Dry deposition and resuspension of particle-associated metals near a freeway in Los Angeles

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Abstract

Dry atmospheric deposition represents a potentially large source of pollutant metal contamination in urban stormwater runoff, yet there is a limited amount of research on the relationship between atmospheric emissions and water quality problems in urban areas. In Los Angeles, significant quantities of toxic materials are released into the atmosphere every day and paved road dust represents the largest source of particle-associated metal emissions to the atmosphere. In order to better understand the role of roadways as a source of localized metal deposition, we characterized the horizontal dry deposition patterns of chromium, copper, lead, nickel and zinc upwind and at increasing distances downwind of the I-405 Freeway in coastal Los Angeles. Dry deposition fluxes and atmospheric concentrations of these metals were highest at the site closest to the freeway, and reduced to approximately urban background concentrations between 10 and 150 m downwind of the freeway. Compared with urban background, atmospheric particle size distributions indicated the freeway was a significant source of these metals on large particles >6 μm in diameter, which deposit close to their source and account for the increased dry deposition flux rates observed near the freeway. The spatial pattern of measured deposition flux was well predicted by a relatively simple line-source Gaussian plume model modified to include particle deposition and resuspension. The model results indicated dilution by vertical dispersion of the plume was the most important mechanism regulating downwind concentrations and deposition.

Introduction

Dry atmospheric deposition near urban centers, especially in semi-arid regions such as southern California, represents a potentially important non-point source of particle-associated metals to waterbodies (Baker *et al.* 1997, Lu *et al.* 2003). Atmospheric particulate matter may be directly deposited onto the surface of a waterbody or may reach the waterbody indirectly through deposition onto the land surface during dry periods, followed by subsequent wash-off during storm events. Atmospheric deposition may be particularly important in the Los Angeles Air Basin, since air quality in this region, with a population greater than 17 million, ranks among the worst in the United States (SCAQMD 2000). Emission inventories of the basin indicate significant quantities of toxic materials are regularly released into the atmosphere (SCAQMD, 2003), and the ultimate fate of the heavy metals in particular is unknown.

In urban areas, emissions from paved roadways are a major source of atmospheric particulate matter (Dunbar 1976; Cowherd *et al.* 1977; Reider 1983; Cowherd and Englehart 1984, 1985). Investigation of the most recent emission inventories for the Los Angeles region by Stolzenbach *et al.* (2003) found resuspended dust represents the largest source of particle-bound pollutant metals in the Los Angeles region, with paved road dust representing the most significant fraction. Paved road dust originates from pavement wear and decomposition, dustfall, litter, mud and dirt carryout, spills, biological debris, and erosion from adjacent areas (Cowherd and Englehart 1984, Chow *et al.* 1990, Chow and Watson 1992). In an urban setting, vehicle exhaust, as well as particulate from vehicle brake and tire wear, is a source of zinc and copper in paved road dust (Watson and Chow 2000, Councell *et al.* 2004).

In response to human health concerns, air quality standards have been set for particles less than 10 μm in diameter; consequently, most research on road dust emissions has focused on particles in this size fraction (Cowherd and Englehart 1984, Kantamaneni *et al.* 1996, Venkatram and Fitz 1998, Fitz 2001). However, the results from a number of studies indi-

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cate that nearly 50% of road dust total suspended particulate matter (TSP) is due to particles larger than 10 μm (Ahuja *et al.* 1989; Houck *et al.* 1989, 1990). Moreover, particles greater than 10 μm in diameter are largely responsible for metal deposition (Lin *et al.* 1993, 1994; Paode *et al.* 1998; Zufall *et al.* 1998). Because coarse particles settle faster due to their greater inertia and gravitational settling, deposition of these particles is likely to occur relatively close to the source (Sehmel 1973). Previous studies have documented a pattern of locally high atmospheric concentrations of particulate matter near roadways using tracers and downwind direct measurements of air and ground surface concentrations (Claiborn *et al.* 1995; Hitchins *et al.* 2000; Zhu *et al.* 2002a, b); however, few studies have focused on particle deposition gradients near roadways, especially for particles larger than 10 μm.

In addition to deposition as a source of metal loading into water bodies, re-entrainment of suspended atmospheric particles contributes to pollutant dispersion and impacts the subsequent mass loading into water bodies. The size of particles that can be easily resuspended ranges from 1 μm to 50 μm in diameter. Resuspended particles are estimated to travel globally; for example, Asian dust has been identified in Hawaii (Parrington *et al.*, 1983) and Sahara dust in the central United States (Perry *et al.* 1997). According to Sternbeck *et al.* (2002), measured metal concentrations in air are generally similar to the chemical profiles of crustal elements. This result indicates resuspension may control particle abundance and chemical composition.

The re-entrainment and suspension of particles in the atmosphere may occur through several natural and anthropogenic processes. Meteorological conditions during and after deposition (e.g., wind speed and rain intensity) and surface characteristics, such as surface roughness and surface moisture, are important influences on natural resuspension (Nicholson 1988). Anthropogenic activities such as vehicular activities, agricultural activities and various cleaning operations induce resuspension (Kashparov *et al.* 1994, Garger *et al.* 1998).

Because of the difficulty of measuring concentrations under different atmospheric stability and roadway configurations, predictions of concentration and particulate dispersion near roadways have been made using line-source Gaussian plume models (Chock 1978, Horst 1978, Sistla *et al.* 1979). The most widely used versions of these models are the modified HIWAY (Zimmerman and Thompson 1975), HIWAY-2 (Petersen 1980), GM (Chock 1978), and CALINE-3 (Benson 1979). However, these models have been used primarily to estimate vapor phase concentrations of constituents such as carbon monoxide, and do not include deposition or resuspension.

This study was designed to gain a better understanding of the role of major roadways, such as a freeway, as a significant source of localized metal deposition to urban surfaces and to understand the role of resuspension in the net deposition and dispersion of particulate matter near roadways. To accomplish this goal, the following objectives were defined: 1) to characterize the horizontal dry deposition gradient and atmospheric concentrations of five pollutant metals (chromium, copper, lead, nickel, and zinc) near a major freeway in Los Angeles; 2) to compare atmospheric particle size distributions of these metals near a freeway with urban background values; and 3) to compare measured horizontal deposition fluxes with the predictions of a Gaussian line-source dispersion model modified to include deposition and resuspension.

Methods

Sampling

The freeway site selected for this study was the I-405 freeway between Wilshire and Sunset Boulevards in West Los Angeles (Figure 1). This site was selected because: 1) this stretch of freeway runs perpendicular to the on-shore, southwest winds that dominate coastal Los Angeles during daytime in the spring; and 2) the I-405 freeway experiences heavy traffic volume, with an annual average daily traffic count of approximately 300,000 (CADOT, 2004). All sampling equipment was located along Constitution Boulevard, which runs perpendicular to the freeway.

Dry deposition flux of metals was measured simultaneously at 10 m (DW1), 150 m (DW2) and 450 m (DW3) downwind, and 150 m upwind (UP) of the I-405 Freeway over a three-week period in April and May, 2003. All deposition measurements were collected during daytime, high traffic hours (~8 a.m. to 5 p.m.) over a period of three to four days for a single sample, in order to obtain sufficient mass. Atmospheric concentrations of TSP were simultaneously collected at 2 downwind locations (DW1 and DW3) and analyzed for both particle mass and metal concentrations. Atmospheric concentrations of metals on four coarse particle size fractions were also measured at the downwind site

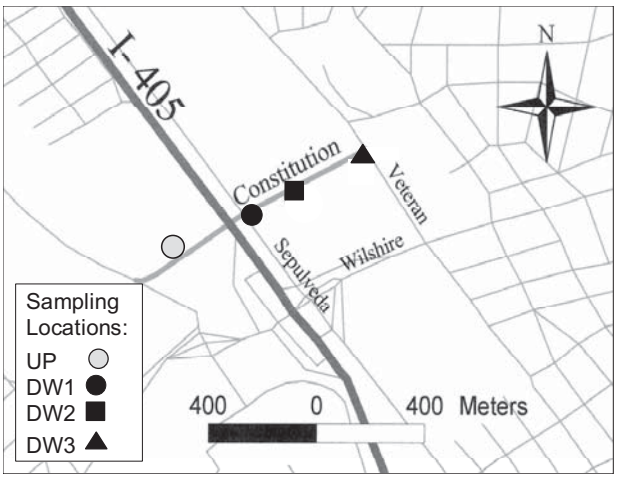

Figure 1. Map of the freeway site and sampling locations.

closest to the freeway (DW1) one day each week. All sampling took place during typical meteorological conditions for spring in Los Angeles during periods with no precipitation. Local wind data were utilized during sampling to confirm the upwind and downwind status of the sampling locations.

Four pieces of sampling equipment were used during this study. First, dry deposition flux measurements were made using a 33-cm diameter circular polyvinyl chloride (PCV) plate with a sharp edge (<10 degree angle), covered with a Mylar® sheet coated with a uniform 10-µm layer of Apiezon L grease. This surrogate surface has been used previously and is described in more detail elsewhere (Lim *et al.* 2006). The second piece of equipment was a filter-based sampling system attached to a vacuum pump used to collect TSP for measurement of atmospheric concentrations of metals. The open-faced inlet was loaded with a 37-mm, 2.0-µm pore Teflon® filter (Pall Life Science), and sampling was done at a flow rate of 10 L minute^{-1}. The third piece of sampling equipment was a portable meteorological station (PortLog, Rain Wise, Inc.), used to measure wind speed and direction, temperature, relative humidity and barometric pressure at the DW2 site. The fourth piece of sampling equipment was a Noll Rotary Impactor (NRI), used to collect size distributions of particulate matter for measurement of atmospheric metals concentrations. This instrument has been used previously to measure air concentrations on coarse particle size fractions and is described in detail in other studies (Mamane and Noll 1985, Noll *et al.* 1985, Lin *et al.* 1994). The NRI operates by simultaneously rotating four rectangular collector stages through the air at high velocity to collect particles directly from ambient air by impaction. Each collector stage is a different width to collect a different particle size fraction. The collector stages were mounted with Mylar strips, sized according to the width of the collector stage, which were coated with Apiezon L grease in the same manner as the Mylar for the deposition plates. The instrument was operated at 320 rpm, producing cut diameters of 6, 11, 20 and 29 µm for the four collector stages. To prevent overloading, the smallest collector stage (stage A) was changed every 2 hours, while the next largest collector stage (stage B) was changed every 4 hours. The two largest collector stages (stages C and D) were not changed during the ~8-hour collection period. The metal concentration of the particle size fraction smaller than 6 µm was obtained by subtracting the metal mass concentration collected on the NRI Stage with the cut diameter of 6 µm from the TSP metal mass concentration.

Prior to sampling, the Mylar was cut to the desired size, rigorously cleaned, prepared for deployment, and stored in airtight containers for transport to the field. After sampling, the Mylar was placed into clean plastic centrifuge tubes, rinsed three successive times with 15 ml of n-hexane to dissolve the Apiezon grease, and a final rinse of 5% Optima Grade nitric acid. The acid and hexane rinses were combined, the hexane was evaporated in a 50º C water bath, and the remaining acidified sample was then heated to 65º C under sonication for a minimum of 24 hours. TSP filters returned from the field did not require hexane rinses, but were directly digested using 5% Optima Grade nitric acid.

All acid-digested samples were analyzed for 26 metals per EPA Method 200.8 using inductively coupled plasma-mass spectroscopy (ICP-MS). Results reported in this study are for chromium, copper, lead, nickel and zinc, because these are the primary metals associated with water quality issues in southern California. Method detection limits were 0.5 ng for lead and 1 ng for all other metals, corresponding to a minimum detectable air concentration of 0.02 ng m^{-3} and a minimum detectable deposition flux of 0.004 mg m^{-2} day^{-1} for lead and 0.01 mg m^{-2} day^{-1} for all other metals. Laboratory blanks, analyzed with each batch of 15 samples, were consistently nondetectable. Field blanks were collected and analyzed along with each sampling event. Field blanks contained detectable levels of metals and all samples were corrected for their respective field blank. Field

duplicates indicated the relative percent difference (RPD) between side-by-side deposition plates, on average, was 31% chromium, 25% copper, 87% nickel, 24% lead, and 47% zinc. These were acceptable levels of precision for field duplicates because differences of less than a factor of two between fluxes measured during different sampling events were not considered significant.

Image analysis

An image processing program was used to count particles and to obtain the particle size and mass distributions of the particles deposited on the NRI stage A (d_p >6μm) from photographs of the Mylar strips taken with an optical microscope (LW Scientific) set at a magnification of 100x. For each image the distribution of the aerodynamic particle diameter d_p was determined using

$$d_p = \frac{1}{\overline{S}_v}\left(\frac{\rho_p}{\rho_o S_D}\right)^{\frac{1}{2}} d_{PA} \qquad (1)$$

Where d_{PA} is the equivalent projected area diameter measured by the image analysis, S_D is a dynamic shape factor set equal to 1.41 (Davies, 1979), ρ_p is the particle density assumed to be 1800 kg m^{-3}, ρ_0 is a unit particle density of 1000 kg m^{-3}, and $\overline{S}_v$ is the volume averaged shape factor set equal to 1.61 for urban sites (Lin *et al.* 1994, Tai *et al.* 1999). The aerodynamic diameter (d_p) and assumed particle density (ρ_p) were then used to calculate total particle volume and mass. The atmospheric concentration was obtained using the known NRI rotation speed and empirically determined collection efficiencies (Noll *et al.* 1985).

Modeling

The model used in this study was based on the ground-level line-source Gaussian plume model, formulated to consider metal deposition and resuspension (Horst 1978). Assumptions for the model were constant emission rate, constant wind speed both in time and space, neutral stability (stability D), and a flat and unobstructed ground surface (Masters 1998). The emissions from the freeway were assumed to form a single, continuously emitting, infinite line source with metal mass flow per unit length q_o (mg m^{-1} second^{-1}).

With these assumptions, the metal concentration in the air down wind at distance x and elevation z from the line source can be described by the following:

$$C(x,z) = \frac{1}{u\sqrt{\pi/2}}\left\{\frac{q_o}{\sigma_z(x)}\exp\left(\frac{-z^2}{2\sigma_z^2(x)}\right) + \int_0^x \frac{m(x)}{\sigma_z(x-\xi)}\exp\left(\frac{-z^2}{2\sigma_z^2(x-\xi)}\right)d\xi\right\} \qquad (2)$$

where u is the wind speed; $m(x)$ is the net metal mass flow per unit ground surface area (mg m^{-2} second^{-1}) at a distance x resulting from deposition and resuspension; $\sigma_z(x) = c(x)^d + f$ is the vertical standard deviation of the plume at a distance x from the plume source, where c, d, and f are constants that are a function of the stability classification (Masters 1998).

The net metal mass flow to the atmosphere per unit ground surface area resulting from deposition and resuspension is computed by:

$$m(x) = \Lambda G(x) - V_d C(x,0) \qquad (3)$$

where $G(x)$ mg m^{-2} is the surface metal mass per unit area, Λ second^{-1} is a specified resuspension rate, $C(x, 0)$ is the ground level metal concentration in the air, and V_d is a specified deposition velocity. The change in surface contamination with time is then given by:

$$\frac{dG(x)}{dt} = -m(x) = V_d C(x,0) - \Lambda G(x) \quad G(x) = 0 \text{ at } t = 0 \qquad (4)$$

The build-up of $G(x)$ is the only time dependent process in the model, although changes in $G(x)$ drive changes in all other variables. A steady state condition where $m(x) = 0$ and $\Lambda G(x) \approx V_d C(x,0)$ is reached in a time of about $1/\Lambda$. The steady state atmospheric metal concentration is given by the first term in equation 2 and is independent of the deposition velocity V_d. and the resuspension rate Λ, but the deposition flux is $V_d C(x,0)$.

Model calculations used a wind speed u = 2 m second^{-1}, which was the average value during the observation period, and a deposition velocity V_d = 0.01 m second^{-1}, which was the mean of the flux-averaged deposition velocities calculated for each metal by dividing the measured deposition flux by the air concentration measured by the TSP sampler at both DW1 and DW3 (Table 1).

Results

There was little day-to-day variability in the meteorological data measured during the sampling at the freeway site (Table 2), and even hour-to-hour variability during the 8 a.m. to 5 p.m. sampling period within the same day was low. Because meteorological conditions were stable throughout the study period, this

Table 1. Measured flux-averaged deposition velocities for different metals.

Metal	*N*	Deposition Velocity (cm second^{-1})
Chromium	6	0.56 ± 0.1
Copper	6	1.4 ± 2
Lead	6	1.3 ± 0.5
Nickel	6	0.26 ± 0.2
Zinc	6	1.1 ± 0.4

study did not attempt to correlate these data with weekly concentration or deposition flux measurements. Wind direction remained predominately from the southwest on all sampling days, as expected for springtime in Los Angeles, maintaining the desired upwind and downwind locations of the sampling sites.

The highest measured deposition fluxes of all five metals were observed at the downwind site closest to the freeway (DW1; Table 3). For copper, lead and zinc, ANOVA indicated that variation between sites was significant ($p < 0.002$). This variation was due entirely to the higher fluxes observed at DW1, based on the Tukey test for pairwise multiple comparisons ($p < 0.003$). Mean fluxes at DW1 were higher than UP and DW3 by factors of two to five, depending on the metal. Mean fluxes at DW1 were higher than DW2 by factors of two to three for all five metals. In contrast, differences in fluxes measured at UP (considered to represent "urban background"), DW2, and DW3 were not significant for all five metals (ANOVA, $p > 0.05$).

TSP metal concentrations were higher at the downwind site closest to the freeway (Figure 2a), although the differences between DW1 and DW3 were only significant for lead and zinc using the paired samples t-test ($p < 0.02$). TSP particle mass concentrations were also consistently higher at the downwind site closest to the freeway (DW1) compared with the furthest site (DW3; Figure 2b), however differences between the sites were not statistically significant. Again, the sample size (n = 3) may have limited this study's ability to detect small differences between locations.

The mean particle size distribution differed between the site nearest the freeway and urban background sites located away from major freeways in Los Angeles, as measured by Lim *et al.* (2006; Figure 3). The primary difference was that more metal mass fraction was observed in the particle fraction >6 µm, especially for copper, lead, and zinc, at the freeway site,. At the urban background site, the majority of the mass fraction (approximately 75%) for all metals was observed in the particle size fraction <6 µm.

Image analysis of stage A NRI strips indicated the particle mass in the fraction >29 µm at the freeway site was relatively constant over time (Figure 4). In contrast, the size fraction >29 µm was absent during the morning period (7:00 a.m. to 11:00 a.m.) at the urban background site. The total particle mass concentration associated with particles >6 µm at the freeway site ranged between 23 and 31 µg m^{-3}, compared with 10 to 16 µg m^{-3} at the urban background

Table 2. Summary of mean meteorological data measured during sampling at the freeway site. Ten minute data were recorded. The 8-hour means are presented here, except as noted.

Week	Date	Temperature (ºC)	Relative Humidity (%)	Wind Speed (m second^{-1})		Wind Direction From:
				Mean 8-hour	Range of 10-minute Max	
1	13-Apr-04	19 ± 1	63 ± 4	2.1 ± 0.6	0.4 - 7.2	Southwest
	14-Apr-04	19 ± 1	62 ± 4	2.1 ± 0.7	2.2 - 7.2	Southwest
	15-Apr-04	20 ± 1	62 ± 2	2.1 ± 0.7	2.7 - 6.7	Southwest
	16-Apr-04	18 ± 1	61 ± 5	2.1 ± 0.6	3.1 - 6.7	Southwest
2	19-Apr-04	18 ± 1	54 ± 4	2.4 ± 0.8	2.7 - 8.0	Southwest
	20-Apr-04	19 ± 1	60 ± 5	2.4 ± 0.7	2.7 - 7.2	Southwest
	21-Apr-04	19 ± 1	64 ± 3	2.4 ± 0.6	3.1 - 7.2	Southwest
3	28-Apr-04	21 ± 2	67 ± 7	2.2 ± 0.5	2.7 - 7.2	Southwest
	29-Apr-04	20 ± 1	55 ± 4	2.4 ± 0.5	3.1 - 7.2	Southwest
	30-Apr-04	19 ± 1	67 ± 5	2.7 ± 0.6	3.1 - 7.2	Southwest
	1-May-04	26 ± 1	38 ± 5	2.5 ± 0.4	4.0 - 6.3	Southwest

Table 3. Mean dry deposition flux ± standard deviation (μgm^{-2} day^{-1}) of trace metals measured at varying distances from the I-405 Freeway. N = 3 for all metals.

Location	Chromium	Copper	Nickel	Lead	Zinc
10 m Downwind (DW1)	4.3 ± 0.3	48 ± 8	3.1 ± 0.6	24 ± 3	140 ± 33
150 m Downwind (DW2)	2.4 ± 1.9	18 ± 7	1.0 ± 1.3	11 ± 4	45 ± 23
450 m Downwind (DW3)	2.5 ± 1.1	14 ± 3	1.2 ± 1.1	7.3 ± 1.8	38 ± 1
150 m Upwind (UP)	2.2 ± 0.5	11 ± 5	1.5 ± 0.9	7.9 ± 1.3	37 ± 10

site. The time variation of the total particulate mass concentration at the freeway site for particles >6 μm did not demonstrate any significant difference between times (ANOVA p >0.5).

Good agreement between the steady state model calculation and measured deposition fluxes was obtained for chromium, copper, lead, and zinc, particularly within 200 m from the source (Figure 5). The data for nickel diverged from the model results, which was not surprising because of the expected absence of a nickel source at the freeway. Both the model and the measurement data indicated deposition and air concentrations return to background between 10 m and 150 m from the source; a result consistent with the findings of Zhu *et al.* (2002b) for ultra fine particles. Model calculations with $\Lambda = 0$ were nearly identical to the steady state solution, indicating that on the scale of this experiment, atmospheric dispersion was the major mechanism determining the spatial distribution of atmospheric metal concentration and metal deposition flux.

Discussion

The findings of this study indicate that metal deposition rates, especially for copper, lead and zinc, increase in the immediate vicinity of a large freeway and quickly reduce to urban background deposition rates between 10 m and 150 m down-

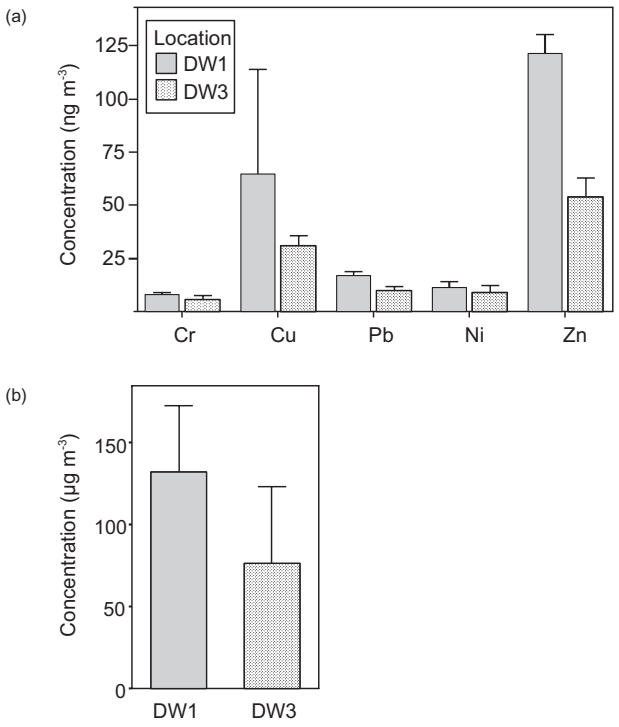

Figure 2. Mean atmospheric TSP concentrations 10 m downwind (DW1) and 450 m downwind (DW3) from the I-405 Freeway for metals (a) and particulate matter mass (b). Error bars represent the standard deviation for the mean.

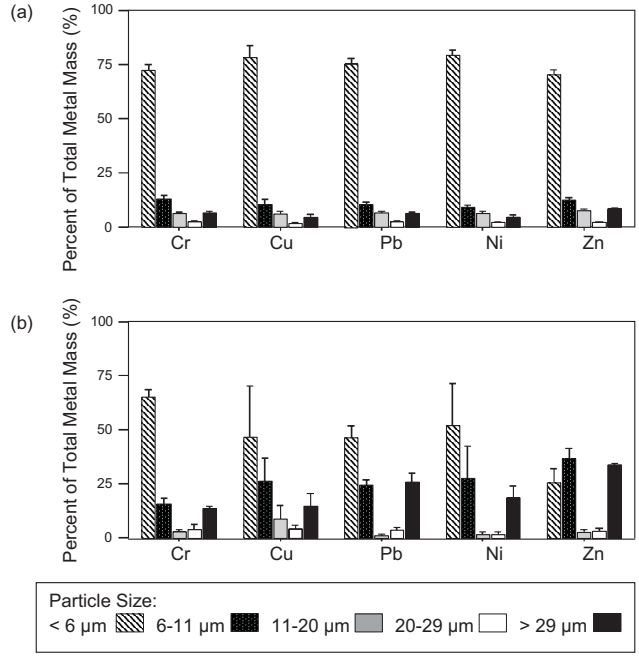

Figure 3. Mean distributions of metal concentrations on five particle size fractions as a percent of the total metal mass measured at urban background sites as measured by Lim et al. 2006 (a) and 10 m downwind of the I-405 Freeway (b). Error bars are the standard error of the mean.

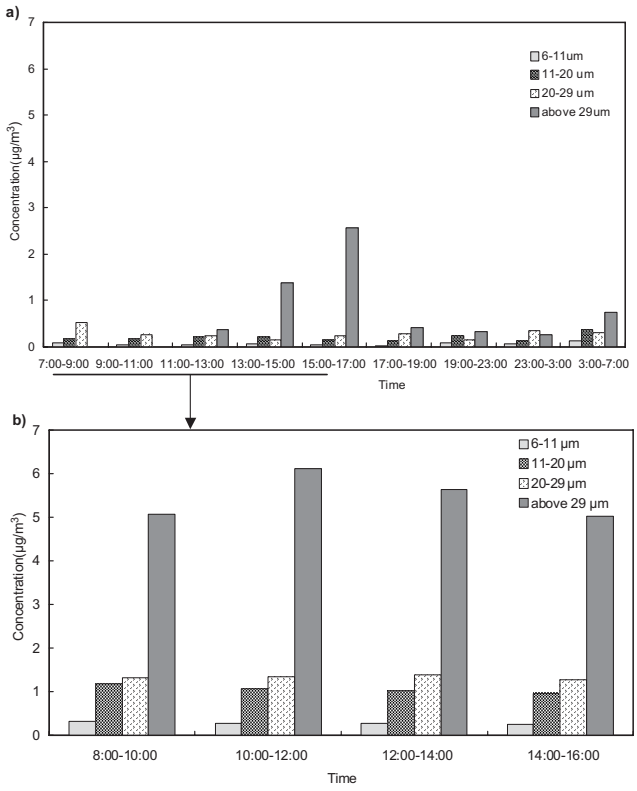

Figure 4. Diurnal pattern of particulate mass concentration measured using image analysis: urban background site as measured by Lim et al. 2006 (a); freeway site, this study (b).

wind of the freeway. These results are similar to the observations of Zhu *et al.* (2002b) for ultra fine particle concentrations (d_p <0.1 µm) measured downwind of the same freeway. In that study, high concentrations near the freeway reduced to urban background within 300 m. This present study's results also suggest that sources of deposited copper, lead, and zinc, which had higher fluxes near the freeway, may be different from chromium and nickel sources, which were not increased near the freeway. The freeway likely represents a source of large particles containing copper, lead and zinc due to resuspension of road dust as vehicles travel on the freeway at high velocities (Sehmel 1973, Nicholson *et al.* 1989) and particulate from vehicular tire and brake wear (Chow *et al.* 1990, Chow and Watson 1992, Councell *et al.* 2004). The small sample size in this study was an important limitation of these data. However, the general trend of decreasing deposition flux with distance from the freeway was consistently observed during the study period.

This study's comparison of trace metal concentrations on different particle size fractions observed near the freeway and at urban background sites indicates that higher concentrations near the freeway are primarily due to increased mass in the particle size fraction >6 µm. One limitation of this study is the absence of simultaneous measurements of the particle size distribution near the freeway and at urban background. However, because both lower TSP metal concentrations at the DW3 site and reduced metal mass fraction due to the largest particles at urban background locations away from the freeway were observed, a hypothesis that the lower metal concentrations were due to removal of the largest particles by deposition near their source was formed. Thus, it was concluded that variance in the particle size distributions of metal mass at the freeway site compared with urban background sites likely resulted from the freeway acting as an emission source for particles >6µm. As coarse particles are more readily removed from the air by deposition close to the source, lower concentrations of these particles and lower percentages of the total mass as distance from the emission source increases were expected. Smaller particles (<6 µm), which are slower to deposit, remain suspended and contribute a higher proportion of the total mass away from local sources. This explains why both atmospheric concentrations of metals on the largest particles and metal deposition fluxes were reduced at sites further removed from a freeway. Corroborating results from this study's deposition plate data indicate that the majority of this removal occurs very near to the source (e.g., between 10 and 150 m), resulting in deposition fluxes and concentrations downwind of the freeway

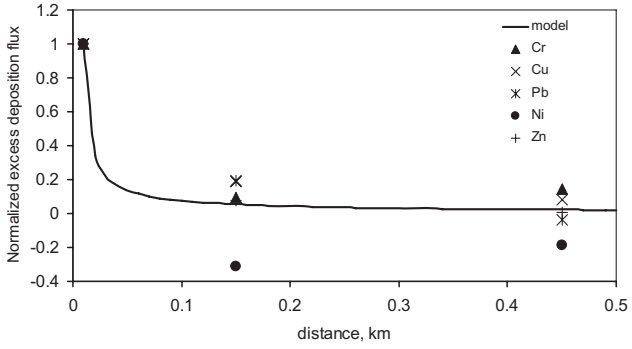

Figure 5. Comparison of computed and measured normalized excess deposition flux where the solid line is the steady state model prediction using a wind speed u = 2 m sec^{-1} and a deposition velocity V_d = 0.01 m sec^{-1}.

comparable to urban background at distances greater than 10 m and within 150 m.

In addition to higher metal fluxes and concentrations near the freeway, this study's image analysis indicated the site nearest the freeway had higher particle mass concentrations throughout the day compared with urban background. In contrast, increased particulate matter concentrations at urban background sites occurred only later in the day, perhaps as a cumulative result of resuspension from adjacent street traffic. The lack of difference between total particulate mass concentrations at the freeway site during different time intervals for particles >6 μm was likely because of the relatively constant traffic flow on this particular freeway over the daytime sampling period.

The modified Gaussian plume model showed relatively good agreement with deposition measurements. In this case, dispersion was the most significant process controlling the spatial variation of concentration and deposition. Resuspension rates Λ reported in the literature for surfaces of all types vary from 10^{-13} second^{-1} to 10^{-6} second^{-1} (Nicholson 1988), and for asphalt surfaces from 5×10^{-9} second^{-1} to 6×10^{-8} second^{-1} (Sehmel 1980). For the present study, values of Λ between 10^{-9} second^{-1} and 10^{-6} second^{-1} reflect the possible range of surface conditions, corresponding to a time to steady state ranging from 10 to 10,000 days, indicating that the surface metal concentration reflects an accumulated average of time-varying conditions. For this reason, the deposition flux measurements were compared with the deposition flux predicted by the steady state model solution. Because the steady state atmospheric concentration distribution reflects a balance between deposition and resuspension, the calculated atmospheric concentration and deposition flux are independent of the assumed resuspension rate. Calculations with $\Lambda = 0$ (i.e., no resuspension, but net loss by deposition) provide an upper bound on the effect of losses by deposition on the atmospheric concentration and associated deposition flux.

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