the sample volume. If necessary, pump flow was adjusted at deployment to achieve 2 Lpm

DEARS Study

A primary goal of the DEARS was to evaluate the uncertainty associated with using community-based monitoring as a surrogate for human exposures to pollution in a metropolitan area impacted by multiple types of sources. A second DEARS research objective evaluated new exposure measurement technologies. Williams et al. (2008b) and Williams (2005) describe the DEARS objectives and study design in detail. CPEMs were deployed during the final two (of six) sampling seasons to assess the sampler performance capabilities to measure $PM_{10-2.5}$ and $PM_{2.5}$ concentrations and evaluate spatial and temporal variability. Summer season samples were collected between July 11 and August 26, 2006, while winter samples were collected from January 23 to February 24, 2007.

Sample collection focused on six Enumeration Measurement Areas (EMAs) around Detroit (Figure 1) selected to highlight specific air pollution source categories in each area. The distance between EMAs varied from 2 to 48 km. EMAs 1 and 5, located immediately southwest of downtown Detroit, examined the impact of industrial sources. Residences in EMA 6 were within 300 m of Highway M39-Southfield Freeway, a major north-south route through the residential neighborhoods in the western portion of Detroit. EMA 4 was a residential neighborhood on the northeast side of Detroit and downwind of the industrial and mobile sources. EMA 7, located west of Detroit in Belleville, had no identified source categories and represented the regional background aerosol. EMA 0, the central monitoring site, was located at the Michigan Department of Environmental Quality (MDEQ) site in Allen Park, MI. MDEQ and U.S. EPA provided meteorological data. MDEQ provided hourly wind speed and direction data at EMA 0. U.S. EPA provided daily average Monin-Obukhov lengths as a measure of atmospheric stability.

Experimental Design

Simultaneous measurements with the CPEM and an Andersen Model SA-244 Dichotomous sampler (Andersen, Smyrna, GA) collocated at EMA 0 assessed the CPEM comparability against an EPA referee sampler. Accuracy (n = 72) was calculated as the difference between the CPEM and referee concentrations divided by the referee concentration, where a value of 0 indicated no difference.

Collocated duplicate CPEM samples (n = 12) deployed at EMA 0 assessed precision, quantified as the coefficient of variation.

CPEMs were deployed daily on a Tuesday through Sunday schedule in each EMA to investigate $PM_{10-2.5}$ spatial and temporal variability. Summer had 35 days of sample collection in each EMA. Winter had 25 days of samples, except in EMA 6 which had 20 due to a lack of participants. The sample collection interval was 24 hours, nominally between 9:00 AM to 9:00 AM. CPEMs located outdoors were mounted on a bluff body about 2 m above the ground (Rodes and Thornburg, 2006). CPEMs deployed in EMA 0 faced northeast due to space limitations on the southwest side of the bluff body. CPEMs deployed in other EMAs were deployed in residential backyards and deliberately oriented to face away from structures or other obstacles.

EMAs 1, 4, 5, and 6 had multiple days each season when two or three outdoor residential locations were sampled simultaneously within the EMA (Table 1). The number of intra-EMA duplicate comparisons varied from a minimum of 9 (summer, EMA 1) to a maximum of 20 (summer, EMA 4). This replication allowed spatial variability on scales less than 1 km to be studied. The residences sampled during these days were labeled as A, B, or C. Residence A was the closest to the Southfield Freeway (EMA 6) or closest to the geographical center of all other EMAs. Residence B was the next closest and Residence C was the furthest away. The average distance between residences was 0.8 ± 0.2 km.

Analysis

Gravimetric analysis of $PM_{10-2.5}$ and $PM_{2.5}$ filters was conducted according to procedures described in Lawless and Rodes (1999). Field and laboratory blanks provided a blank correction factor applied to all mass concentrations.

SAS version 9 (SAS Inc., Cary, NC) was used to perform all statistical analyses using an alpha of 0.05. Concentrations were log-transformed for statistical analyses comparing EMAs to satisfy normal distribution assumptions. The Tukey-Kramer least squares analysis procedure was used to determine if concentration differences between EMAs, within EMAs, and seasonal variations were statistically significant. The Coefficient of Divergence (COD) between EMAs assessed $PM_{10-2.5}$ spatial homogeneity (Pinto et al., 2004). A COD of 0 indicates complete homogeneity and a value of 1 indicates maximum differences. The Pearson Correlation Coefficients (r) assessed the temporal homogeneity in $PM_{10-2.5}$

between EMAs. Spatial-temporal modeling was conducted with a non-linear, autoregressive analysis procedure to assess the concentration differences within and between EMAs. The autoregressive structure controlled for the daily variability in meteorology conditions such that comparisons between the independent variables could be made. The model included distance between EMAs, weekday/weekend, wind direction, wind speed, and atmospheric stability.

Results

The 2 Lpm CPEM performed well compared to the 16.7 Lpm dichotomous sampler for 24 hour integrated samples (Figure 2). The R^2 from the linear regression was 0.91 for $PM_{2.5}$ and 0.81 for $PM_{10-2.5}$. For $PM_{2.5}$, the statistically significant linear regression slope was 0.97 (p-value < 0.0001) and not different from unity (95% C.I.: 0.90 to 1.05). The 1 $\mu g/m^3$ intercept was statistically insignificant (p-value = 0.14). The $PM_{10-2.5}$ linear regression slope of 0.77 was also statistically significant (p-value < 0.0001) but was different from unity (95% C.I.: 0.68 to 0.86). Again, the 1 $\mu g/m^3$ intercept was statistically insignificant (p-value = 0.13). Accuracy (Figure 3) and precision (Figure 4) of the CPEM $PM_{10-2.5}$ and $PM_{2.5}$ data typically became poorer as the concentration decreased below 10 $\mu g/m^3$. However, there were exceptions to this trend.

Figure 5 shows the $PM_{10-2.5}$ geometric mean and standard deviation for each EMA as a function of season. $PM_{10-2.5}$ spatial variability between EMAs was evident. EMAs 1 and 5, with average summer concentrations greater than 12 $\mu g/m^3$, had significantly greater $PM_{10-2.5}$ than the other EMAs (p-value < 0.01). Summer $PM_{10-2.5}$ in EMAs 0, 4, 6, and 7 were statistically similar, with average concentrations between 6.3 and 7.5 $\mu g/m^3$. Winter 2007 $PM_{10-2.5}$ spatial variability between EMAs showed a different trend. EMA 7 concentrations were lower (p-value < 0.05) than concentrations in EMAs 0, 1, 5, and 6. EMA 4 concentrations were lower (p-value < 0.05) than concentrations in EMAs 0 and 6. Winter $PM_{10-2.5}$ in EMAs 0, 1, 5, and 6 were similar and varied from 9.4 to 11.0 $\mu g/m^3$.

Figure 5 also shows a seasonal influence on the $PM_{10-2.5}$ spatial distribution. EMAs 1, 5, and 7 had significantly greater $PM_{10-2.5}$ in summer than in winter (p-value < 0.05). Winter $PM_{10-2.5}$ was greater than summer $PM_{10-2.5}$ in EMAs 0 and 6 (p-value < 0.05). Seasonal $PM_{10-2.5}$ variations in EMA 4 were small and statistically insignificant.

Table 2 presents r and COD values between EMAs for each season. All summer r values were statistically significant (p-value < 0.02) and ranged from 0.28 to 0.63. Winter r values ranged from 0.03 to 0.76. Winter EMA 6 $PM_{10-2.5}$ was not correlated temporally (p-values > 0.4) with the $PM_{10-2.5}$ other EMAs. COD values also showed a seasonal pattern. Summer CODs ranged from 0.17 to 0.41. Winter CODs spanned 0.26 to 0.50. For each season, the paired r and COD values were logically divided or "clustered" into 3 distinct groups. In summer, the first cluster was the singular comparison of EMAs 1 and 5, which had the lowest COD. CODs between EMAs 0, 4, 6, and 7, varying from 0.21 to 0.26, comprised the second summer cluster. The third summer cluster, with CODs generally greater than 0.33, compared EMAs 1 and 5 to EMAs 0, 4, 6, and 7. The first winter group contained EMA 6 comparisons to the other EMAs because of the statistically insignificant r values. The second winter group, with COD values greater than 0.3 and statistically significant r values compared EMAs 0, 1, 5. EMA 4 and 7 comparisons with the other EMAs formed the third winter cluster.

The DEARS sample scheme planned for duplicate participants within certain EMAs to be sampled weekly. This scheme allowed a comparison of the intra-EMA variability in the particulate matter concentrations accounting for daily variability in the concentrations (Table 1). $PM_{10-2.5}$ daily variability was evident for both summer and winter, as shown by the statistically significant p-values for the "Day" variable. The statistically insignificant p-values for the "Location" variable indicate $PM_{10-2.5}$ was spatially similar within most EMAs. Only winter $PM_{10-2.5}$ within EMA 6 varied spatially (p-value < 0.05). The trend indicated $PM_{10-2.5}$ decreased with increasing distance from the Southfield Freeway.

Spatial-temporal modeling determined atmospheric stability was the only significant variable (p-value < 0.01) affecting summer and winter $PM_{10-2.5}$ concentrations. In summer, greater atmospheric instability, as indicated by small Monin-Obukhov lengths, promoted $PM_{10-2.5}$ concentration homogeneity between EMAs. Conversely, atmospheric stability during the winter produced PM10-2.5 concentrations differences between EMA 6 and the other EMAs. Distance between EMAs, weekday/weekend, wind speed, and wind direction did not influence summer or winter $PM_{10-2.5}$ concentrations across Detroit .

Discussion

CPEM Performance

The easy maintenance, assembly, and deployment of the CPEM made the sampler ideal for field use. All samples were processed in a field office without the need for impactor greasing or special filter handling. Trained technicians unloaded returned CPEMs and prepared new CPEMs at a rate of 12 per hour. Use of Teflo filters for sample collection will allow chemical speciation analyses to be conducted in the future. CPEM sample completeness exceeded the DEARS minimum data quality objective of 90%. Summer 2006 and winter 2007 had 97% and 96% valid samples. DEARS data quality objectives for CPEM $PM_{2.5}$ and $PM_{10-2.5}$ performance were ± 20% accuracy and ± 20% precision (Williams et al., 2000). CPEM $PM_{2.5}$ data achieved the accuracy and quality objectives. $PM_{10-2.5}$ performance achieved the precision data quality objective of ± 20% precision, but the cumulative accuracy was 24%, slightly higher than the target.

CPEM $PM_{10-2.5}$ measurement performance is very similar to that of other low flow particulate matter samplers used for saturation studies. Chen et al. (2007) used PM_{10} and $PM_{2.5}$ MiniVol samplers (Airmetrics, Eugene, OR) to calculate $PM_{10-2.5}$ by difference. The performance of the MiniVol was nearly identical to the CPEM performance during DEARS. Their study showed the MiniVol had a negative bias of 20% compared to the reference method and an R^2 value of 0.78. Chen et al. attributed the differences to differences in face-velocity through the filter, aspiration efficiencies into the inlet, or particle bounce in the dichotomous sampler. Ott et al. (2008) recently reported $PM_{10-2.5}$ measured with the Wagner and Leith passive sampler (Wagner and Leith, 2001) during a field study in Iowa. Their results showed excellent precision, yet a 29% positive bias compared to the dichotomous sampler was observed. This systematic bias was attributed to assumptions made in calculating mass concentration from the SEM images, and not with the coarse particle collection efficiency. The Personal Respirable Particulate Sampler (PRPS) is the most similar to the CPEM in operation in that $PM_{10-2.5}$ is measured directly, although the unit sampled at 5 Lpm and collected $PM_{10-2.5}$ on a PUF substrate (Demokritou et al., 2003). Case et al. (2008) collocated a modified version of the PRPS with an Andersen dichotomous sampler. Their analysis showed a 10% positive bias, a statistically significant R^2 of 0.87, and coefficient of variation generally less than 2.

The negative bias in the CPEM $PM_{10-2.5}$ may have been caused by the orientation of the sampler when mounted on the bluff body. The CPEM uses a MSP Model 200 inlet cap that is inherently

directional when used outdoors, as opposed to the omni-directional dichotomous sampler inlet. In addition, the bluff body system is designed to simulate personal inhalation exposures to particulate matter, where the presence of the human body is known to alter particle flow trajectories (Kenny et al., 1997). In studies such as the DEARS where comparisons are made between personal and outdoor sampling technologies, the use of a bluff body to simulate the human form should provide similar flow profiles to bias both personal and outdoor measurements in the same manner. As a result, the bluff body is not an omni-directional platform. The bluff body mounted CPEM faced northeast at EMA 0, when the prevailing wind direction during both seasons covered an arc from the south to northwest. That meant the bluff body was a barrier that could have prevented complete collection of the coarse particles under the prevailing wind direction. With the CPEM on the downwind side, the airflow around the bluff body would have formed eddy vortices that could have carried the particulate matter away from the bluff body and decreased the coarse particle concentration in proximity to the CPEM inlet. Both wind speed and direction would then be expected to influence the effect of the eddy vortices on the sampled concentration. In wind tunnel experiments, Kenny et al. (1997) demonstrated that personal aerosol samplers located on the back, downstream side of a manikin sampled 80% to 90% of 6 μm diameter particles. This study was not designed to characterize this effect, and insufficient data were available to develop a statistically valid concentration correction factor accounting for wind speed and wind direction. However, the analysis did suggest the magnitude of the bluff body effect decreased the measured CPEM $PM_{10-2.5}$ by approximately 2 to 5 $\mu g/m^3$, equivalent to the -20% bias seen in the data.

Instances of poor CPEM accuracy and precision at concentrations less than 10 $\mu g/m^3$ can be explained by a propagation of error analysis. Potential sources of error in the calculation of the particulate matter concentration were the gravimetric mass, sample flow, and sample time measurements. The error associated with digital flowmeters and a synchronized clock was minimal, especially when the variability in the pump flow and sample collection period was negligible. However, the error associated with gravimetric measurement of the mass collected on the filter can be significant. Across both seasons, the error in the $PM_{10-2.5}$ gravimetric analysis, calculated from a combination of field and laboratory blanks, was 2.8 μg. When converted to a concentration, the error in the CPEM and dichotomous sampler measurements was 1 $\mu g/m^3$ and 0.17 $\mu g/m^3$, respectively. Considering that almost

70% of the $PM_{10-2.5}$ measured during DEARS were less than 10 $\mu g/m^3$, the impact of the low sample volume on the accuracy and precision of the CPEM concentration measurements was magnified compared to the dichotomous sampler measurements.

Based on the preceding analysis, we concluded the CPEM data were suitable for investigating $PM_{10-2.5}$ spatial and temporal variability across Detroit.

Comparison with FEM Requirements

The CPEM comparability data were compared against the EPA federal acceptance criteria for federal equivalent methods (FEM). The federal acceptance criteria for $PM_{10-2.5}$ Class II instruments specify a regression slope of 1 ± 0.1, intercept of -3.8 to 3.8 $\mu g/m^3$, R^2 greater than 0.9, and precision of 15% (U.S. EPA, 2007). Class II $PM_{2.5}$ FEM acceptance criteria are the same except the intercept range is -1.5 to 1.5 $\mu g/m^3$. CPEM performance for $PM_{2.5}$ met all the EPA acceptance criteria. However, the CPEM $PM_{10-2.5}$ performance only satisfied the criteria for the y-intercept. The linear regression slope showed the CPEM underestimated $PM_{10-2.5}$ by an average of 23%, 11% more than the minimum acceptable level. The R^2 of 0.81 was slightly less than the target value. The aggregate CPEM $PM_{10-2.5}$ precision, calculated as the root mean square of all comparisons, was 18%. Although all FEM criteria were not achieved, Chow and Watson (2008) recently noted that a portable, battery powered sampler like the CPEM can sacrifice FEM requirements to achieve study objectives. Research studies like the DEARS require inexpensive, portable, easy to use instrumentation, like the CPEM, to provide representative spatial and temporal $PM_{10-2.5}$ mass and chemical composition distributions at a reasonable cost.

Summer $PM_{10-2.5}$ Concentration Gradients

$PM_{10-2.5}$ during DEARS showed spatial differences between EMAs during the summer. $PM_{10-2.5}$ in EMAs 1 and 5 was more than 5 $\mu g/m^3$ higher than the other EMAs. Corresponding CODs between EMAs 1 or 5 with the other EMAs were greater than 0.33, also indicating spatial variability. The summer spatial $PM_{10-2.5}$ gradients possibly were due to localized industrial sources in those areas and higher population density (Phillips et al., 2008). Burton et al. (1996) reported statistically significant differences of up to 3 $\mu g/m^3$ across different portions of Philadelphia due to the presence of local sources in more densely

populated areas. Higher $PM_{10-2.5}$ is expected in industrial, urban areas, although periods do occur when $PM_{10-2.5}$ in rural areas is greater than urban areas (Querol et al., 2008).

Even though spatial concentration gradients existed, summer $PM_{10-2.5}$ was temporally correlated across all EMAs. The highly unstable atmosphere induced by daytime heating promoted mixing and dispersion of $PM_{10-2.5}$ during the summer to yield statistically significant r values, ranging from 0.28 to 0.63. As a result of the atmospheric instability, a large portion of the $PM_{10-2.5}$ in the summer may have originated from a common source that impacted all of Detroit.

Although a detailed $PM_{10-2.5}$ size distribution characterization was not part of the DEARS study design, we hypothesize that these findings suggest the $PM_{10-2.5}$ in Detroit during the summer is comprised of two components. One component, most likely consists of particles smaller than 5 μm that can be transported over substantial distances, especially in unstable atmospheres, to create an urban background concentration. $PM_{10-2.5}$ in EMAs influenced the most by the urban background, and probably minimally impacted by local sources, consistently exhibited the lowest concentrations, statistically significant correlations, low CODs, and large distances between them. EMAs 0, 4, 6, 7 during the summer fit this category. Alternatively, EMAs with high $PM_{10-2.5}$, high correlations and low CODs signified areas impacted by the same, local source(s). Therefore, the other $PM_{10-2.5}$ component consists of larger particles from local sources to generate areas of elevated $PM_{10-2.5}$. Reid et al. (2003) suggested these larger particles are between 5 to 12 μm. Summer $PM_{10-2.5}$ in EMAs 1 and 5 fit this profile.

Winter $PM_{10-2.5}$ Concentration Gradients

Spatial differences also existed in the winter. $PM_{10-2.5}$ in EMAs 7 and 4 were as much as 8 $\mu g/m^3$ lower than EMAs 0 and 6. CODs for these four comparisons were also the highest (>0.43). Furthermore, the temporal variability indicated by the statistically insignificant r values corresponding to EMA 6 suggested the presence of a unique source limited to that EMA. The spatial and temporal variability possibly was caused by the presence of resuspended road salt from the Southfield Freeway (EMA 6) and Interstate 75 (near EMA 0). These two freeways average more than 100,000 vehicles per day, and frequent snow events necessitated recurrent application of salt to the roads. Whereas other EMAs were residential areas with few major roads an insignificant daily traffic counts to suspend the road salt that

was applied. In addition, the highly stable atmosphere inhibited dispersion and mixing of $PM_{10-2.5}$ resuspended from the roads.

The proximity of the EMA 6 sampling locations to the highway and the high concentration of resuspended road salt probably caused the statistically insignificant r values. The more stable atmosphere in the winter, identified as a statistically significant variable by the spatial-temporal modeling, probably minimized the mixing and dilution of the resuspended road salt. As a result, elevated $PM_{10-2.5}$ concentrations were not correlated with the other EMAs. Within EMA 6, the decrease in $PM_{10-2.5}$ over a lateral distance of 300 m from the Southfield Freeway indicated the atmospheric stability limited dispersion and dilution of the suspended road salt over short distances.

The size of the resuspended coarse particles also probably influenced the insignificant r values between EMA 6 and other EMAs. A detailed assessment of the $PM_{10-2.5}$ particle size distribution was not part of the DEARS study design. However, Eleftheriadis and Colbeck (2001) reported the size distributions for a number of crustal elements in the coarse fraction. Their urban data had a modal size generally greater than 5 μm. Assuming their findings are applicable to Detroit, the particles sampled in EMA 6 probably were greater than 5 μm. Therefore, the increased mass of these "larger" particles minimizes their long range transport. Because detailed $PM_{10-2.5}$ size distribution measurements were not obtained using multi-stage impactors or real-time instruments, this hypothesis cannot be confirmed until size distribution and composition of the collected particles is measured using electron microscopy.

Central Site Representativeness

The central monitoring site in EMA 0 is also the MDEQ SLAMS site for Wayne County. Based on the DEARS data for summer 2006 and winter 2007, the applicability of this site for monitoring $PM_{10-2.5}$ for the entire Detroit metropolitan area was evaluated. Monn (2001) noted the spatial variation of $PM_{10-2.5}$ in an urban environment makes the collection of representative samples critical for accurate exposure assessment. When applied to ambient monitoring, representative sample collection refers to placement of the monitor in a location that characterizes the largest percentage of the population. Factors to consider include micro-meteorology, population density, types of sources (point vs. line vs. area), number of sources per square kilometer, and source emission rates.

$PM_{10-2.5}$ at EMA 0 were not as representative for the entire Detroit metropolitan area as $PM_{2.5}$ (Rodes et al., 2008). $PM_{10-2.5}$ in EMA 0 differed by -6 to + 6 $\mu g/m^3$ from the other EMAs during both seasons. EMA 0 exhibited spatial differences in $PM_{10-2.5}$ with EMAs 1 and 5 during the summer, and EMAs 4 and 7 during the winter. Temporal differences were noted between EMAs 0 and 6 during the winter.

Multiple mechanisms possibly can influence the representativeness of EMA 0 as a central monitoring site. Turbulent mixing near the source and atmospheric instability are the two mechanisms that disperse $PM_{10-2.5}$ in the atmosphere. Turbulent mixing probably suspends particles to a sufficient height where the thermally induced atmospheric instability creates the buoyancy to carry $PM_{10-2.5}$ to a height that favors long range transport (Hasegawa et al., 2007). Deposition rates are controlled by Stokes settling (Noll and Aluko, 2006), and therefore are strongly proportional to particle diameter. The tree canopy (Freiman et al., 2006; McDonald et al., 2007) and possibly building density also promote deposition and hinder long range transport, especially as particle diameter increases to 10 μm. Another consideration is the effect of topography on coarse particle transport. Intervening ridges may concentrate coarse particles in one valley and minimize their transport up the slope to the top of the ridge and into neighboring valleys. Although this phenomenon is not applicable in Detroit, cities like Birmingham, AL and Pittsburgh, PA may be affected. Understanding the impact of these mechanisms on the size distribution and composition of the $PM_{10-2.5}$ at multiple locations in an urban airshed is necessary to develop the criteria for representative monitoring of $PM_{10-2.5}$. Alternatively, this type of detailed data can be used to develop models (Georgopoulos et al., 2005) and other exposure surrogates for epidemiological studies and risk assessments.

Conclusions

The evaluation of the new CPEM sampler during DEARS effort demonstrated its versatility and compactness to allow its application in future studies as a stationary or personal sampler. The accuracy and precision of the CPEM $PM_{10-2.5}$ and $PM_{2.5}$ concentrations, compared to a dichotomous sampler, were similar to other portable instruments.

$PM_{10-2.5}$ was measured during summer 2006 and winter 2007 of the DEARS. Summer $PM_{10-2.5}$ exhibited spatial variability across the Detroit airshed, whereas winter $PM_{10-2.5}$ showed spatial and temporal variability. The monitoring locations suggested the summer $PM_{10-2.5}$ spatial gradients were caused by localized industrial sources. Temporal consistency of summer $PM_{10-2.5}$ concentrations caused by atmospheric instability suggested a potential regional $PM_{10-2.5}$ background concentration consisting of particles between 2.5 to 5 μm . Spatial and temporal gradients between an area most likely impacted by suspended road salt and the other areas were found in the winter. A stable atmosphere limited mixing and dispersion of the $PM_{10-2.5}$ particles from the near roadway area, except within 300 m of the monitored freeway. Wind speed and direction, distance, and day of week were variables that did not influence the spatial and temporal concentration gradients between areas. The size distribution of the $PM_{10-2.5}$ particles most likely was also a key factor, and should be measured in future studies. The tree canopy, building density, and topography are other factors to be considered when characterizing $PM_{10-2.5}$ spatial and temporal variability.

$PM_{10-2.5}$ speciation data are needed to definitively identify the sources and components that contribute to the spatial and temporal variability. Besides the standard inorganic and organic chemical speciation analyses, the possibly large microbiological contribution to the $PM_{10-2.5}$ mass should be quantified. This speciation data then could identify a marker, like sulfate for $PM_{2.5}$, for penetration of $PM_{10-2.5}$ of ambient origin into buildings.

This research provided additional information for the development of National Ambient Air Quality Standard compliance monitoring requirements for $PM_{10-2.5}$. These findings indicated $PM_{10-2.5}$, although highly correlated in some instances, vary in magnitude sufficiently that a central monitoring site may not adequately represent the population's exposure in a complex urban airshed. Factors that should be considered when developing compliance monitor placement criteria are micro-meteorology, population density, types of sources (point vs. line vs. area), number of sources per square kilometer, and source emission rates.

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REFERENCES

Brunekreef, B. and Forsberg, B., 2005. Epidemiological evidence of effects of coarse airborne particles on health. European Respiratory Journal 26(2), 309-318.

Burton, R., Suh, H., and Koutrakis, P., 1996. Spatial variation in particulate concentrations within metropolitan Philadelphia. Environmental Science and Technology 30, 400-407.

Case, M.W., Williams, R., Yeatts, K., Chen, F., Scott, J., Svendsen, E., and Devlin, R. 2008. Evaluation of a direct personal coarse particulate matter monitor. Atmospheric Environment. 42, 4446-4452.

Chen, F., Williams, R., Svendsen, E., Yeatts, K., Creason, J., Scott, J., Terrell, D., and Case, M., 2007. Coarse particulate matter concentrations from residential outdoor sites associated with the North Carolina Asthma and Children's Environment Studies (NC-ACES). Atmospheric Environment 41, 1200-1208.

Chow, J.C. and Watson, J.G., 2008. New Directions: Beyond compliance air quality measurements. Atmospheric Environment. 42, 5166-5168.

Demokritou, P., Gupta, T., Ferguson, S., and Koutrakis, P., 2003. Development and laboratory performance evaluation of a personal cascade impactor. Journal of Air and Waste Management Association 52, 1230-1237.

Dockery, D., Pope III, C.A., Xu, X., Spengler, J.D., Ware, J.H., Fay, M.E., Ferris, B.J., and Speizer, F.E., 1993. An association between air pollution and mortality in 6 US cities. The New England Journal of Medicine 329, 1753-1759.

Eleftheriadis, K. & Colbeck, I., 2001. Coarse atmospheric aerosol: size distributions of trace elements. Atmospheric Environment 35, 5321-5330.

Freiman, M.T., Hirshel, N., and Broday, D.M., 2006. Urban-scale variability of ambient particulate matter attributes. Atmospheric Environment 40, 5670-5684.

Georgopoulos, P., Wang, S., Vyas, V., Sun, Q., Burke, J., Vedantham, R., McCurdy, T., and Ozkaynak, H., 2005. A source to dose assessment of population exposures to fine PM and ozone in Philadelphia, PA during a summer 1999 episode. Journal of Exposure Science and Environmental Epidemiology 15, 439-457.

Guerra, S., Lane, D., Marotz, G., Carter, R., and Hohl, C., 2006. Effects of wind direction on coarse and fine particulate matter concentrations in southeast Kansas. Journal of the Air & Waste Management Association 56, 1525-1531.

Hasegawa, S., Wakamatsu, S., Ohara, T., Itano, Y., Saitoh, K., Hayasaki, M., and Kobayashi, S., 2007. Vertical profiles of ultrafine to supermicron particles measured by aircraft over Osaka metropolitan area in Japan. Atmospheric Environment 41, 717-729.

Hopke, P.K., Ramadan, Z., Paatero, P., Norris, G.A., Landis, M.S., Williams, R.W., and Lewis, C.W. 2003. Receptor modeling of ambient and personal exposure samples: 1998 Baltimore Particulate Matter Epidemiology-Exposure Study, Atmospheric Environment, Volume 37, 3289-3302.

Ito, K., Thurston, G., Nadas, A., and Lippmann, M., 2001. Monitor-to-monitor temporal correlation of air pollution and weather variables in the North-Central U.S. Journal of Exposure Analysis and Environmental Epidemiology 11, 21-32.

Kim, E., Hopke, P.K., Pinto, J.P., and Wilson, W.E., 2005. Spatial variability of fine particle mass, components, and source contributions during the regional air pollution study in St. Louis. Environmental Science and Technology 39, 4172-4179.

Kenny, L.C., Aitken, R., Chalmers, C., Fabries, J.-F., Gonzalez-Fernandez, E., Kromhout, H., Liden, G., Mark, D., Riediger, G., and Prodi, V., 1997. Outcome of a collaborative European study of personal inhalable sampler performance. Annals Occupational Hygiene. 41, 135-154.

Kinney,P.L., Aggarwal, M., Northridge, M.E., Janssen, N.A.H., and Shepard, P., 2000. Airborne concentrations of $PM_{2.5}$ and diesel exhaust particles on Harlem sidewalks: a community-based pilot study. Environmental Health Perspectives 108 (3), 213–218.

Landis, M., Norris, G., Williams, R., and Weinstein, J., 2001. Personal exposures to $PM_{2.5}$ mass and trace elements in Baltimore, Maryland. Atmospheric Environment 35, 6551-6524.

Lawless, P. and Rodes, C., 1999. Maximizing data quality in the gravimetric analysis of personal exposure sample filters. Journal of the Air & Waste Management Association 49, 1039-1049.

McDonald, A.G., Bealey, W.J., Fowler, D., Dragosits, U., Skiba, U., Smith, R.I., Donovan, R.G., Brett, H.E., Hewitt, C.N., and Nemitz, E., 2007. Quantifying the effect of urban tree planting on concentrations and depositions of PM_{10} in two UK conurbations. Atmospheric Environment 41, 8455-8467.

Monn, C., 2001. Exposure assessment of air pollutants: a review on spatial heterogeneity and indoor/outdoor/personal exposure to suspended particulate matter, nitrogen dioxide, and ozone. Atmospheric Environment 35, 1-32.

Noll, K. and Aluko, O., 2006. Changes in large particle size distribution due to dry deposition processes. Journal of Aerosol Science 37, 1797-1808.

Ott, D., Cyrs, W., and Peters, T., 2008. Passive measurement of coarse particulate matter, $PM_{10-2.5}$. Journal of Aerosol Science 39, 156-167.

Phillips, M.J., Rodes, C.E., Thornburg, J., Shamo, F., Whitmore, R., Chowdhury, D., Allpress, J., Vette, A., and Williams, R.W., 2008. Optimization of recruitment and retention strategies for the Detroit Exposure and Aerosol Research Study. submitted to Atmospheric Environment.

Pinto, J., Lefohn, A.S., and Shadwick, D.S., 2004. Spatial variability of $PM_{2.5}$ in urban areas in the United States. Journal of the Air & Waste Management Association 54, 440-449.

Querol, X., Alastuey, A., Moreno, T., Viana, M., Castillo, S., Pey, Rodriguez, S., Artinano, B., Salvador, P., Sanchez., M., Santos, S., Garraleta, M.D., Fernandez-Patier, R., Moreno-Grau, S., Negral, L., Minguillon, M.C., Monfort, E., Sanz, M.J., Palomo-Marin, R., Pinilla-Gil, E., Cuevas, E., de la Rosa, J., and Sanchez de la Campi, A., 2008. Spatial and temporal variations in airborne particulate matter (PM_{10} and $PM_{2.5}$) across Spain 1999-2005. Atmospheric Environment. 42, 3964-3979.

Reid, E. A., J. S. Reid, M. M. Meier, M. R. Dunlap, S. S. Cliff, A. Broumas, K. Perry, and H. Maring (2003), Characterization of African dust transported to Puerto Rico by individual particle and size segregated bulk analysis, J. Geophys. Res., 108(D19), 8591.

Rodes, C.E., Thornburg, J., 2006. Breathing Zone Exposure Assessment in Aerosols Handbook: Measurement, Dosimetry, and Health Effects. Eds: Ruzer, L.S., Harley, N.H. CRC Press, Boca Raton, FL.

Rodes., C.E., Lawless, P.A., Thornburg, J., Williams, R.W., and Croghan, C., 2008. DEARS Particulate Matter Relationships for Personal, Indoor, Outdoor, and Central Site Settings for a General Population Cohort. submitted to Atmospheric Environment.

Suh, H.H., Nishioka, Y., Allen, G.A., Koutrakis, P., and Burton, R.M., 1997. The metropolitan acid aerosol characterization study: results from the summer 1994 Washington, DC field study. Environmental Health Perspectives 105, 826-834.

US EPA. 2007. Ambient air monitoring reference and equivalent methods. *40 CFR 53*.

Wagner, J., and Leith, D., 2001. Passive aerosol sampler. Part I: Principle of operation. Aerosol Science and Technology 34(2), 186-192.

Williams, R., Suggs, J., Rodes, C., Lawless, P., Zweidinger, R., Kwok, R., Creason, J., and Sheldon, L., 2000. Comparison of $PM_{2.5}$ and PM_{10} monitors. Journal of Exposure Analysis and Environmental Epidemiology. 10, 518-532.

Williams, R., Oct. 2005. EPA's Detroit Exposure and Aerosol Research Study. EPA Research Highlights. AWMA Environmental Manager, pp43.

Williams R., Case, M., Yeatts, K., Chen, F., Scott, J., Svendsen, E., and Delvin, R. 2008a. Personal coarse particulate matter exposures in an adult cohort. Atmospheric Environment. In press. doi:10.1016/j.atmosenv.2008.05.034.

Williams, R.W., Rea, A., Vette, A., Croghan, C., Whitaker, D., Stevens, C., Mcdow, S., Fortmann, R., Sheldon, L., Wilson, H., Thornburg, J., Phillips, M., Lawless, P., Rodes, C., and Daughtrey, H., 2008b. The design and field implementation of the Detroit Exposure and Aerosol Research Study. Journal of Exposure Science and Environmental Epidemiology. doi: 10.1038/jes.2008.61

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

Wilson, W. and Suh, H., 1997. Fine particles and coarse Particles: concentration relationships relevant to epidemiologic studies. Journal of the Air & Waste Management Association. 47, 1238-1249.

Yeatts, K., Svendsen, E., Creason, J., Alexis, N., Herbst, M., Scott, J., Kupper, L., Williams, R., Neas, L., Cascio, W., Devlin, R., and Peden, D., 2007. Coarse Particulate Matter ($PM_{2.5-10}$) Affects Heart Rate Variability, Blood Lipids, and Circulating Eosinophils in Adults with Asthma. Environmental Health Perspectives 115(5), 709-714.

Zhao, W., Hopke, P., Norris, G., Williams, R., and Paatero, P., 2006. Source apportionment and analysis on ambient and personal exposure samples with a combined receptor model and an adaptive blank estimation strategy. Atmospheric Environment 40, 3788-3801.

Figure 1. Locations of the central monitoring site (EMA 0) in relation to the Enumeration Measurement areas (EMAs) where outdoor residential $PM_{10-2.5}$ and $PM_{2.5}$ measurements were conducted. EMA 7 is located approximately 23 km west of the central site, along Interstate 94.

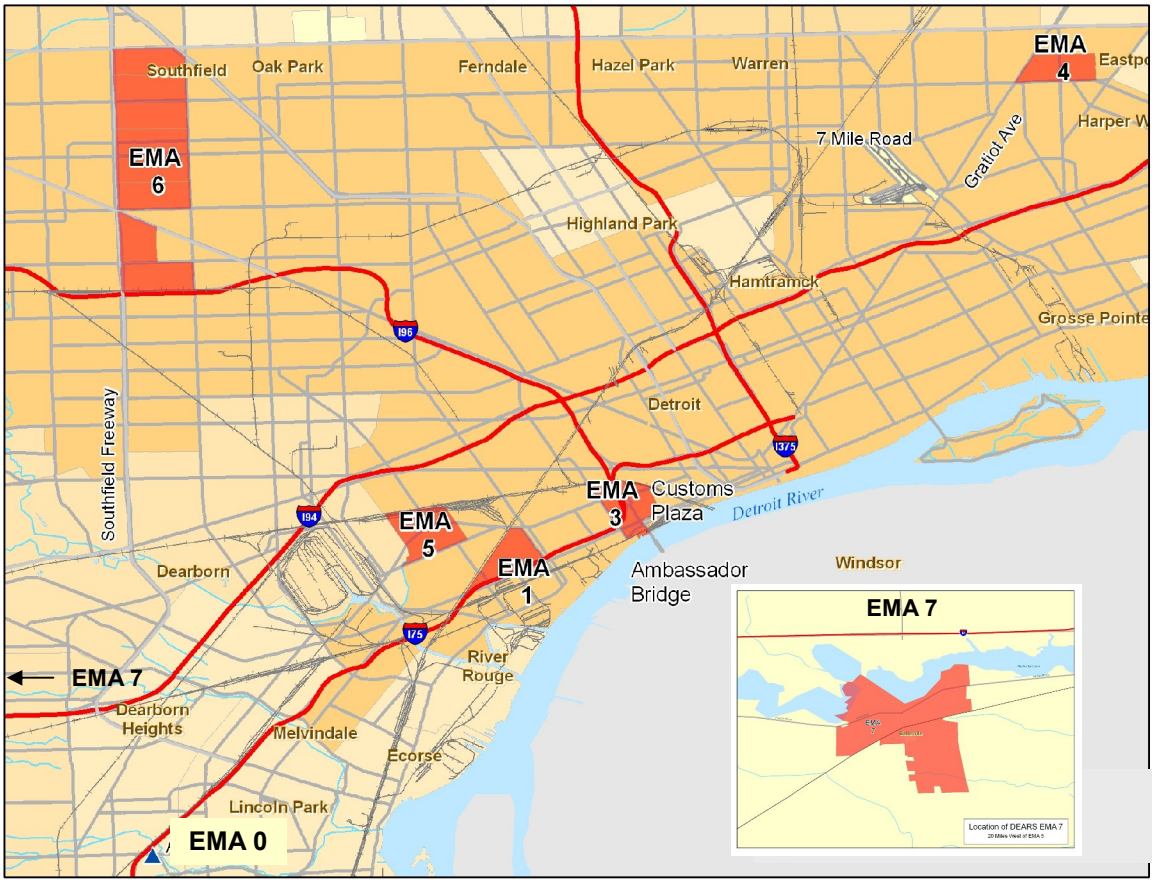
Figure 2. Scatterplot of $PM_{10-2.5}$ and $PM_{2.5}$ concentrations measured by the CPEM and Andersen Dichotomous sampler at the central monitoring site (EMA 0)

Figure 3. Accuracy of the CPEM compared to the Andersen Dichotomous sampler. $PM_{10-2.5}$ and $PM_{2.5}$ sample sizes were 72 and 66. A fractional difference of 0 indicates perfect agreement.

Figure 4. Precision of the CPEM based on collocated duplicate samples. Precision presented as the coefficient of variation in the measurements. $PM_{10-2.5}$ and $PM_{2.5}$ sample sizes were 72 and 66.

Figure 5. Seasonal and spatial $PM_{10-2.5}$ variability in each EMA. Error bars show 1 standard deviation of the geometric mean concentration. EMAs marked with asterisks had statistically significant differences between seasons ($p < 0.05$). Concentration bars with number symbols (#) or carets (^) indicate statistically significant spatial differences in summer or winter, respectively.

Figure 1.


Southfield
Oak Park
Ferndale
Hazel Park
Warren
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Eastpo
Harper W
7 Mile Road
Gratiot Ave
EMA 6
Highland Park
Hamtramck
Grosse Pointe
196
Detroit
Southfield Freeway
I375
EMA 3
Customs Plaza
Detroit River
194
EMA 5
EMA 1
Ambassador Bridge
Windsor
Dearborn
175
River Rouge
EMA 7
Dearborn Heights
Melvindale
Ecorse
Lincoln Park
EMA 0
EMA 7
Location of DEARS EMA 7

Figure 2.

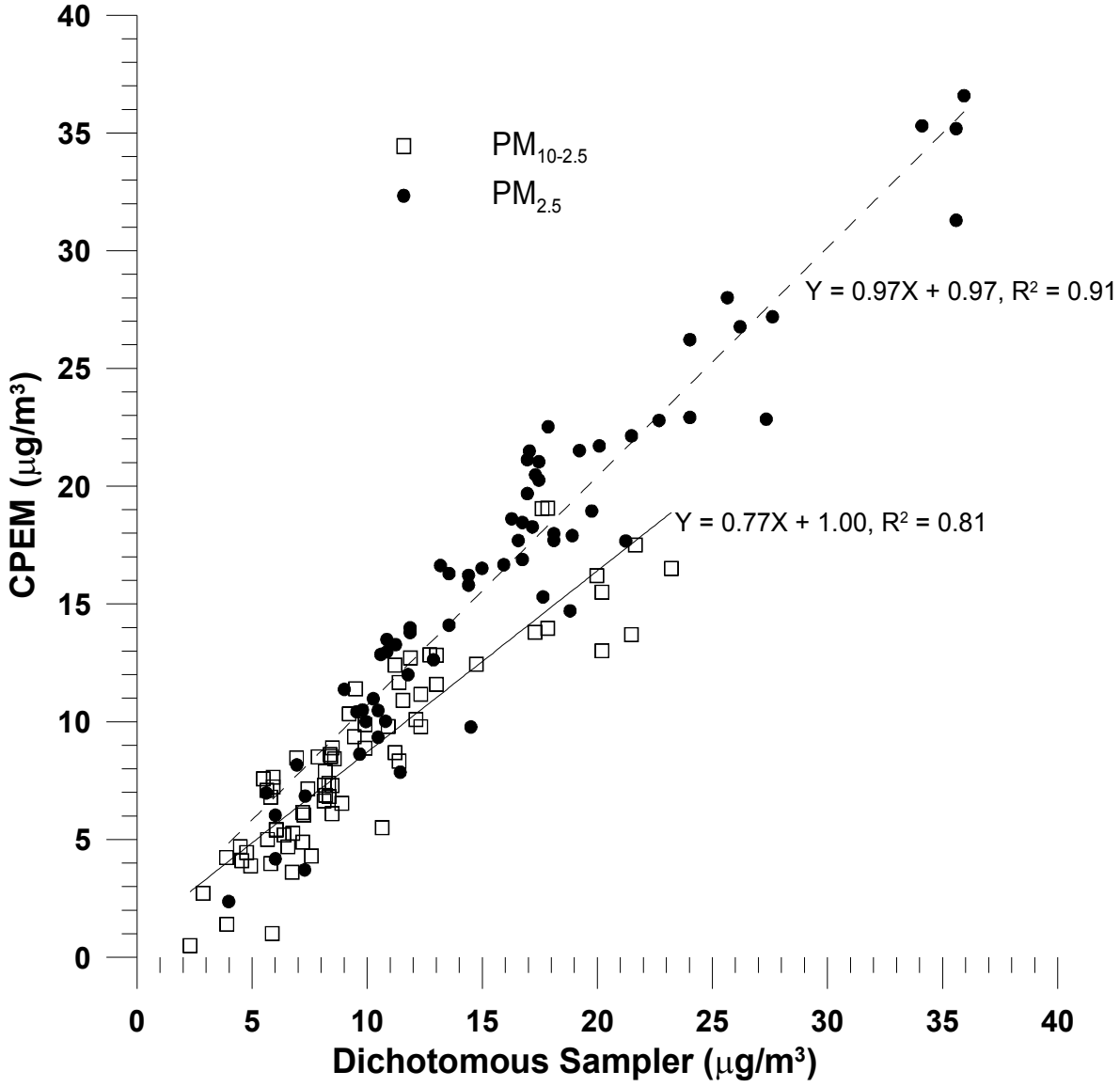
PM10-2.5
PM2.5
Y = 0.97X + 0.97, R2 = 0.91
Y = 0.77X + 1.00, R2 = 0.81
CPEM (μg/m3)
Dichotomous Sampler (μg/m3)
0
5
10
15
20
25
30
35
40

Figure 3.

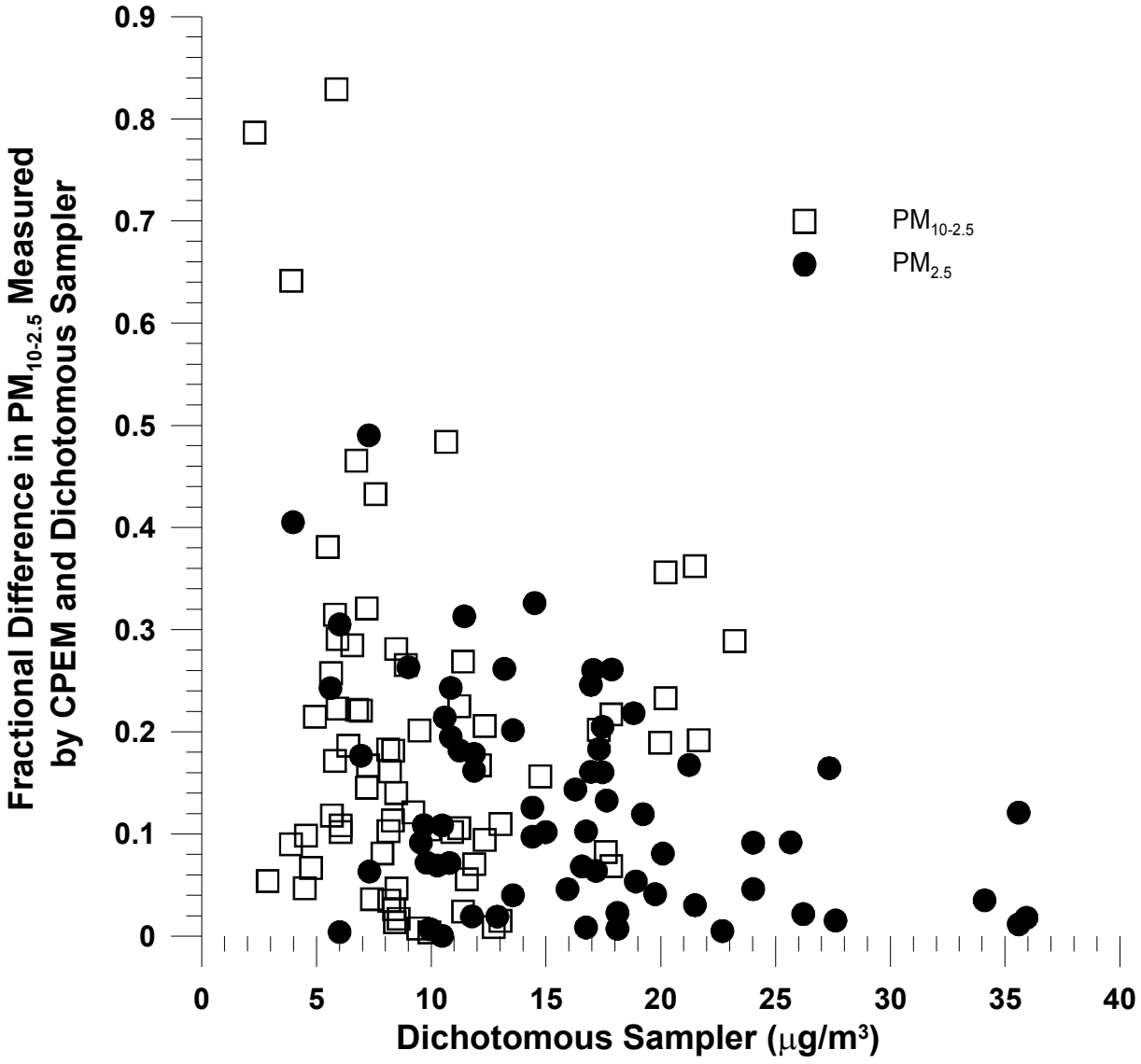
Fractional Difference in PM10-2.5 Measured by CPEM and Dichotomous Sampler
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
PM10-2.5
PM2.5
0
5
10
15
20
25
30
35
40
Dichotomous Sampler (μg/m3)

Figure 4

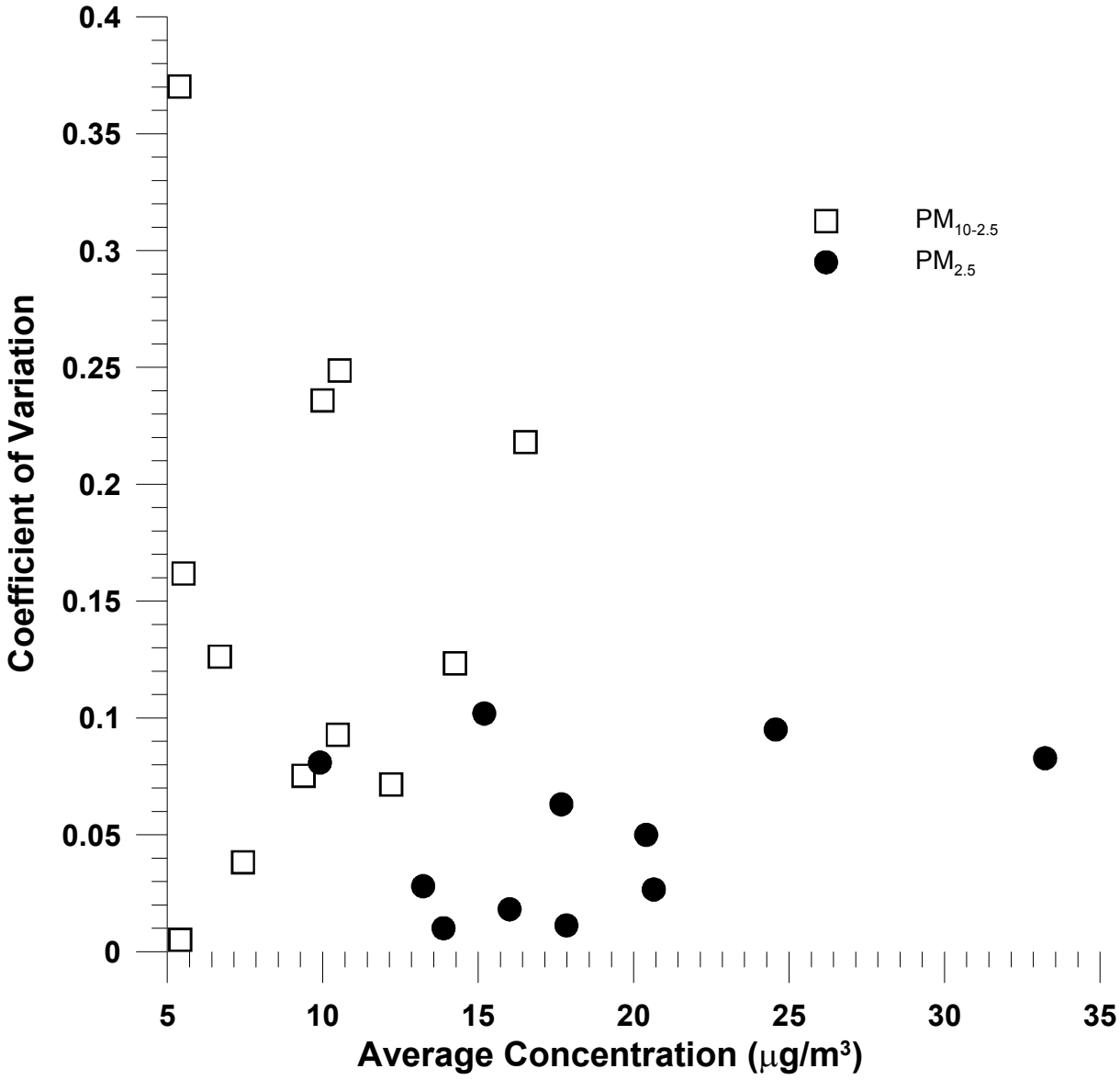
0.4
0.35
0.3
0.25
0.2
0.15
0.1
0.05
0
Coefficient of Variation
PM10-2.5
PM2.5
5
10
15
20
25
30
35
Average Concentration (μg/m3)

Figure 5.

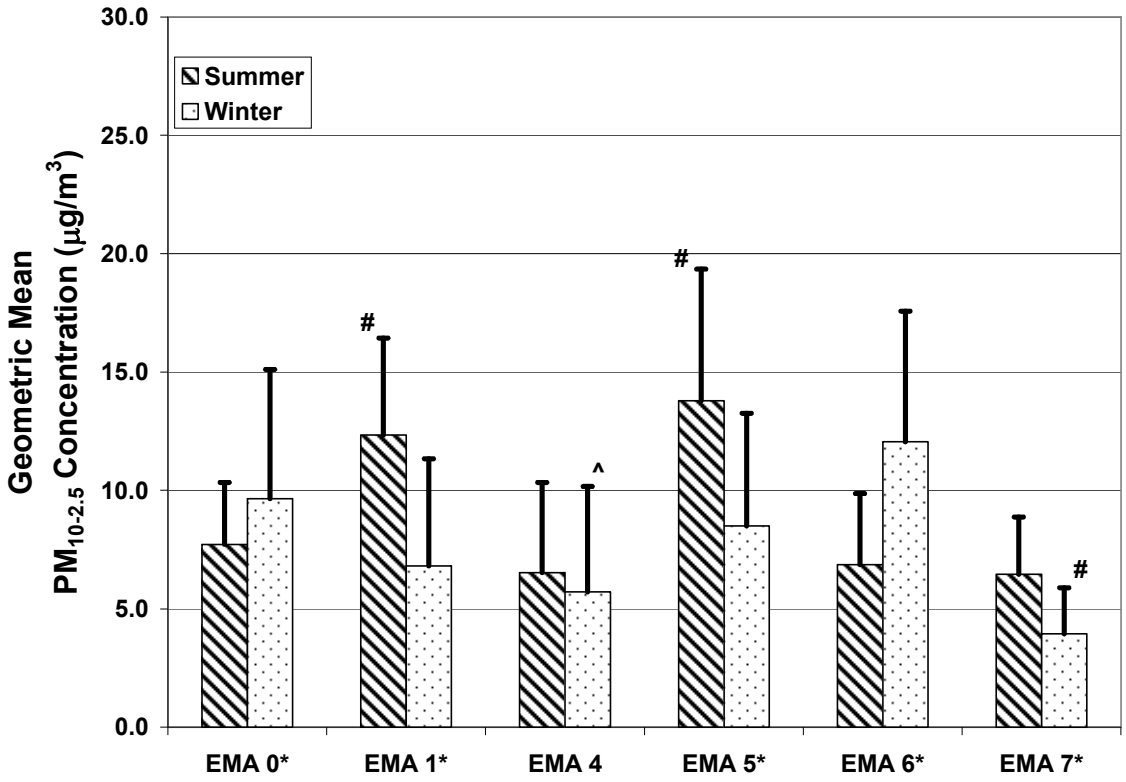
Summer
Winter
Geometric Mean
PM10-2.5 Concentration (μg/m3)
30.0
25.0
20.0
15.0
10.0
5.0
0.0
#
#
^
#
EMA 0*
EMA 1*
EMA 4
EMA 5*
EMA 6*
EMA 7*

Table 1. Statistical analysis of the $PM_{10-2.5}$ spatial variability (Location) within an EMA accounting for daily variations in $PM_{10-2.5}$ (Day)

Table 2. Coefficients of divergence (COD) and Pearson correlation coefficients (r) between $PM_{10-2.5}$ measured at the EMAs. Correlations in bold text are statistically insignificant (p-value > 0.05) to highlight temporal differences between EMAs.

Table 1.

Summer

	EMA 1 (n = 9)		**EMA 4 (n = 20)**		**EMA 5 (n = 10**		**EMA 6 (n = 19)**	
	df	p-value	df	p-value	df	p-value	df	p-value
Location	1	0.26	1	0.41	1	0.68	1	0.48
Day	8	*< 0.001*	19	*0.002*	9	*0.003*	18	*0.001*
Error	8		19		9		18	
Group A mean	12.6 ± 4.3		6.5 ± 3.3		13.0 ± 6.5		6.9 ± 2.6	
Group B mean	13.2 ± 4.4		7.3 ± 5.1		13.5 ± 4.6		6.5 ± 3.2	

Winter

	EMA 1 (n = 15)		**EMA 4 (n = 15)**		**EMA 6 (n = 15)**	
	df	p-value	df	p-value	df	p-value
Location	2	0.47	1	0.89	1	*< 0.001*
Day	9	*< 0.001*	14	0.07	14	*< 0.001*
Error	13		14		14	
Group A mean	4.9 ± 3.7		6.1 ± 5.5		14.5 ± 5.2	
Group B mean	5.8 ± 4.4		6.3 ± 4.3		11.1 ± 5.5	
Group C mean	3.0 ± 2.6		-		-	

Note: EMA 5 did not have duplicate participants in the same week in Winter.

Table 2.

Summer					Winter				
Cluster	**EMAs**	**Distance (km)**	**COD**	**r**	**Cluster**	**EMAs**	**Distance (km)**	**COD**	**r**
1	1 to 5	2	0.17	0.49	1a	5 to 6	16	0.29	***0.13***
2	4 to 6	21	0.23	0.57		1 to 6	16	0.29	***0.09***
	0 to 7	23	0.21	0.60		0 to 6	24	0.26	***0.31***
	0 to 6	24	0.23	0.52	1b	4 to 6	21	0.43	***0.03***
	0 to 4	31	0.23	0.54		6 to 7	34	0.50	***0.10***
	6 to 7	34	0.24	0.56	2	1 to 5	2	0.31	0.70
	4 to 7	48	0.26	0.32		0 to 5	11	0.30	0.63
3	0 to 5	11	0.33	0.41		0 to 1	12	0.31	0.76
	0 to 1	12	0.28	0.53	3	1 to 4	18	0.40	0.50
	5 to 6	16	0.40	0.45		4 to 5	19	0.41	0.47
	1 to 6	16	0.33	0.63		0 to 7	23	0.46	0.61
	1 to 4	18	0.37	0.59		5 to 7	31	0.42	0.67
	4 to 5	19	0.41	0.28		0 to 4	31	0.47	0.54
	5 to 7	31	0.41	0.37		1 to 7	32	0.40	0.62
	1 to 7	32	0.33	0.54		4 to 7	48	0.40	0.64