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Spatial Characteristics of Fine Particulate Matter: Identifying Representative Monitoring Locations in Seattle, Washington

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ABSTRACT

This study investigates how $PM_{2.5}$ varies spatially and how these spatial characteristics can be used to identify potential monitoring sites that are most representative of the overall ambient exposures to $PM_{2.5}$ among susceptible populations in the Seattle, WA, area. Data collected at outdoor sites at the homes of participants of a large exposure assessment study were used in this study. Harvard impactors (HIs) were used at 40 outdoor sites throughout the Seattle metropolitan area. Up to six sites at a time were monitored for 10 consecutive 24-hr average periods. A fixed-effect analysis of variance (ANOVA) model that included date and location effects was used to analyze the spatial variability of outdoor $PM_{2.5}$ concentrations. Both date and location effects were shown to be highly significant, explaining 92% of the variability in outdoor $PM_{2.5}$ measurements. The day-to-day variability was 10 times higher than the spatial variability between sites. The site mean square was more than twice the error mean square, showing that differences between sites, while modest, are potentially an important contribution to measurement error.

Variances of the model residuals and site effects were examined against spatial characteristics of the monitoring sites. The spatial characteristics included elevation, distance from arterials, and distance from major $PM_{2.5}$ point sources. Results showed that the most representative $PM_{2.5}$ sites were located at elevations of 80–120 m above sea level, and at distances of 100–300 m from the nearest arterial road. Location relative to industrial $PM_{2.5}$ sources is not a significant predictor of residential outdoor $PM_{2.5}$ measurements. Additionally, for sites to be representative of the average population exposures to $PM_{2.5}$ among those highly susceptible to the health effects of $PM_{2.5}$, areas of high elderly population density were considered. These representative spatial characteristics were used as multiple, overlapping criteria in a Geographic Information System (GIS) analysis to determine where the most representative sites are located.

IMPLICATIONS

Air pollutant monitoring is used to determine compliance with the National Ambient Air Quality Standards, detect sources of pollutants, and investigate the health effects of air pollution in epidemiologic studies. Monitor location determines to what extent monitoring sites are representative of overall ambient concentrations to which a population of interest is most exposed. This paper provides a framework for determining the most representative locations for monitoring population exposures to $PM_{2.5}$. This framework could be used by both regulatory agencies and researchers. Using measurements from sites that are most representative of population exposures to $PM_{2.5}$ should lead to more rational control assessments and more robust health findings in epidemiologic studies.

INTRODUCTION

Many studies have shown that community levels of particulate matter (PM) less than 2.5 μm in diameter ($PM_{2.5}$) are significantly associated with adverse health conditions,

including increased mortality rate, increased respiratory irritation, use of asthma medications, and hospitalizations for asthma.[1-9] Elderly persons with chronic obstructive pulmonary disease (COPD) or coronary heart disease are most susceptible to these effects.[1-7] The U.S. Environmental Protection Agency (EPA) promulgated National Ambient Air Quality Standards for $PM_{2.5}$ in 1997 that require states to establish $PM_{2.5}$ monitoring networks. These network monitoring data are often used in epidemiology studies to estimate population exposures to pollutants and the associated health effects,[1-9] although the implications of using measurements from fixed monitoring sites for population or individual exposure assessment are still being debated.[10-12]

With the need for representative monitoring of $PM_{2.5}$, regulatory agencies face the question of where to site the monitor(s) to suit both regulatory and health study needs. EPA defines six representative measurement scales—micro, middle, neighborhood, urban, regional, and national/global scales—in its guidelines for siting State and Local Air Monitoring Stations (SLAMS) and National Air Monitoring Stations (NAMS).[13] The SLAMS/NAMS network is designed to be representative of the spatial area being studied and to characterize one or more of the following air quality features: (1) highest concentrations in the network area, (2) representative concentrations in areas of high population density, (3) impacts from significant sources, (4) general background concentrations, (5) regional pollutant transport among populated areas, and (6) welfare-related impacts in more rural or remote areas.[13] Selection of a monitoring location is a complex process, which should take into account the location of sources of emissions, spatial variability of ambient pollutant concentrations, meteorological conditions, and population density. In addition, the siting guidance prepared by Watson et al.[14] defined representative population sites for $PM_{2.5}$ to be less than a 20% variation in the mean $PM_{2.5}$ concentration of the entire urban area. Ideally, air quality monitoring sites should address the needs of both the regulatory and scientific research communities.[15] Such monitoring sites should be suitable for the assessment of both compliance and health risks and, thus, should include sites that are located where the measurements are most representative of urban population exposure [air quality feature (2)].

In the past 2 years, the Washington Department of Ecology has faced this siting issue in Seattle for $PM_{2.5}$ monitors, PM speciation monitors, and hazardous air pollutant monitors. To optimize scarce resources, a decision team was formed and a "siting" workshop was conducted on May 20, 2000, to consider the criteria. Preliminary siting decisions were made, based on results from past and current studies on air quality at a limited number of locations, location of elderly populations, and existing regulatory siting requirements.[16-18] These latter requirements, including elevation, distance from roads, and population density, were then used in a Geographic Information System (GIS) analysis to highlight the monitoring locations of interest. In this paper, we significantly expand that initial analysis by using a comprehensive set of outdoor residential $PM_{2.5}$ measurements to quantify the spatial characteristics that make a site representative of the average population exposure in Seattle.

Seattle's diverse topography, traffic patterns, and the widespread use of woodstoves contribute to spatial variability and make monitor siting a challenge. Elevation is important in siting because the atmospheric mixing height for pollutant dispersion determines the vertical profile of $PM_{2.5}$. In some cases, elevation represents the exposure of residents living at that particular location. Using a mobile nephelometer, Larson et al. observed an inverse relationship between elevation and particulate concentration (with a size range of 0.1–1.4 μm) under nearly stagnant conditions.[17] They reported that particles tended to concentrate in valleys during nighttime drainage flow, with concentration peaking in the Seattle suburbs during wintertime inversions. Sheppard et al.[18] investigated potential exposure misclassification using estimates from central site measurements in Seattle. A topographic index, defined as a function of elevation and slope, was found to relate to population exposure estimates. Their results indicated that distribution of particle concentration depended on the distribution of particle sources and surface features that determined airflow.[18] Their study, however, only used four fixed monitoring sites to estimate population exposure and did not quantitatively consider proximity to sources of directly emitted $PM_{2.5}$ (such as traffic or industrial point sources).

Guidance for monitor siting[13,14] as well as papers examining spatial variation of air pollutants within an urban area are available.[17-23] Chan and Hwang[24] provided a statistical method to calculate the representativeness of an existing monitoring site. Croxford and Penn[25] focused on issues regarding curbside monitoring for CO. However, we know of no study that provides a systematic, objective, and quantitative means for selecting new monitoring sites that are to be used for assessment of both compliance and population health risks. The goal of this paper is to build a framework that would help decision-makers select ambient air monitoring sites. This framework builds on three major components: (1) determination of the spatial variability of $PM_{2.5}$; (2) determination of the spatial characteristics of a monitoring site that would best represent the overall population exposure to ambient $PM_{2.5}$, especially among those who are most susceptible to the effects of $PM_{2.5}$; and (3) determination of the location of areas with these spatial characteristics using a GIS.

METHODS

Data

This study used the outdoor $PM_{2.5}$ measurements taken as a part of a larger study of personal particle exposure assessment of high-risk subpopulations.[26] The high-risk subpopulations in this analysis included elderly COPD patients living in the Seattle metropolitan area. Subjects were recruited through retirement group establishments, senior centers, health clinics, retirement associations, and newspaper advertisements. Our monitoring network is based on subject availability with the constraint that all sites be middle- to neighborhood-scale according to EPA siting criteria.[13,14] Subject selection met strict health guidelines for a given disease state. A total of 40 residences of the subjects were monitored over thirteen 10-day monitoring sessions between October 26, 1999, and August 11, 2000. For the 13 monitoring sessions, three sessions occurred during the fall, three during the winter, five in the spring, and two in the summer. These 13 monitoring sessions were to cover low, median, and high wood smoke periods during the fall, winter, and spring. Each session, starting on a Tuesday, included between four and six home sites and one central site. Home sites were monitored in as many as four different sessions. The Beacon Hill site, an urban-scale monitoring site operated by the Washington Department of Ecology, was used as the "central site" and was monitored during all sessions except for a few days in the first session. The maximum distance between home sites was 19.5 mi (31.4 km), and no home was farther than 14.5 mi (23.3 km) from the central site.

The $PM_{2.5}$ samples were collected using Harvard impactors (HIs) (Air Diagnostics and Engineering, Inc.). The HI consists of one impactor stage and one filter stage. Particles of aerodynamic diameter greater than 2.5 μm are removed by an oiled impaction plate, and particles of aerodynamic diameter smaller than or equal to 2.5 μm are collected downstream on a 37-mm Teflon filter [poly(tetrafluoroethylene) (PTFE) with support ring, cat. no. 225-1709, SKC, Inc.]. The HI is connected to a pump (Gast Manufacturing, Inc., model DOA-V191-AA) that operated at 10 L/min. The on and off pump flow rates were calibrated daily with a DryCal flow meter (Bios International, DC-Lite). The monitoring duration was from 4:00 p.m. to 4:00 p.m. (±2 hr) at each site for 10 days, resulting in 10 samples collected at each site. The exposed filters were weighed using a microbalance (Mettler Toledo, model UMT2) in our controlled environmental weighing chamber. All filters were conditioned in the weighing chamber for at least 24 hr before and after exposure at a temperature of 22 ± 2 °C and relative humidity of 35 ± 3%.[27] For quality control, the total number of HI field blanks equaled 11% of all samples and field duplicates equaled 10% of all samples. The limit of detection of the HI measurements, defined as 3 times the SD of field blanks (N = 132), is 1.0 μg/m^3. The precision of the HI, calculated as 1 SD of the differences between field duplicates divided by $\sqrt{2}$ (N = 52 pairs), is 0.6 μg/m^3. The HI measurements agreed very well with the Beacon Hill site Federal Reference Method for $PM_{2.5}$ measurements (mean difference = 0.4 μg/m^3, R^2 = 0.99, N = 55).

In addition to $PM_{2.5}$, 10-day average NO_2 measurements were collected using the Ogawa passive badges for NO_2 (Ogawa & Co.).[28] The passive badge contained two cellulose filters coated with triethanolamine to collect NO_2. These NO_2 measurements were taken along with the HI measurements at all sites. The limit of detection based on field blanks (21% of all samples, N = 45) is 0.87 ppb for 10-day measurements, and the precision based on field duplicate samples (17% of all samples, N = 36 pairs) is 1.7 ppb. The mean difference between the badge and the central site chemiluminescence measurements is 0.2 ± 1.9 ppb (N = 11).

All measurements were screened for sampling problems (e.g., hole in the filter, pump malfunction), and anomalous samples were flagged and removed from analysis. As a result, 94% of the samples were used in the subsequent analysis. Monitoring sites were also examined for anomalies. Sites that were located adjacent to a major freeway, Interstate 5, had extremely high measurement variability. Several sites had unusual monitor placement (confined in a courtyard by high rises). One site was almost 700 m from the nearest arterial (few areas in Seattle are greater than 300 m from an arterial). Finally, sites with fewer than 7 days of valid samples during any given session were considered to lack statistical power for subsequent analysis. Seven sites were removed from the final analysis for these reasons. When a monitor was placed at a different location at the same address for a repeat session, usually on different floors at the same group retirement establishment, it was treated as a separate site. Because $PM_{2.5}$ measurements at the Beacon Hill site were collected in all 13 sessions, 30% of these measurements were randomly selected for analysis so that Beacon Hill was not overrepresented in our analysis. As a result, 37 outdoor sites (Figure 1 and Table 1) with a total of 497 measurements were considered in this analysis.

Statistical Modeling

Not all sites were monitored during each of the 13 sessions. Therefore, it is important to account for daily variations in $PM_{2.5}$ levels due to atmospheric conditions that might otherwise be ascribed to spatial variability. To take into account the effect of location and day-to-day

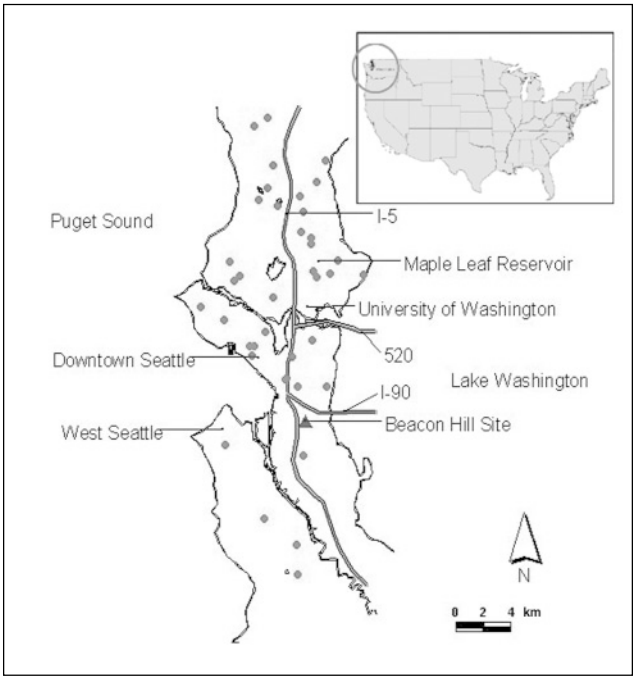

Figure 1. Map of monitoring locations in Seattle (filled circles indicate the monitoring sites).

variation, we constructed a factorial fixed-effect analysis of variance (ANOVA) model as follows:

$$\mathrm{Log}_{10}(\mathrm{PM}_{2.5})_{ijk} = \mu + SITE_i + DATE_j + \varepsilon_{ijk} \quad \text{(model 1)}$$

This model predicts the log of outdoor $PM_{2.5}$ for observation *k* at site *i* on date *j* from the overall average of all sites (μ) after accounting separately for day-to-day and site-to-site variability. $PM_{2.5}$ in the model refers to the 24-hr average outdoor $PM_{2.5}$ concentration (in μg/m^3) taken from a particular site during a given sampling period (4:00 p.m.–4:00 p.m.). The log transform of $PM_{2.5}$ concentration was found to produce normally distributed model residuals. The *DATE* factor accounts for the day-to-day variability in the overall site mean, while the *SITE* factor accounts for systematic differences between monitoring locations between sampling periods. The estimate of the *SITE* factor is referred to as the "site effect." Because it is not possible to know the "true" outdoor concentration of ambient $PM_{2.5}$ in Seattle, we used the overall average concentration of all sites on a particular day as the best estimate for the "true" population exposure to $PM_{2.5}$. The error term, ε_{ijk}, represents the residual error of the model. The variance of ε_{ijk} or VAR(ε) for a given site *i* is indicative of how well the concentration at that site is correlated with the overall mean of all sites on any given day *j*.

There is a possibility that some sites could be spatially correlated and thus bias the model toward the mean of the oversampled areas. However, based on the relatively uniform spatial distribution of the sites (see Figure 1) and the high degree of spatial correlation between all sites (median pairwise Pearson's $r = 0.89$ for the 135 pairs of concurrent sites), oversampling or overrepresenting of certain areas is not a major concern in this analysis. From the epidemiology viewpoint, a good site would produce little error or variability in its ability to measure the true overall mean ambient concentration for a given sampling period.[11] A site that has low precision [high VAR(ε)] but high accuracy (low $SITE_i$) measures the overall mean concentration, but does not capture critical peaks or changes in the daily average concentration over time. A site with high precision [low VAR(ε)] and low accuracy (high $SITE_i$) is consistently high or low compared with the overall mean concentration, but it closely reflects changes in the daily average concentration over time. Precise but inaccurate measurements could be systematically corrected to represent the overall mean value. Both VAR(ε) and $SITE_i$ were categorized according to different spatial characteristics to determine factors affecting outdoor $PM_{2.5}$ concentrations. These spatial characteristics (elevation, distance from arterials, and distance from point sources) were compared against VAR(ε) and $SITE_i$ both graphically and in linear regression models.

GIS

We used a GIS to analyze spatial data. The GIS allows display of spatial data as well as manipulation of data to produce map overlays. The GIS study area, selected after the site selection, consists of the Seattle metropolitan area,

Table 1. Summary of $PM_{2.5}$ (24-hr average) and NO_2 (10-day average) measurements and spatial characteristics of the study sites.

Characteristic	N	5%	10%	25%	50%	75%	90%	95%
$PM_{2.5}$ (μg/m^3)	497	3.2	3.8	5.3	7.5	10.8	15.1	18.4
NO_2 (ppb)	78	7.4	8.8	12.8	16.2	19.5	24.2	28.1
Site elevation (m)	37 sites	33	51	68	91	121	135	142
Distance to nearest arterial (m)	37 sites	4	8	38	69	180	235	315
Distance to nearest point source (m)	37 sites	875	1730	2908	4581	6208	7237	8011
Size of $PM_{2.5}$ point sources (tons/year)	7 sources	0.2	4	11	58	58	122	128
Elderly population density (persons/km^2)	148 tracts	3.9	7.1	14.1	29.1	45.2	62.7	141.2

bounded between 47.77° and 47.49° latitude, –122.42° and –122.25° longitude. GIS manipulations were conducted in ArcView version 3.2 (ESRI). A map of study monitoring sites was created by plotting subjects' addresses in geographic coordinates using the geocoding feature in ArcView (see Figure 1). Site elevations were obtained from the U.S. Geological Survey 10-m digital elevation models in the form of digital image files in 50-km square chunks.[29] The arterial roads data were available as shape files created by King County.[29]

The Seattle Transportation Department classifies streets through interdepartmental review, public involvement process, and city ordinance using a number of factors including average daily traffic (ADT), number of lanes, and street function. Most Seattle arterials have an ADT count of 3000 cars or greater and legal speeds of 30 mph or greater. However, some smaller streets with ADT between 1000 and 5000 and typical speed of 30 mph may also be designated as an arterial ("collector arterial") because they collect and distribute traffic from higher level arterials to access streets. Most neighborhood streets are defined as "commercial access streets" or "residential access streets," with ADT <3000 and typical speed of 25 mph or less. They were excluded from the statistical analysis. ADT count data were not available for all the streets in the study area.

Population density was calculated from the 1990 U.S. Census data for King County.[29] Census tract data consisted of tract numbers and areas. These data were merged with a Census attribute data file that included tract population over the age of 65. The elderly population density was calculated as the population over the age of 65 divided by the area (persons/km^2). High elderly population density was defined as a population density greater than the 75th percentile of Seattle Census tracts, or 45.2 persons/km^2. All plots and files were converted into Universal Transverse Mercator Zone 10 for North American Datum 1927 to correct for differences in geographic projection.

$PM_{2.5}$ point sources and locations were obtained from EPA's AIRData NET Facility Emissions Report.[30] The emissions report included geographic coordinates and amount of pollutants emitted in tons per year. Both the size of the sources and the distance between the sources and home sites were considered in the analysis. Distances from each monitoring site to the nearest point source, arterial road, and freeway were calculated using the "near features" extension in ArcView.

RESULTS

Data Description

The weather during the "heating" season in our first six monitoring sessions was mild, with an average temperature of 48 ± 5 °F in the fall of 1999 and 43 ± 3 °F in the winter of 2000 compared to the 10-year average fall and winter temperature in Seattle of 47 ± 7 and 44 ± 5 °F, respectively. However, it was slightly wetter than usual, with a fall and winter mean relative humidity of 87 and 83%, respectively. There was not much seasonal difference in $PM_{2.5}$ (Figure 2a): the mean $PM_{2.5}$ concentration was similar in the fall and winter (9.6 ± 5.4 vs. 9.4 ± 4.7 μg/m^3) and only slightly higher than in spring and summer (7.8 ± 4.4 and 8.2 ± 2.9 μg/m^3, respectively). The outdoor $PM_{2.5}$ measurements during the monitoring period were lognormally distributed, ranging between 1.0 and 26.0 μg/m^3 (arithmetic mean of 8.9 ± 4.5 μg/m^3), with a geometric mean (GM) of 7.6 μg/m^3 and geometric standard deviation (GSD) of 1.7 μg/m^3. For NO_2, the overall mean was 16.5 ± 6.1 ppb (N = 69). The mean of the two summer sessions (9.2 ± 4.1 ppb) was almost 2 times lower than those during the colder seasons (between 16.1 and 18.5 ppb) (Figure 2b).

The characteristics of the monitoring sites are summarized in Table 1. Elevation of the monitoring sites ranged between 27 and 142 m (mean = 92 m), distance to nearest arterial road was equal to or smaller than 325 m (mean = 105 m), and distance to nearest point source ranged between 0.5 and 8.5 km (mean = 4.0 km). Seven $PM_{2.5}$ point sources in Seattle were identified on the emission inventory, with

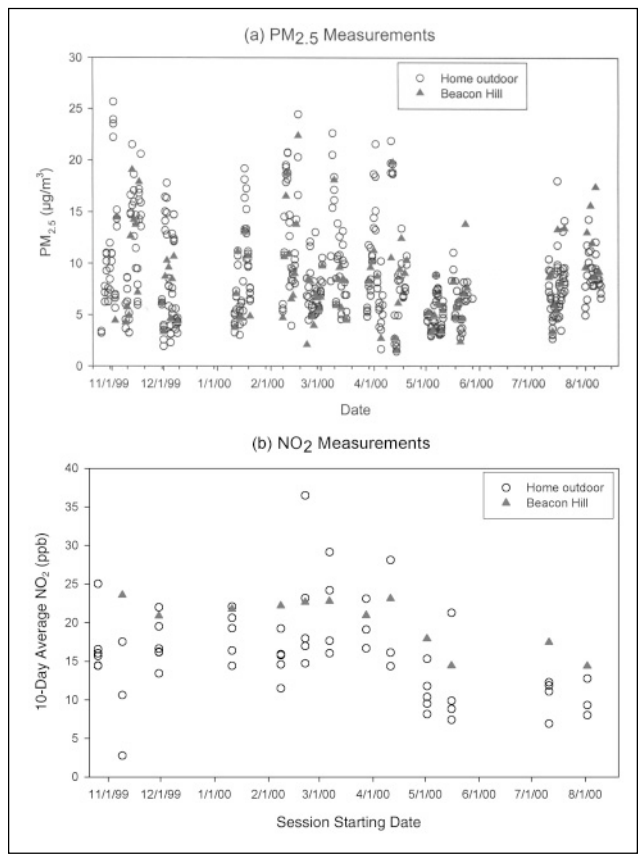

Figure 2. (a) Daily $PM_{2.5}$ and (b) 10-day average NO_2 measurements at home and at the Beacon Hill sites.

emission rates ranging between 0.17 and 198 tons/year. The elderly population density in King County Census tracts ranged between 0 and 439 persons/km^2 (median = 29 and mean = 40 persons/km^2).

Developing Siting Criteria

Results from the fixed-effect ANOVA model (model 1) are shown in Table 2. The model explained 92% of the variability in outdoor $PM_{2.5}$ measurements. The mean sum of squares (MSS) for the date effect was ~10 times that for the site effect, showing that day-to-day variability was much larger than differences between sites. The site MSS was more than twice the error MSS, showing that differences between sites, while modest, were potentially an important contribution to measurement error. The residuals, ε_{jik}, are normally distributed. The variance of the residuals, VAR(ε), calculated for each site, was lognormally distributed with a GM of 0.017 (or $10^{0.017} = 1$ μg/m^3) and a GSD of 2.40 (Figure 3a). The site effects, not far from a normal distribution, had a mean of 0 and an SD of 0.12 (or $10^{0.12} = 1.3$ μg/m^3) (Figure 3b).

Different spatial characteristics were examined to determine factors significantly affecting the magnitude of VAR(ε) and $SITE_i$ (Table 3). When VAR(ε) was grouped into three elevation categories, linear regression results showed that VAR(ε) decreases with elevation (model 2 in Table 3). Figure 4a shows that the variance and the mean of the VAR(ε) decrease with elevation. The highest mean (the dashed line) of VAR(ε) is in the 27- to 69-m group. Although the highest median (the solid line) of VAR(ε) is in the 70- to 100-m group, it is not significantly higher than the other two groups. When the elevations are grouped further into five representative groups in Figure 4b, it shows that 79- to 122-m groups have the smallest variance in VAR(ε). In summary, Figure 4a shows that the smallest variance in VAR(ε) lies somewhere between elevation 70 and 142 m, and Figure 4b shows that it is probably between 79 and 122 m. We tried several other groupings, and the range for the smallest variance in VAR(ε) is always bound around these two elevations (79 and 122 m).

In comparing VAR(ε) with distance to the nearest arterial road (model 3 in Table 3), there was no obvious relationship. However, a site may be very close to a road and show a systematically higher or lower site effect without necessarily a higher VAR(ε). In other words, sites located closer to roads have higher between-site variability (low accuracy) than those farther away, but any given site near the road can still have low within-site variability (good precision). As shown in Figure 5, the between-site variation decreases with distance from arterial roads. The figure indicates that it was more likely to have either larger negative or positive site effect values near arterials than away from arterials, with a greatly reduced variability in $SITE_i$ for sites located more than 100 m from the nearest arterial. A linear regression model (model 4 in Table 3) using the absolute values of $SITE_i$ to examine how the magnitude of the site effects varied away from roads resulted in an r of 0.29 ($p < 0.10$). The seemingly paradoxical finding of the negative site effect near arterials may be explained by the fact that all the monitoring sites were residential locations: an arterial within 20 m of a home is likely to be a small arterial with little traffic compared with one at 100 m from a home that may be much larger with more traffic. This reinforces the fact that our results are only applicable to monitoring sites in residential areas.

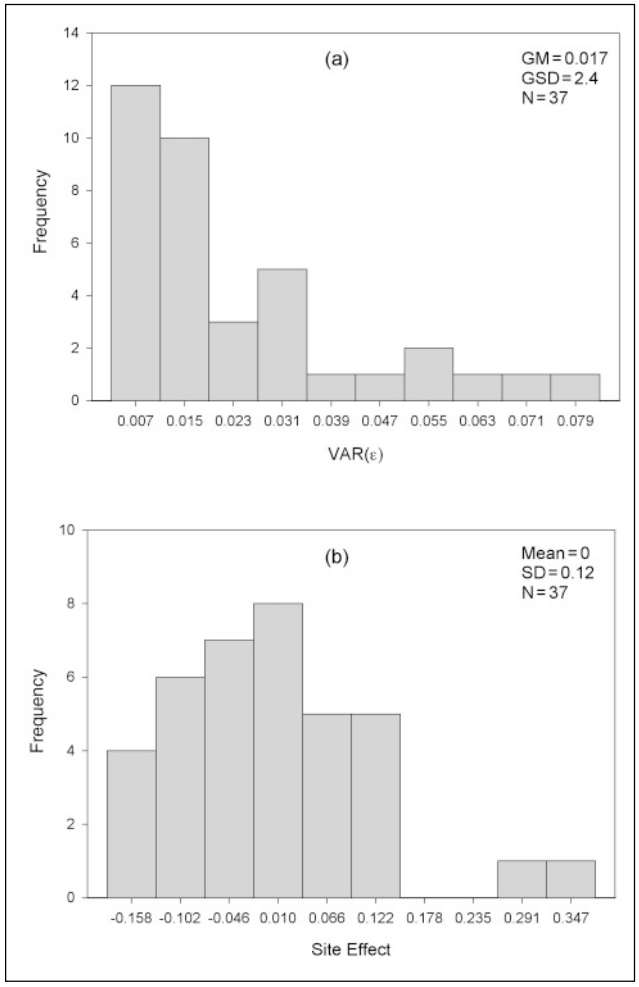

Figure 3. Histogram of (a) variance of model residual and (b) site effect.

Table 2. Results from the fixed-effect ANOVA (model 1 in test; $R^2 = 0.92$).

Parameter	DF	Type III SS[a]	MSS	*f* Value
Date factor	129	118.35	0.917	26.54[b]
Site factor	36	3.16	0.088	2.54[b]
Residual error	331	11.44	0.035	

[a]SS is sum of squares; [b]p value < 0.001.

Table 3. Univariate linear regression models examining factors affecting the variance of fixed-effect ANOVA model residual, VAR(ε), and the site effect.

Model	Dependent Variable	Independent Variable	Intercept	Coefficient	SE	*p* Value	*r*
2	VAR(ε)	Three elevation groups (m)	0.038	–0.007	0.004	0.077	0.29
3	VAR(ε)	Distance to arterial (km)	0.024	0.0003	0.027	0.991	0
4	Site effect[a]	Distance to arterial (km)	0.111	–0.167	0.096	0.088	0.29
5	VAR(ε)	Distance to source (km)	0.019	0.001	0.002	0.58	0.09

[a]Absolute value of site effect.

Neither $SITE_i$ nor VAR(ε) was related to distance from the nearest point source (model 5, Table 3). We further considered the effect of the emission rate of a source on the level and variability of the outdoor $PM_{2.5}$ measurements. Based on the Gaussian dispersion concept, concentration near the source is proportional to emission rate (Q) and inversely proportional to wind speed and distance squared (D^2). An analysis based on Q/D^2 gave no clear pattern. Because Seattle PM is known to be largely from vehicle exhaust and biomass (wood) burning, with <8% from industrial sources,[16,31] it is not surprising that these sources contribute little to the overall spatial distribution of PM. Thus, we do not include distance from these sources in the GIS analysis in the next section.

Ambient NO_2 levels are often correlated with ambient $PM_{2.5}$ levels because both pollutants are emitted primarily from the same sources, such as automobiles and industrial emissions.[32] In addition, the health effects of NO_2 often coincide with the health effects of $PM_{2.5}$.[33] Therefore, we also examined the spatial variability of NO_2 to enhance our understanding of the spatial pattern of $PM_{2.5}$. Using the standardized values for NO_2 measurements,[19] we found no relationship between NO_2 concentration and elevation. An earlier study by Chan and Hwang[24] in Taiwan also reported comparable NO_2 concentrations between 3 and 20 m of elevation. Both $PM_{2.5}$ and NO_2 had approximately the same relationship with distance from arterial roads, with the distance to arterial explaining only 9.2% of the variability in the NO_2 standardized values ($p < 0.05$). Analyses of NO_2 yielded similar siting criteria of 100–300 m from the nearest arterial.

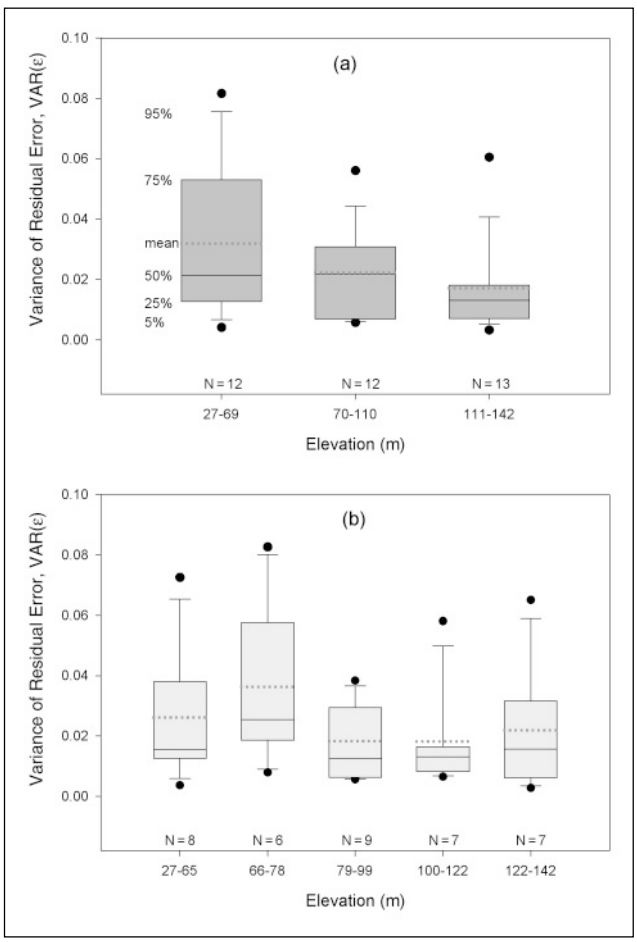

Figure 4. Variance of residual vs. elevation in (a) three elevation groups and (b) five elevation groups.

GIS Query Analysis

The optimum siting criteria were derived from the previous spatial characterization. Locations that are most representative of the average population $PM_{2.5}$ exposure [lowest values of $SITE_i$ and VAR(μ)] are those with the following characteristics:

(1) 79–122 m in elevation (see Figure 4b). Areas that match this criterion are shown in Figure 6a.

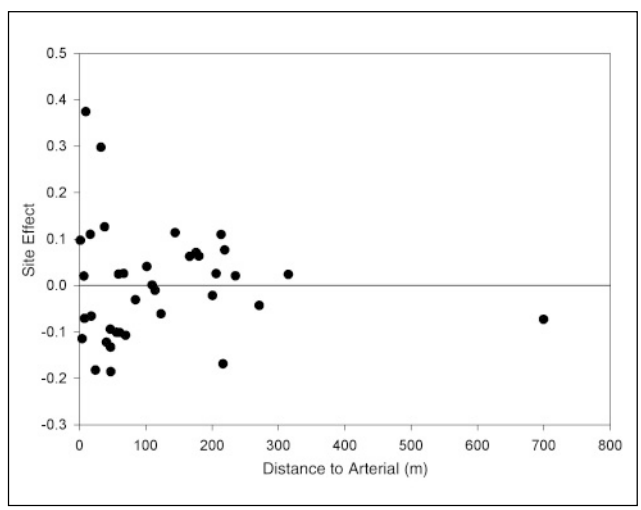

Figure 5. Site effect vs. distance to nearest arterial.

(2) 100–300 m from the nearest arterial (see Figure 5). Areas that match this criterion are shown in Figure 6b. The criterion for distance from arterials was truncated at 300 m because the distance between most of our study sites was smaller than 300 m from arterial roads. Additionally, very few areas in Seattle are more than 300 m from the nearest arterial road.

(3) Elderly population density greater than the 75th percentile population density of elderly for the Seattle metropolitan area, or 45.2 persons/km^2. Areas that match this criterion are shown in Figure 6c.

The overlap of these three maps is shown in Figure 7. These are deemed the best areas for monitoring the average population exposure to $PM_{2.5}$.

DISCUSSION

To determine candidate urban-scale sites in Seattle, we first constructed a fixed-effect ANOVA model (model 1) to take into account the spatial and temporal variability. It was possible that the spatial and temporal effects interact, meaning that the difference between any two sites varies systematically over time, perhaps due to different weather conditions. This can be assessed by comparing the residual mean-squared error in model 1 to the analytical precision of the $PM_{2.5}$ measurement. The square root of the residual mean square is 0.088 on a logarithmic scale, corresponding to multiplying or dividing by $10^{0.088} = 1.22$. Applying this to the mean $PM_{2.5}$ measurement of 9 μg/m^3 gives a typical error of 2 μg/m^3. Because this error includes the analytical precision, 1 μg/m^3, the remaining interaction is ~1 μg/m^3, about the same size as the measurement error. If the spatial pattern had changed markedly over time, the small number of sites (up to six) per day would have been a serious limitation of this study. In fact, the variability over time in the spatial pattern is relatively small—it is just one component of the residual mean square in our model—and so cannot be much larger than the precision of measurement. While the interaction is very small in our data set, the winter of 1999–2000 was mild, and important spatiotemporal interactions may still be present during wintertime inversions, which did not occur during our monitoring. This would explain the difference between our findings on interaction and those of Sheppard et al.[18] Future monitoring may be able to resolve this discrepancy.

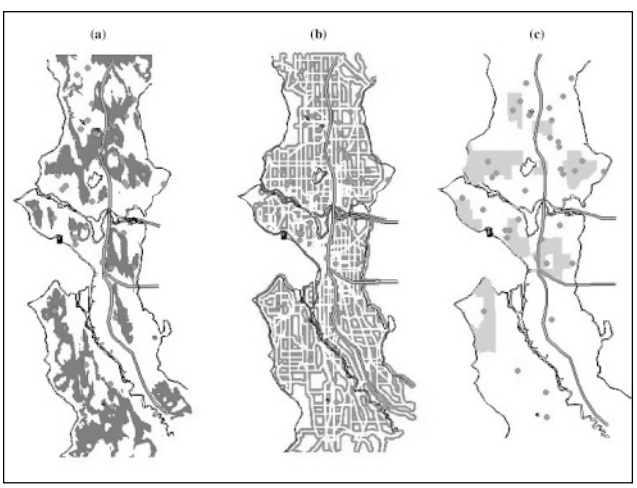

Figure 6. Areas fulfilled siting criteria: (a) elevation 79–122 m, (b) 100–300 m distance from arterial roads, and (c) elderly population density greater than 45.2 persons/km^2. Filled circles indicate study sites.

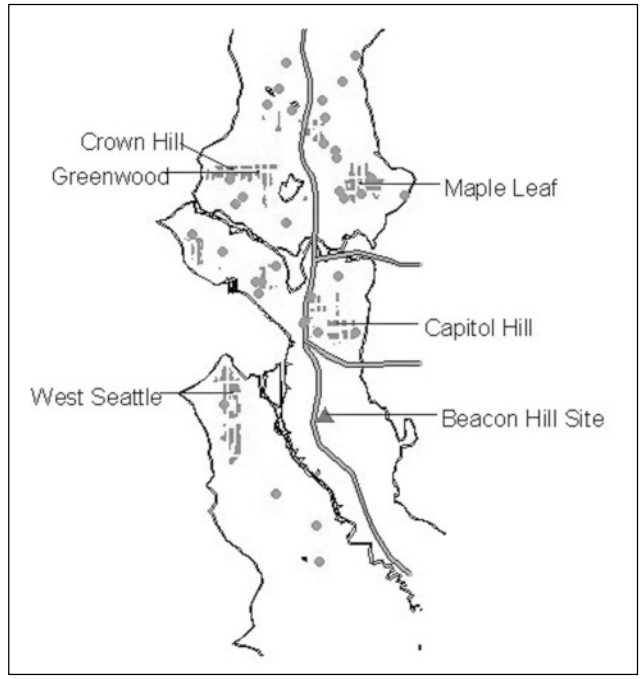

Figure 7. Potential monitoring sites are shown in shaded areas.

The relationship between VAR(ε) and elevation (see Figure 4) can be explained by meteorology. Sites at higher elevations with low VAR(ε) are not subject to trapping inversions and subsequent high ground-level concentrations. Our findings agree well with those of Larson et al.[17] and Sheppard et al.,[18] who reported that there was spatial variability of $PM_{2.5}$ with respect to topography in Seattle. Our analyses identified a range of elevations (79–122 m) that was most representative of daily variations in the average $PM_{2.5}$ outdoor concentration in this urban area.

Automobile exhaust is one of the major sources of $PM_{2.5}$ in Seattle. It was expected that $PM_{2.5}$ measurements at sites closer to arterials would be higher than those at sites farther away. EPA's siting criteria for PM_{10} samplers requested certain spacing from a roadway, depending on ADT, the scale of the monitoring site, the sampler height, effects of surrounding streets, wind speed, wind direction, and topography.[34] In general, a sampler located greater than 50 m from a busy paved roadway is considered outside the

road's immediate zone of influence for a rooftop sampler. This type of site is suitable for either middle- or neighborhood-scale samplers, depending on ADT. Urban-scale sites are to be at least 70 m away from a roadway. Hoek et al.[35] reported that there is a significant difference in the concentrations of black smoke within areas less than 100 m, compared with those greater than 100 m, from a major roadway in the Netherlands. This agrees well with our results in Seattle.

The site effects and VAR(ε) reported in this study were not very large. This may be due to the mild and wet winter in 1999–2000. The study period did not include a typical Seattle winter, which has many days of temperature inversions. Future analyses may include data from a more typical winter in Seattle. Additionally, many of our study sites were not in high-wood-smoke areas, where a large number of residents regularly use fireplaces or woodstoves in winter.

Wind speed and wind direction are important factors to consider in monitor siting.[14,25] Depending on whether a study site was upwind or downwind of sources of $PM_{2.5}$, the changes due to wind effects are reflected in negative and positive site effects, respectively. Sites that are located downwind of much of Seattle may experience greater variability in measurements or positive site effects compared with the average of Seattle. Likewise, sites that are upwind of much of Seattle may experience less variability in measurements or negative site effects compared with the average of Seattle. Ideally, a monitoring site would be located in an area in which wind direction does not greatly affect measurements; that is, the upwind sources in the prevalent directions are representative of the overall average of Seattle. However, the only wind data were collected at the Beacon Hill site (in addition to the Seattle-Tacoma International Airport). We did not include wind direction and speed in our spatial analysis because observations from the Beacon Hill site may not represent the actual variation by location, season, and time of day at our study sites. Future analyses may include wind direction, wind speed, relative humidity, solar radiation, mixing height, and ventilation index from a more elaborate monitoring scheme (instead of relying only on one site) and may incorporate them in the spatial analyses.

It is not certain what role natural sources of airborne PM (e.g., roadside dust, forest fires, marine aerosols, etc.), particulate condensation, or gas-to-particulate transformation play in the local, neighborhood-scale distribution of $PM_{2.5}$. Roadside dust may be an important contributor to the spatial pattern of $PM_{2.5}$, while forest fires (through long-range transport) and particulate condensation or transformation may be more important to the regional distribution. To answer this question, it would be necessary to perform source apportionment analysis at various sites and examine how the source contributions vary by location.

The recommended areas for $PM_{2.5}$ monitor siting in Seattle include the Crown Hill, Greenwood, Capitol Hill, Maple Leaf, and West Seattle neighborhoods (see Figure 7). Maple Leaf is considered the most suitable site because it has many potential monitoring locations, including a water reservoir. Other areas, such as West Seattle or Greenwood, also provide potential siting areas. However, West Seattle is near Puget Sound and generally upwind of downtown Seattle in the winter, which may not be ideal for urban-scale sampling. Furthermore, there were few monitoring sites in our study to provide data for verification in both the West Seattle and the Greenwood neighborhoods. The established urban-scale Beacon Hill monitoring site matches all the siting criteria identified by this study except for elderly population density. However, its VAR(ε), 0.04, was greater than the mean of the variances of all the sites, 0.017. This value corresponded to ~10% [$100\%*(10^{0.04}-1)$] variation or a difference of 1 $\mu g/m^3$ in $PM_{2.5}$. The site effect was –0.02, which corresponded to a systematic underestimation of only 5% [$100\%*(10^{-0.02}-1)$] less than the "true" mean of Seattle. These effects are small and alone would make Beacon Hill a suitable site. However, Beacon Hill is in an area where relatively few elderly people reside, while there are many other areas in Seattle where the elderly population density is greater.

CONCLUSIONS

The outdoor concentration of $PM_{2.5}$ varies spatially in the Seattle area. A fixed-effect ANOVA model was constructed to take into account the spatial effect and the $PM_{2.5}$ daily variation. Results from linear regression analyses of the site effect and VAR(ε) against spatial characteristics determined representative ranges of elevation, distance between site and nearest arteries, and distance between sites and nearest sources. Nitrogen dioxide, a co-pollutant of $PM_{2.5}$, also varied with distance from roadways.

The representative ranges of these spatial characteristics were used as overlaying criteria in GIS analysis for identifying the most representative areas in Seattle for $PM_{2.5}$ monitoring. After considering existing data, wind directions, and elderly population density, we determined the best monitoring areas were in the Maple Leaf neighborhood. The Washington Department of Ecology took this recommendation and set up a $PM_{2.5}$ and air toxics monitoring site in that neighborhood in January 2001.

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