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Seasonal spatial and temporal distribution of daytime inhalation-level particulate matter in Washington, DC

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Abstract

The spatial and temporal variations of inhalation-level airborne particulate matter in four Wards of Washington, DC were characterized during two six-week observation periods in the summer and fall of 2003. The average aerosol mass densities during the summer (19.3 μg m^{-3}) were roughly double the average values observed during the fall (10.0 μg m^{-3}). In general, a concentration gradient is observed from north to south across the District of Columbia with the gradient becoming more pronounced in the summer time. The fine aerosol size fractions centered at 0.15 and 0.30 microns dominated the mass distributions during both observation periods and the smallest size fraction showed a strong (0.79) correlation with wind speed during the summer IOP but negligible correlations with temperature and relative humidity. This may be an indication that non-local sources are contributing significantly to the fine aerosol fraction during this season.

***Key words**: Mass density; Number density; Particulate matter distribution; Washington, DC*

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INTRODUCTION

Elevated particulate matter concentrations, measured as PM_{10} or as $PM_{2.5}$, have long been implicated in contributing to respiratory problems, increased mortality (Schwartz et al., 2000; Ostro et al., 2007; Dominici et al. 2006; Franklin et al. 2007; Laden et al. 2006; Schwartz et al. 2002; Pope and Dockery, 2006), and their role in climate change (Girgždienė and Rameikytė, 2007). High $PM_{2.5}$ levels have been shown to reduce visibility which may affect transportation safety and aesthetics (Shendriker and Steinmetz, 2003). Urban particulate matter (PM) originates from a variety of stationary and mobile sources and may be directly emitted or formed in the atmosphere by transformation of gaseous emissions. Atmospheric chemical reactions also contribute to total $PM_{2.5}$ mass. Conversion of volatile organic carbon into secondary organic aerosols in urban air sheds is of increasing concern for regulating the fine aerosol fraction. Parkhurst et al. (1999) show correlations as high as 0.74 between daily average $PM_{2.5}$ concentrations and 1-h maximum ozone levels (DeGaetano and Doherty, 2004).

Other sources of atmospheric particulate matter point to the importance of human factors in determining the total loading of airborne aerosols in urban areas including combustion-generated particles, such as diesel, soot and fly ash; photochemically produced particles, such as those found in urban haze; wildfires, and particles from re-suspended dust (Girgždienė and Rameikytė, 2007). Automotive engine combustion accounts for a significant portion of urban $PM_{2.5}$ loads (Fraser et al., 2003).

The characterization of fine particles has become essential for informing policy on both air quality and climate (Dockery et al., 1993; IPCC, 2007). The large-scale spatial and temporal variation of urban particulate matter

concentration is influenced by a variety of anthropogenic and meteorological factors and has been studied extensively. Laakso et al. (2003) reported the maximum particle mass and nucleation mode concentrations both in urban and rural conditions in the spring and autumn seasons. Hien et al. (2002) indicated that the most important determinants $PM_{2.5}$ concentrations, for both the winter and summer monsoon periods, are wind speed and air temperature, while rainfall and relative humidity largely control the daily variations of $PM_{2.5-10}$, indicating the high abundance of soil dust in this fraction. Bogo et al. (2003) reported the correlation of $PM_{2.5}$ concentrations and traffic emission during the winter period and less correlation for the case of PM_{10}, with an important contribution of other sources such as suspended materials.

Washington, DC has sustained a reputation for poor air quality and at the same time a striking racial disparity in public health records. Curiously, there have been few published field studies focused on reporting the level of inhalation-level PM exposures within the District of Columbia. Washington, DC is also one of the most densely populated cities in the nation. Recent work from our group has reported on the relationships between ambient PM in the District of Columbia and increased health risk (Greene and Morris, 2006). This study showed non-negligible risks for Ward-specific pediatric asthma emergency room visits; and lifetime excess lung cancer risks, exceeding the 1×10^{-6} threshold for the measured levels of particulate matter and heavy metals (chromium and arsenic) on behalf of various subpopulations in four most heavily populated municipal districts (i.e. Wards) of DC. A similar study in New York City showed the association between elevated $PM_{2.5}$ concentration and lung cancer mortality. Lung cancer mortality was observed to increase by approximately 8% for each $10\ \mu g\ m^{-3}$ increase above the average ambient concentration $PM_{2.5}$ (Mehta et al, 2008).

In this paper, we report on analysis of geo-referenced microphysical measurements of inhalation-level PM collected over a five month period from the end of June through early December 2003. Hourly number density and mass density data were collected using a six-channel laser particle counter (LPC) and a six-stage quartz crystal microbalance (QCM) cascade impactor, respectively, at thirty-two sites in four heavily-populated Wards of Washington, DC. The objectives of this study were to determine the spatial-temporal distribution of ambient mass and number densities of inhalation-level PM within urban of Washington, DC. The observed patterns were interpreted with respect to local emission sources and meteorological conditions.

MATERIALS AND METHODS

Site selection

Site selection for the monitoring was based on several factors including the total area of each Ward, land use, proximity of the site to potential sources of particulate matter, security, and accessibility. The monitoring sites were selected to be nearly evenly distributed throughout each Ward, have sufficient security to ensure uninterrupted and untampered measurements during the observations conducted during each six-week period. Based on these criteria, a total of thirty-two monitoring sites were selected: seven in Wards one and seven, and nine in Wards four and five. At least two of the monitoring sites in each Ward were located within 1.61 km (upwind and downwind) of the strongest potential PM source identified in the Ward. This feature ensured a measure of control for understanding the influence of point source emissions. The total number of stationary sources of heavy metals, such as welding services, recycling sites, and gas stations in each Ward was identified and the monitoring sites were designated to provide near-uniform coverage of the Ward. Wards 1 and 7 each had a total of two significant stationary sources of particulate matter (Ward 1: one construction and auto body shop and the Howard University power plant. Ward 7: the Pepco-Benning power plant and a recycling site), Ward 4 had three (one gas station, one recycling site, and one waste reduction site), and Ward 5 had four stationary sources (one glass, one woodworking, and one welding facility, and two waste recycling sites).

Full coverage of the District of Columbia was not conducted due to the limitations of resources and transport times in between sites. Further, the motivation of this study was the characterization of the air quality in the areas distinguished by their health disparities. A map of the study area is shown in Figure 1.

Description of the selected wards

The District of Columbia is comprised of eight municipal districts, i.e. Wards (Figure 1), with a total area of 177.0 km^2 making it one of the most densely populated areas in the United States. There were 577,371 inhabitants based on the 2003 population estimate with a population density of 3,416/km^2 (http 1). In this study, we have investigated only four (Wards 1, 4, 5, and 7) of the eight Wards of Washington, DC. Wards 4, 5, and 7 are characterized as susceptible Wards due to their public health statistics. Of particular concern are their persistently high cancer death rates, Table 1. Wards 4, 5, and 7 are distinctive because of their status as high-risk areas for cancer and heart disease relative to the national averages.

A recent CDC report lists Washington, DC as one of the highest overall rate of cancer incidence (455.0) in the nation (CDC, 1999-2004; CDC 2007). Ward 4 is of particular concern due to its persistently high cancer deaths with an average of 247.8 followed by Wards 5 and 7 with average cancer deaths of 217.8 and 179.8, respectively.

Table 1. Cancer deaths during 1995-2000 for selected Wards in Washington, DC (D.C. 1995-2000)

Ward 1	Ward 4	Ward 5	Ward 7
139	265	238	213
137	250	224	197
112	218	191	155
136	265	209	179
134	232	210	183
127	257	235	152
130.8	**247.8**	**217.8**	**179.8**

Ward 1 is primarily residential, with more than 80% of its land devoted to housing units. Ward 4 has 87% of its land devoted to residential use, which is the highest percentage in the District of Columbia. It contains a stretch of the city's longest commercial corridor, Georgia Avenue, which essentially bisects Ward 4. Ward 5 is home to two major commuter arteries, New York Avenue and Rhode Island Avenue, which are gateways into the District to/from Maryland. This Ward has more industrial acreage than any other in the city. Ward 7 uses about 50% of its land as parkland (Green, 2006) and sits on the right bank of the Anacostia River. However, this Ward is home to the Pepco-Benning power plant, a primary source of heavy metals contributing approximately 13% of the annual total emissions based on the 1999 EPA $PM_{2.5}$ emissions data for the District of Columbia.

Measurements

This study was comprised of two intensive observation periods (IOPs) executed during summer and fall of 2003. The summer and fall IOPs were performed from June 23rd to August 8th of 2003 and October 20th to December 4th of 2003, respectively. Samples were collected from a height of two meters at each location using instrumentation mounted on the top of a mobile platform. The mobile platform rotated through the locations continuously throughout the course of each day. The vehicle was parked and turned off for an average of ten minutes prior to any measurement to avoid influence from its exhaust. In order to eliminate the "Sunday effect" samples were taken only on the weekdays (Monday through Thursday) between the hours of 10:00AM and 4:00PM. The "Sunday effect" is characterized by high late-week (such as weekends) pollution as opposed to the early week (Cerveny and Balling, 1998).

Aerosol number and mass densities measurements were obtained using a laser particle counter and a six-stage quartz crystal microbalance cascade impactor, respectively. The laser particle counter (CLiMET Instruments model 550, California, U.S.A.) units capable of measuring number densities in six size fractions: 0.3, 0.5, 1.0, 3.0, 5.0, and 10.0 microns with a precision of 15%. A six-stage PC-6S2 quartz crystal microbalance cascade impactor (California Measurements, Inc.) was used to measure aerodynamically fractionated mass densities at 0.15, 0.30, 0.60, 1.2, 2.5, and 5.0 microns. While this instrument is dually capable of collecting the size-fractionated samples for single particle analysis they were not periodically employed for this purpose during this study. The details of the chemical analyses are reported elsewhere (Green, 2006).

A portable weather tracker, Kestrel 4000, (Nielsen-Kellerman, PA, U.S.A.) was used to record significant in wind speed, temperature and relative humidity. In addition to measuring the current weather conditions, Kestrel 4000 tracks and stores up to 2000 sets of data, and data can be uploaded to a personal computer with the optional Kestrel Interface. The precision of the Kestral unit for wind speed, temperature, and relative humidity was ± 3%, ± 1ºC, and ± 3%, respectively, (http 2). The relative humidity and temperature measurements were corroborated with the measurements obtained from temperature and relative humidity sensors on the laser particle counter and compared to the National Weather Service values reported for Washington, DC. Outliers were removed if the values differed by more than 10%. These cases accounted for less than 20% of the total data collected.

Samples were collected at approximately two meters above the ground at each location in effort to characterize inhalation-level aerosols. All measurements were performed in triplicate to reduce biases due to local disturbances and to minimize random interferences. This resulted in maximum total measurement times of twenty minutes at each location. A transit time between locations was generally seven minutes or less.

The mass density data were collected during both the summer and fall IOPs, while the number density data was collected only during the fall IOP. All ambient aerosol data acquired from the IOPs were integrated into a Geographical Information System (GIS) database to enable spatial analysis of the aerosols across the region of interest. The National Oceanic and Atmospheric Administration (NOAA) Hybrid Single Particle Lagrangian Trajectory (HYSPLIT) back trajectory model (Draxler, R.R., 2001) was used to assist in the evaluation of potential influences from nonlocal sources and larger scale air mass trajectories during the summer and fall IOPs.

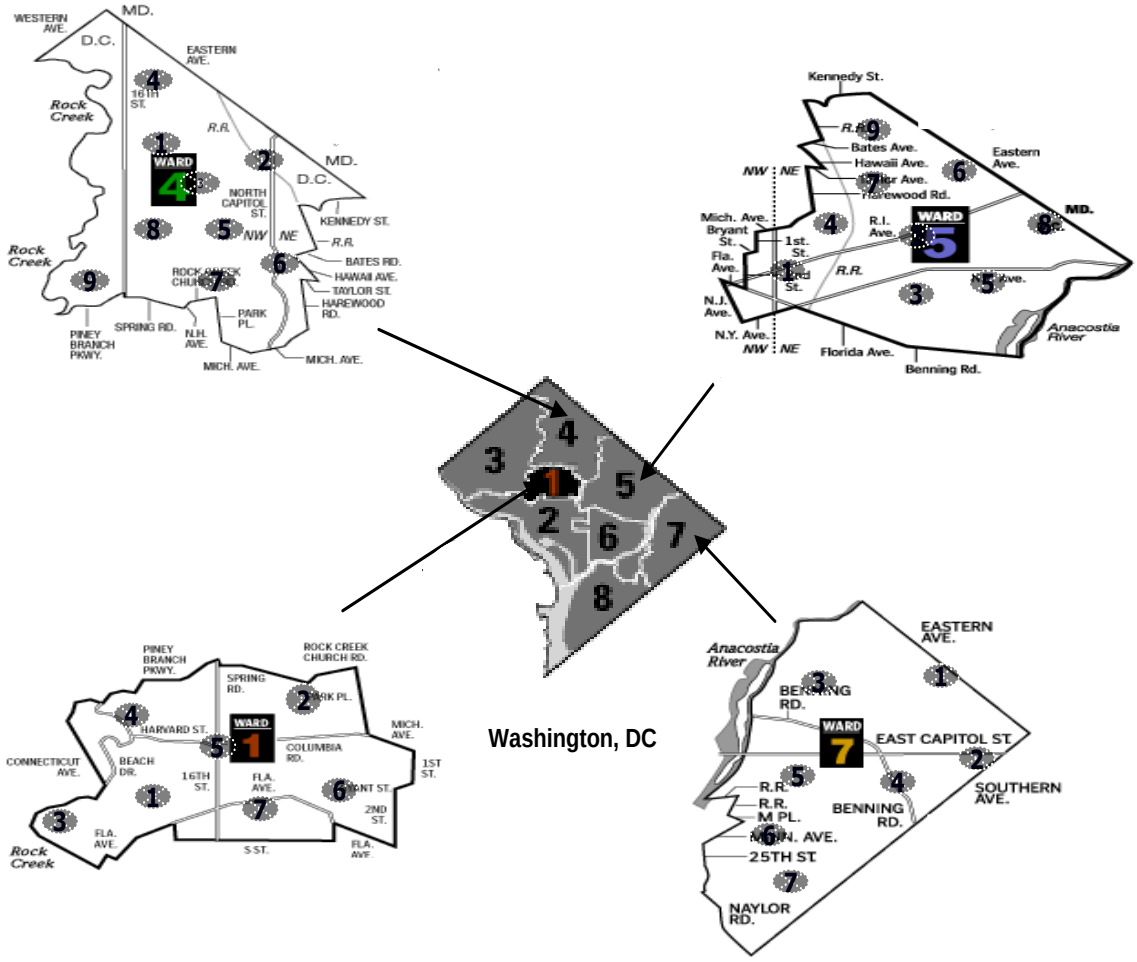

Figure 1. Schematic of the eight municipal districts within Washington, DC and the measurement locations within the four selected wards

Backround information on PM $_{2.5}$ mass distribution

Table 2 provides data for PM emissions based on EPA records from 1999-2004. The annual mean mass density of PM in Washington, DC exceeds that of the Baltimore but is lower than the values reported for New York and Philadelphia. The District's mean mass density over the six year period exceeds the annual National Ambient Air Quality Standard (NAAQS) of 15 μg m^{-3} by 11.3%. However, it decreased steadily and the annually averaged values for 2003 and 2004 are essentially at the threshold value.

It is interesting to note that Washington, DC, with a square area of less than 15% of that of New York City and a population of only 7% of the New York City metropolitan area, still shows an annual mean aerosol mass density that differs by only 6%. This suggests the existence of either significant local production that far exceeds EPA estimates from stationary sources (i.e. transportation) or a significant regional influence on the urban air shed.

Table 2. $PM_{2.5}$ (μg m^{-3}) Annual Mean Mass density (EPA, 1999-2004)

	DC	New York	Baltimore	Philadelphia
1999	18.1	19.3	17.4	15.5
2000	18.9	18.5	15.0	23.4
2001	16.9	18.0	16.1	17.0
2002	16.3	16.7	14.5	15.6
2003	14.9	18.6	15.0	16.1
2004	15.1	15.6	15.2	14.5
Average	16.70	17.78	15.53	17.02

RESULTS AND DISCUSSION

GIS Mapping: PM mass and number densities

The four Wards of study are generally arranged from north to southeast and align with major traffic corridors that connect northern Virginia to southern Maryland.

The GIS maps shown in Figure 2 represent the spatially-averaged PM2.5 mass densities measured for the summer and fall IOPs. The spatial averages are taken over the individual Wards in each map and radial distribution functions are used for the smoothing.

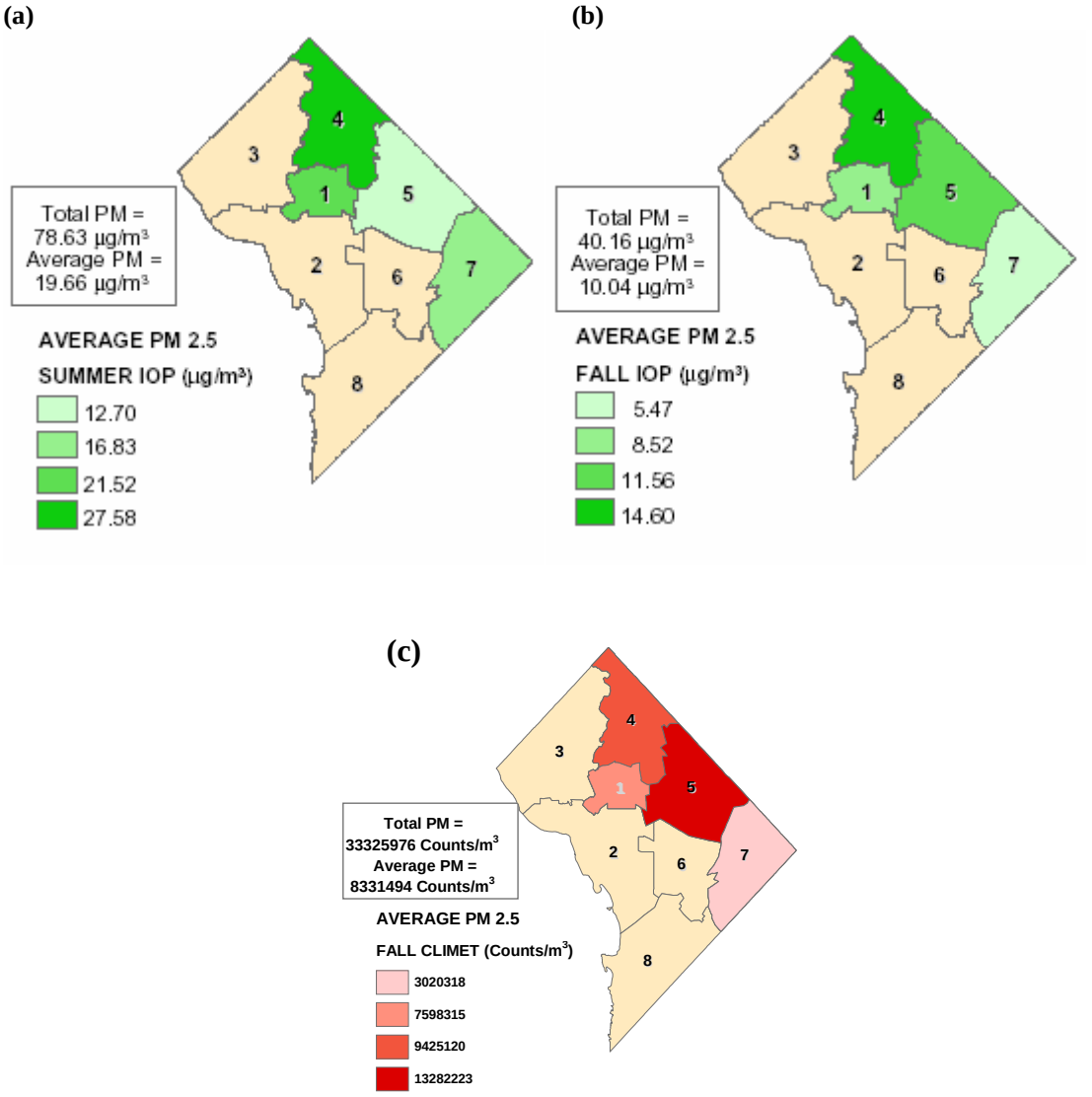

Figure 2. GIS Maps of the spatially-averaged $PM_{2.5}$ mass densities by Ward in (a) summer and (b) fall and (c) fall $PM_{2.5}$ spatially-averaged number density by Ward

The summer average mass densities ranged from 12.7 to 27.6 µg m^{-3} with an overall average concentration of 19.7 µg m^{-3}. The mean mass densities of the four Wards in the fall were about a factor of two less than the summer mass densities. While there was a general decrease observed in each of the Wards, the increase was dramatic in three of the Wards; 1 (-47%), 4 (-60%), and 7 (-67%) and essentially invariant in Ward 5 (-1%). Ward 4 (the northernmost sector) has the largest aerosol mass density in PM2.5 during both IOPs, with 27.6 µg m^{-3} in the summer and 14.6 µg m^{-3} during the fall. The southward pattern of decreasing aerosol mass density, ρ_i, is $\rho_4 > \rho_1 > \rho_5 > \rho_7$, with a 63% gradient from northernmost to southernmost sector in the summer and 39% gradient in the fall. Though we observe a 22% reduction in PM mass between the summer and fall IOPs in Ward 4, it remains the most polluted sector in both seasons. The second most-polluted Ward with respect to $PM_{2.5}$ shifted from Ward 1 to Ward 5. These observations are consistent with a primary influence of the traffic corridor.

The ordering of the spatially-averaged aerosol mass densities in Wards 1 and 5 is reversed in fall relative to the summer. This is due to the relative constancy of the average mass density measured in Ward 5 with respect to that measured in Ward 1. The Ward 5 average PM2.5

aerosol mass density for the summer (11.6 μg m^{-3}) only exceeds the fall average concentration by 1%. However, the Ward 1 mass density changes far more dramatically, a 60% reduction relative to the summer. The overall enhancement in mean PM mass concentration during the summer IOP is likely due to the lower wind speeds and elevated emissions due to higher energy consumption during the summer period. The average summer time $PM_{2.5}$ mass densities; ρ_1, ρ_4, and ρ_7 in Wards 1, 4, and 7 in the summer exceeded the fall values by a factor of 2.5, 2.0, and 3.0, respectively. We note that Ward 5 is one of the two Wards without a power plant and this may result in a more stable seasonal average mass density; ρ_5 (summer) ~ ρ_5 (fall), despite the greater power consumption during the summer. The distributions observed in this Ward should be more representative of micrometeorological factors than human factors.

Spatial mapping of the fall IOP number distribution for fine particulates is represented in the third frame shown in Figure 2. The GIS mapping of the number density displays a slightly different trend (with respect to Wards 4 and 5) in comparison to the mass distribution. Generally, fine aerosols contribute a smaller proportion of the total mass loading but a greater fraction of the total number of aerosols. The fractional amount of larger (super-micron) aerosols is greater in Ward 5 despite the larger total number density in Ward 4. Further, since accumulation mode usually occurs between 0.1 and 1 μm (Ruzer and Harley, 2005) there is characteristically a lag time between number and mass mode correlations. We observe a similar southward gradient in number densities except for significantly larger number density in Ward 5.

In the fall IOP, fine aerosol of diameters less than 0.5 microns dominate the distribution and are ranked in the order of Wards 1, 4, 5, and 7. Consistent with the fall IOP QCM data, Wards 4 and 5 exhibit the largest number distributions of particles in the 0.5 to 1.0 μm and 1.0 to 3.0 μm range, followed by Ward 1 and 7 respectively. A comparison of the $PM_{2.5}$ number distributions indicates that the ρ_5(N) exceeds ρ_4(N) by 30%.

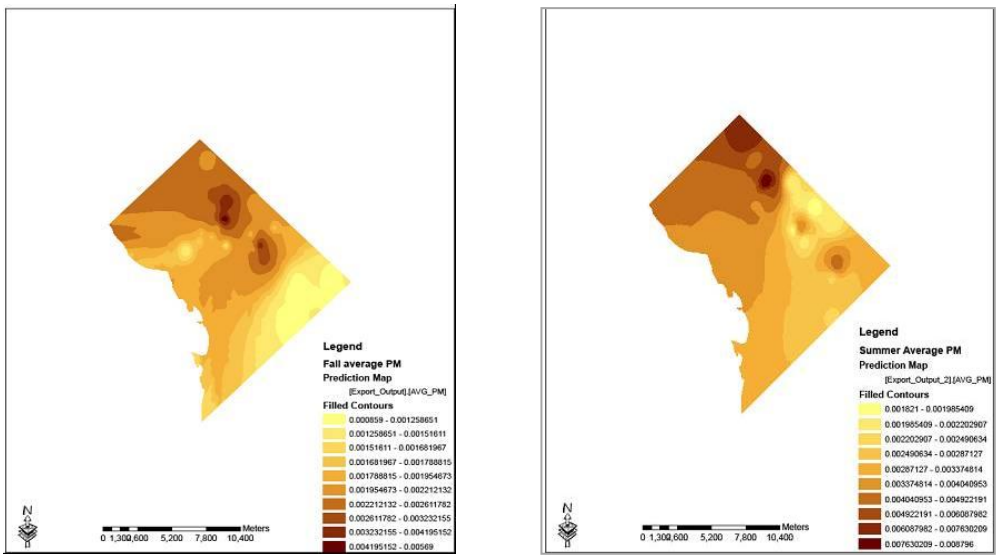

Figure 3. GIS images of the spatially-averaged summer and fall $PM_{2.5}$ concentrations over the entire District of Columbia

The IOP data were used to create GIS images of the spatially-averaged summer and fall $PM_{2.5}$ concentrations over the entire District of Columbia. These images (Figure 3) show behavior that is consistent with the Ward-averaged images (Figure 2). The decreasing trend from north to south is evident and is stronger in the summer than in the fall. These images more clearly indicate "hotspots" within the Wards, which are more pronounced in the fall than in the summer. The hotspots are located in the north central zones of the District but do not appear to be associated with the power plants in Ward 1 and 7. They are roughly coincident with two heavily trafficked intersections in the District.

Mass density distribution of PM

According to the reports available at the time of the study the Pepco-Benning emitted a total of 65,000 kg of PM into the DC environment, whereas all of the other major point sources in Ward 5 were estimated to have emitted a combined 28,000 kg (private communication). The Ward 5 aerosol mass concentrations observed during fall do not exceed the average values observed during the summer. However, the average mass density of Ward 5 is larger than the values observed in Wards 1 and 7 in the fall IOP. This differential is also reflected in the fall IOP number density data, in which Ward 5 shows a greater total number of particulate than Wards 1, 4, and 7.

Analysis of the fall IOP wind data revealed enhanced transport with northerly winds upwards of 5.7 m s^{-1} and south-westerly winds of 8.8 m s^{-1}.

The mean mass distributions of PM observed during the summer and fall IOPs are presented in Figure 4. The weekly averaged $PM_{2.5}$ mass distribution varies by 25% across all Wards, in the summer IOP and 30% during the fall IOP with respect to the mean IOP values. The mean mass density distribution, during the summer IOP, was more uniform compared to the fall IOP. Previous studies (Suh et al., 2000) have also indicated that ambient $PM_{2.5}$ and PM_{10} concentrations in the Eastern United States were relatively uniform across large (regions with populations of 3,000,000 or more) metropolitan areas, especially in the summer months.

In the summer IOP, a near monomodal distribution is observed in all Wards with the peak mass density ocurring in the 0.3 μm size fraction (~47%). As noted previously, the Ward 4 total mass density is about twice as large as the mass densities observed in the other three Wards.

The fall IOP mass distribution remains predominantly monomodal but the primary modes tend to shift to the smaller size fractions (0.15 μm). The distributions within each Ward compared across seasons are quite consistent as the observed relative mass fractions are essentially constant. In the fall IOP, the highest mass fraction was observed at the 0.15 μm size (~42%) for Wards 1, 4, and 7 – Ward 5 being the lone exception (Figure 4b). Summer values for both Wards 1 and 7 were larger than those observed in Wards 4 and 5 relative to the fall averages, in particular for the 0.15μm size fraction.

The fine aerosol mass fraction is generally greater in the summer than the fall, in particular, the 0.3 μm size fraction. We also note that the submicron distribution and fractional contribution to total mass are largely constant across the Wards in summer. In the fall IOP, the average total mass density for Wards 4 and 5 are somewhat similar in magnitude (16 and 12 μg m^{-3}, respectively) but these values are at least double the magnitudes of the average total mass densities of the remaining Wards, 1 and 7. The contributions of the submicron size fractions to the total mass distributions were 81% and 69% for summer and fall, respectively. The particulate distribution shifts dramatically from being dominated by the 0.3 μm mass fraction (41%) during the summer IOP to 0.15 micron mass fraction (33%) during the fall IOP.

Distribution of Total PM and mean $PM_{2.5}$

A comparison of the seasonal characteristics of the $PM_{2.5}$ spatial distributions are shown in Figure 5. The mean $PM_{2.5}$ concentrations ranged from 12.7 to 27.6 μg m^{-3} for the summer IOP and from 5.5 to 14.6 μg m^{-3} for the fall IOP. Consistent with the earlier discussion, the fall period exhibits lower $PM_{2.5}$ mean concentrations than the summer period. This can be explained by higher mobile emissions and higher power consumption during the summer season. These results are in agreement with those reported by other researchers (Chaulya, 2004; Crabbe et al., 2000). The highest mass concentration was observed in Ward 4 with 27.5 and 14.6 μg m^{-3} for the summer and fall IOPs, respectively. The summer IOP mean values in all Wards, except Ward 5, exceeded the NAAQS annual standard mean value for $PM_{2.5}$ (15 μg m^{-3}), indicating that the significant increase in seasonal fine particulate pollution is enough to cause an exceedance locally. In contrast to the summer IOP, the fall IOP mean values for $PM_{2.5}$ in all Wards were below the NAAQS threshold limits.

The higher mean concentration of $PM_{2.5}$ in Ward 7 during the summer IOP may likely be due to emission from the local power plant, a primary stationary source, in this Ward. Most power plants in the study area utilize less energy during the fall compared to that of the summer season (Melaku et al., 2008; private communication). The lower temperatures, an average of 12.4°C during the fall IOP compared to 30.2°C during the summer IOP, curb the use of air conditioners during the fall season, reflecting lower power demand and thus, lower emissions.

Influence of meteorological parameters on distribution of PM

Meteorological parameters are generally regarded as important factors for the aerosol transport and distribution (Marcazzan et al., 2002; Wise and Comrie, 2005; Aldrin and Haff, 2005). The average daily temperature ranged from 29°C (Ward 7) to 32°C (Ward 1) during the summer IOP and from 7°C (Ward 1) to 17°C (Ward 5) during the fall IOP. The average relative humidity ranged from 41% (Ward 1) to 49% (Ward 5) during the summer and over a broader range from 24% (Ward 1) to 53% (Ward 5) during the fall IOP. The average instantaneous wind speeds were typically light and ranged from 0.87 m s^{-1} (Ward 4) to 1.6 m s^{-1} (Ward 5) and 1.5 m s^{-1} (Ward 7) to 2.6 m s^{-1} (Ward 1) during the summer and fall periods, respectively.

Linear regressions of the mass densities for the individual size fractions, temperature, relative humidity, and measured wind speed were performed to determine Pearson correlation factors for these parameters. The results of these analyses are shown in Table 3. Generally speaking, the aggregate seasonal correlations between temperature and relative humidity and the size-resolved mass densities tended to be in phase with each other but out of phase with wind speed. The larger modes were found to have significant (> 0.5) correlations with wind speed and the smaller modes correlated better with temperature and relative humidity. The exceptions were the fall PM_5 fraction and the 0.3 and 0.15 size fractions. These findings agree with the data reported by Dainius et al., 2004. During the highest humidity episodes, the particle hygroscopic growth and condensation likely result in an increase of the coarse aerosol fraction; hence, fewer particles were collected as part of the fine PM fractions.

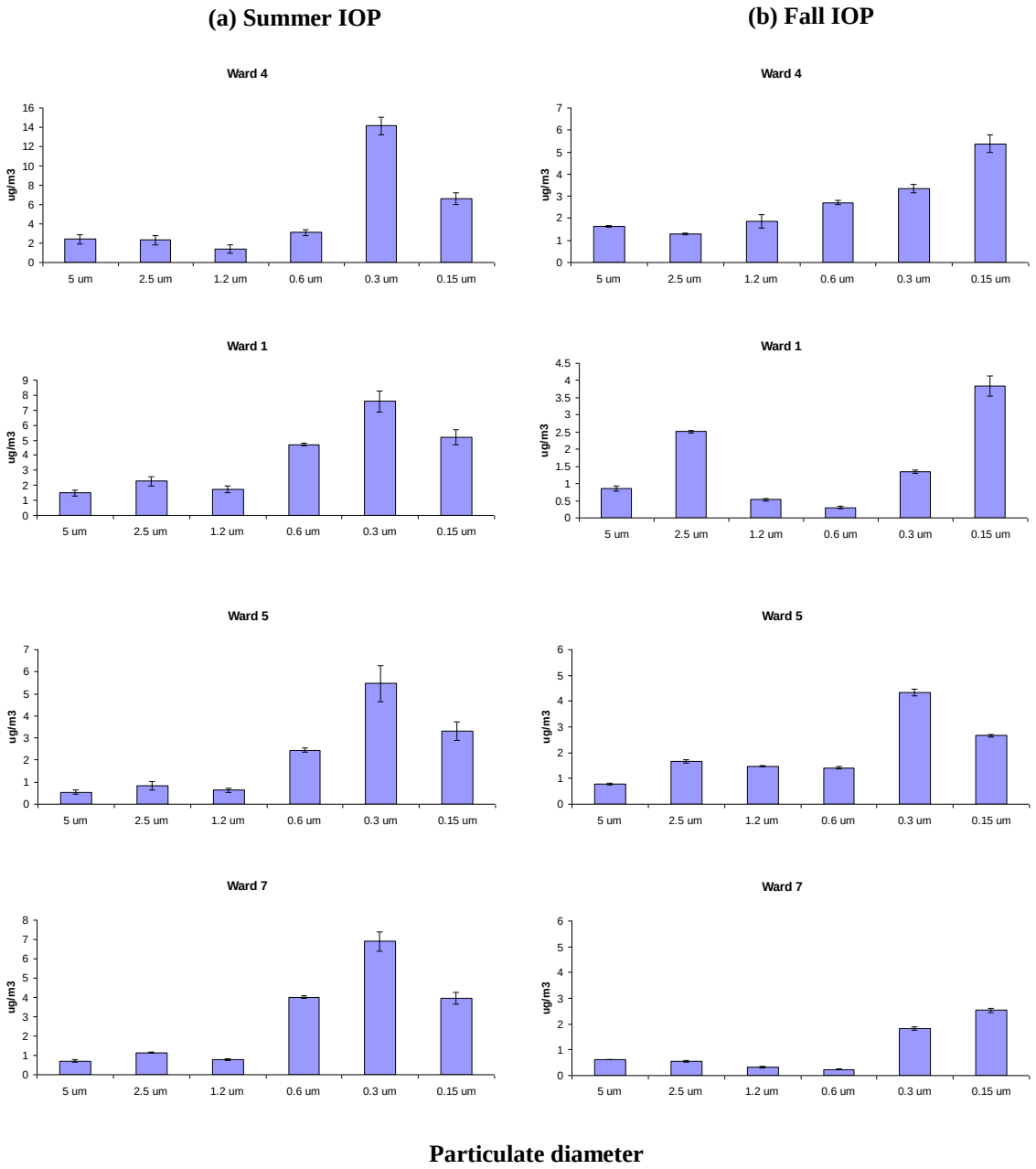

Figure 4. Seasonal mean mass distributions: (a) Summer IOP, (b) Fall IOP

The PM mass density distribution for $PM_{5.0}$, $PM_{2.5}$, $PM_{1.5}$, and $PM_{0.3}$ showed significant correlation ($R^2 \geq 0.5$) with lower wind speeds during the summer; and $PM_{2.5}$ ($R^2 = 0.84$) during the fall periods. The overall increase in concentration with decreasing wind speed, in this study, is in general agreement with other studies (Adams et al., 2001) but the association between the fine mode (0.3 and 0.15 μm) and wind speed is distinctive.

Analyses of the climatological wind patterns reveal a general trend of westerly transport through the Ohio River Valley region into the DC metropolitan area (http 3). HYSPLIT backward trajectories were computed for each day of the IOP at 1000-m above ground level, air mass transport from the southwest of this area 44% of the time during the summer and 21% of the time during the fall IOP. A compilation of the surface wind data collected from NOAA station #13743 located at the National Airport just to the south of Washington, DC reveals a slightly different picture. Figure 6 shows the surface (10-m) wind rose plots associated with the summer and fall IOP. Northerly winds appear to dominate the low-level flow with winds in the NNE sector accounting for over 60% of the flow during the summer.

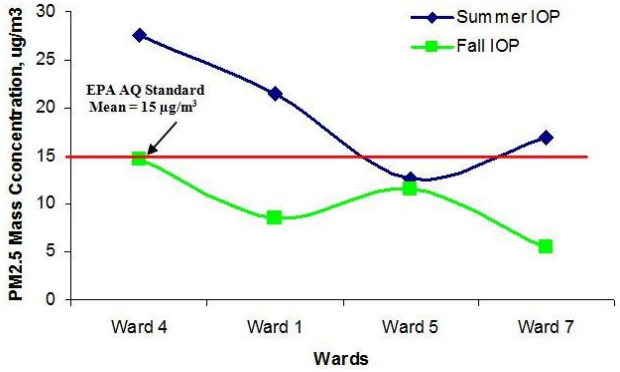

Figure 5. $PM_{2.5}$ means for summer and fall IOPs

Table 3. Correlation between PM mass density and meteorological parameters

Size Fraction	Correlation Coefficient, R^2					
	Temperature, °C		Relative Humidity, %		Wind Speed, m s^{-1}	
	Summer	Fall	Summer	Fall	Summer	Fall
PM_5	0.09	0.01	0.04	0.04	0.51	0.21
$PM_{2.5}$	0.10	0.39	0.07	0.17	0.57	0.84
$PM_{1.2}$	0.72	0.16	0.64	0.51	0.03	0.12
$PM_{0.6}$	0.47	0.10	0.77	0.38	0.01	0.08
$PM_{0.3}$	0.01	0.60	0.01	0.91	0.79	0.01
$PM_{0.15}$	0.11	0.18	0.07	0.01	0.53	0.35
Total PM	0.10	0.02	0.07	0.25	0.57	0.29

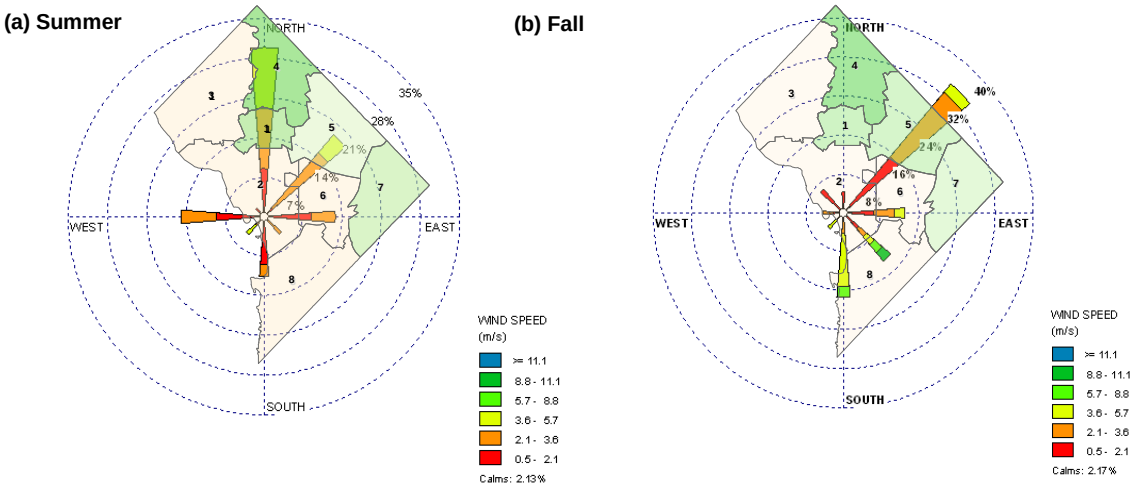

Figure 6. Wind Rose Plot and $PM_{2.5}$ contour map for the mass densities: (a) summer IOP, (b) fall IOP

These winds align well with the known traffic corridors from Maryland and could be responsible for enhancing fine aerosol distributions in the northern sectors. The surface wind distribution in the fall is still dominated by flow from the same sector but the balances of the winds are largely from the SSE sector, rather than evenly distributed. This tends to align with traffic corridors from Maryland during the summer IOP and from more coastal (and less industrial) regions during the fall IOP.

Urban topography is an additional factor contributing to the ambient distribution of particulates at inhalation-level. However, in the study design sampling sites were selected to minimize biases due to sharp gradients in altitude, shadowing from architectural structures, and other features that might lead to stagnation or restricted air flows.

CONCLUSIONS

The spatial and temporal distributions of particulate matter were characterized in Washington, DC Wards during the summer and the fall intensive observation periods in 2003. Higher PM mass densities were observed during the summer observation period compared to that of the fall. However, the PM mass concentration is much lower at the same sites during the fall due to lower energy consumption resulting in lower emission of PM into the atmosphere. The 0.15 µm and 0.3 µm size fractions were the major contributors to the total PM distributions during both IOPs. The mass fraction of particulate matter distribution for the 0.3 µm size fraction increased from the fall to the summer IOP by a factor of 1.2 (Ward 5) to 2.3 (Ward 1).

Elevated PM concentrations during the summer observation period were enhanced by low wind speeds and power consumption practices that could be attributed to emissions from the local sources. Additionally, the wind tended to originate from a direction that would inhibit the dispersion of the mobile source plume out of the District. We also observe that the spatial variability for the submicron fraction was greater within individual IOPs than across the summer and fall seasons. The levels of $PM_{2.5}$ in

Wards 1, 4, and 7 during the summer IOP exceeded the United States' annual average National Ambient Air Quality Standard threshold limit of 15 μg m^{-3}.

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