Analysis of Fine Particulate Matter near Urban Roadways

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The emission and dispersion of particulate matter near urban roadways has become an issue of increasing concern because of the possible health risks to humans associated with the inhalation of small particulates. Despite the potential health risk, little is known about the concentration of particulates near urban roadways or the particulates emission rates of various vehicles. Particulate matter smaller than 2.5 micrometers (microns), typically denoted $PM_{2.5}$, was studied. Data were collected along paved roads on the University of Washington campus. The results of the data collection and subsequent statistical analysis indicated, as expected, that urban buses are by far the major source of particulate emissions and that buses with low exhaust pipes generate higher concentrations of roadside fine particulate matter than buses with elevated exhausts. The findings suggest that the Environmental Protection Agency's procedure AP-42 for calculating resuspended particulate matter near urban roads is grossly inaccurate, producing values that are 9 to 20 times higher than observed fine particulate levels.

Highway traffic has long been identified as a significant source of air pollution in urban areas. Of the numerous pollutants associated with highway vehicles, particulate matter has gained attention because of recent evidence of health impacts at levels below current federal standards (*1*). Near paved urban roadways there are numerous sources of airborne particulate matter, including emission of incomplete combustion products from the tailpipe and resuspension of material from the road surface. In general, combustion processes produce smaller particles—functionally defined for sampling purposes as $PM_{2.5}$ [i.e., particles with sizes less than or equal to 2.5 micrometers (μm)] in aerodynamic diameter (*2*). Particles larger than this are generally of mineral origin, including soil particles, tire and brake wear products, and ice control compounds (*3*).

Different types of fuels, engine control technologies, and vehicle types influence the characteristics of particulate emissions. For example, emission characteristics of diesel, leaded gasoline, and unleaded gasoline vehicles differ. Also, the gross vehicle weight and the available horsepower vary the emission of particulates. Table 1 summarizes the size distribution of particles emitted by vehicles, expressed as the cumulative fraction of particulate mass smaller than a given diameter.

Among the negative impacts associated with airborne particulate matter are impaired visibility, unsightly settlements on surrounding buildings and plant life, and diminishment of road sign reflectivity and the illumination of roadway lighting. A recent study in France produced evidence that 70 to 80 percent of the soiling of objects along roadways is due to transportation particulate matter (*4*). The health effects associated with particulate matter depend on particle size. Those particles $\leq$ 10 μm in diameter (PM_{10}) are small enough to penetrate the nose or mouth and thereby deposit in the respiratory tract. Therefore the sources of both $PM_{2.5}$ and PM_{10} are of interest, including sources associated with paved urban roadways.

Of all particulate sizes, $PM_{2.5}$ is arguably the least studied, primarily due to difficulties in finding equipment sensitive enough to measure such small particulates. The established standards for estimating $PM_{2.5}$ vehicle emission rates are contained in the Environmental Protection Agency's (EPA's) AP-42 document on particulate emissions (*3*). However, other studies [e.g., Black et al. (*5*)] suggest that the AP-42 standards grossly overestimate $PM_{2.5}$ vehicular emission. Given the potentially detrimental health effects associated with $PM_{2.5}$, the significance and consequences of such overestimation could have an undesirable and misguided effect on transportation pollution control policy. The object of this study is to provide additional evidence on the suitability of AP-42 for estimating $PM_{2.5}$ vehicle emission rates.

DETERMINATION OF FINE PARTICLE IMPACTS

Study Site

The site selected for the $PM_{2.5}$ analysis is Stevens Way, located on the Seattle campus of the University of Washington. This site was chosen because of its proximity to our research laboratory, its high ratio of buses to automobile traffic, and its representative urban terrain. Stevens Way is a two-lane, paved road that passes through the University of Washington campus in Seattle, Washington. It has a curb, gutter, and concrete sidewalk and is on about a 2 percent grade. Near the study area, the buildings on the upwind side of the road and the trees, shrubs, and sloped ground on the downwind side produced both canyon and line source effects.

The buildings in this area were approximately 13 m from the centerline, on the upwind, or west, side of the roadway. Trees and shrubs were 5 to 7 m from the centerline of the roadway. On the downwind, or east, side of the roadway was an incline at about 7 m from the roadway. This incline rose about 2 m above the road and leveled off. During peak periods automobile traffic volumes can exceed 500 vehicles per hour (vph), and transit and tour bus volumes can exceed 30 buses per hour. Traffic counts during this study indicated that approximately 97 percent of the buses traveled upgrade.

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TABLE 1 Particle Size Distribution by Type of Fuel (Proportion)

Diameter	0.2 μm	1.0 μm	1.0 μm	2.5 μm	10 μm
leaded	0.23	—	0.43	—	0.64
unleaded*	0.87	—	0.89	—	0.97
unleaded**	0.42	—	0.66	—	0.90
diesel	0.73	0.86	0.90	0.92	1.00

* with catalytic converter
** without catalytic converter

Measurement Equipment

The equipment used to measure $PM_{2.5}$ must be highly sensitive and accurate. An integrating nephelometer is an appropriate $PM_{2.5}$ measuring device. It uses the fact that particulate matter in air scatters light. Integrating nephelometers measure the optical scattering coefficient (defined by the variable b_{sp}) from light in a sensing volume, integrated over all scattering angles. Many studies have shown high correlations between the scattering coefficient (b_{sp}) and particulate matter with diameters less than or equal to 2.5 μm. Waggoner and Weiss (*6*) showed that these two measures are a constant ratio with a correlation coefficient greater than 0.95.

The integrating nephelometers used in this study were designed and built by Radiance Research, Seattle, Washington. Nephelometers measure b_{sp} in the ranges of 0 to $10^{-3}m^{-1}$ or from 0 to $10^{-2}m^{-1}$. They operate at a wavelength of 475 nanometers (nm) with a Type 1A filter or at 525 nm with a Type 59 filter. The nephelometers used in this research operated at 475 nm. This is a satisfactory wavelength for the measurement of b_{sp}. The b_{sp} can be used to calculate $PM_{2.5}$, with a lower particle size limit of 0.1 μm. Data can be stored internally in intervals of 5 min or read directly in ½- or 1/15-sec intervals. Portable computers were used to record real-time data from these nephelometers using ½-sec intervals.

To accompany $PM_{2.5}$ site measurements, wind speed and wind direction were collected on a Weather Pro Model TWR-3 portable weather station. The weather station anemometer is accurate from 3 to 120 mph in 1-mph increments, or 5 to 190 kph in 1-kph increments. Wind direction was reported in 10-degree increments. In addition, traffic volumes were closely monitored by an observer who recorded information on a laptop computer so that the effects of individual vehicle types on $PM_{2.5}$ concentration could be determined.

The collection of these related traffic data to accompany $PM_{2.5}$ concentrations allowed us to statistically isolate the determinants of $PM_{2.5}$ concentrations. This was achieved by regression analysis, as described later in this paper.

Data Collection

At this location data were collected on 2 days: July 11, 1991, from 3:50 to 5:05 p.m. and July 29, 1991, from 3:35 to 5:30 p.m. Site data included distance from the edge of pavement to the nephelometers (2 m) and distances to trees and shrubs. b_{sp} readings were taken in ½-sec intervals for 190 min, or approximately 23,000 ½-sec periods. b_{sp} data were also converted to 5-min averages to total 38 periods. Automobile counts were taken in 5-min periods. The precise times that buses passed the sampling site were recorded. Whether the exhaust was above or below the bus was also recorded. Wind speed and wind direction were recorded at the sampling location when a change was noted. In general, all testing was performed on partly cloudy days with temperatures of 75°F to 85°F, wind speeds of 1 to 2 kph, wind gusts to 5 kph, and barometric pressures of 760 to 766 mm Hg. The equipment setup is shown in Figure 1. Nephelometers were placed 2 m from the roadway, and upwind and downwind concentrations were estimated in approximately ½-sec intervals.

Summary of Measurements

On July 11, 1991, ½-sec concentrations were calculated only for the downwind location, whereas on July 29, 1991, ½-sec concentrations were calculated for both the downwind and upwind locations. These measurements (see Figure 2) indicate

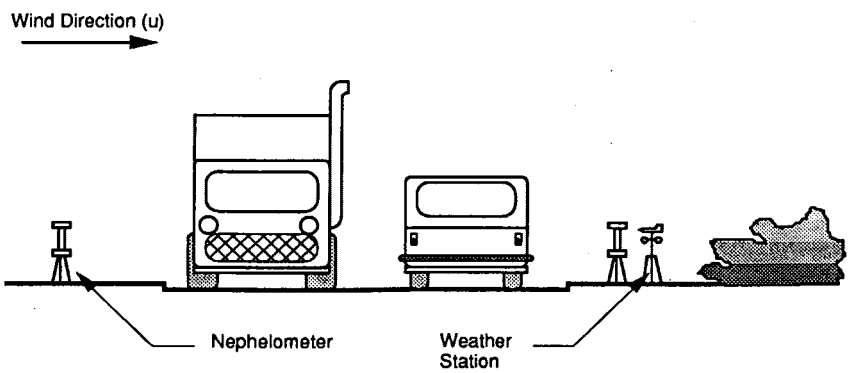
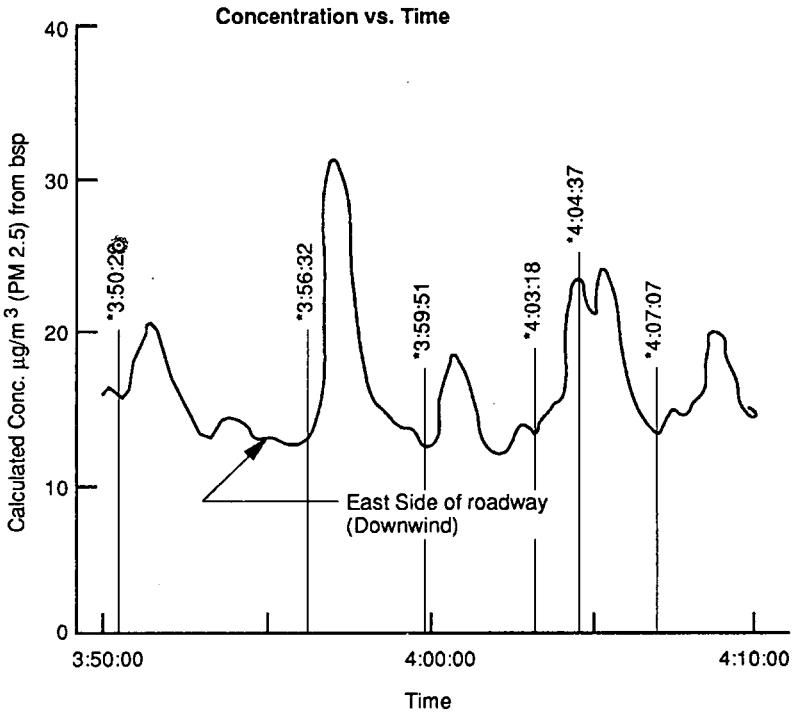


FIGURE 1 Equipment setup procedure.



* Time bus passed sampling location

FIGURE 2 $PM_{2.5}$, July 11, 1991, 3:50 to 4:10 p.m.

that each time a bus passed the sampling location, the downwind $PM_{2.5}$ concentration rose from approximately 5 to 15 g/m^3 and then returned to its initial level over a period of 1 to 1.5 min. The result was short-term, high concentrations in $PM_{2.5}$ each time a bus passed.

On the upwind side of the roadway on July 29th, three short-term spikes in concentrations were observed. These concentrations ranged between 35 and 65 g/m^3. Two of these spikes could be directly traced to tour buses with exhaust below the bus (see Figure 3).

Our data indicate that the effects of one bus were not always additive to that of a previous bus. Depending on the frequency of the buses, their emissions could be additive or their wakes could be mitigative. Automobiles tended to have very little effect on $PM_{2.5}$ concentrations. As the number of automobiles rose, the $PM_{2.5}$ concentrations also rose, but at a very low, consistent rate. When congestion occurred or the traffic speed became very low, the concentration rose and tended to stay at a high level for a longer period. Typically, congestion occurred only in one direction; therefore, while the vehicle turbulence effects were lost in the congested direction, the uncongested direction continued to cause turbulence. Also, a larger vehicle passing by slowly in the congested lane could still cause enough turbulence to lower the $PM_{2.5}$ concentration.

DATA ANALYSIS

Figure 4 shows the various analysis procedures used to compare our results with the AP-42 emission factors for $PM_{2.5}$ via resuspension and with previously reported direct measurements of vehicle tail pipe emissions. The typical approach for analyzing air pollution along roads is to estimate vehicle emission rates and put these rates in dispersion models to calculate air pollution concentration. The approach taken in this research is to measure air pollution concentrations and put these values in both a regression and a dispersion model to uncover the underlying vehicle emission rates for fine particles emitted by various vehicles.

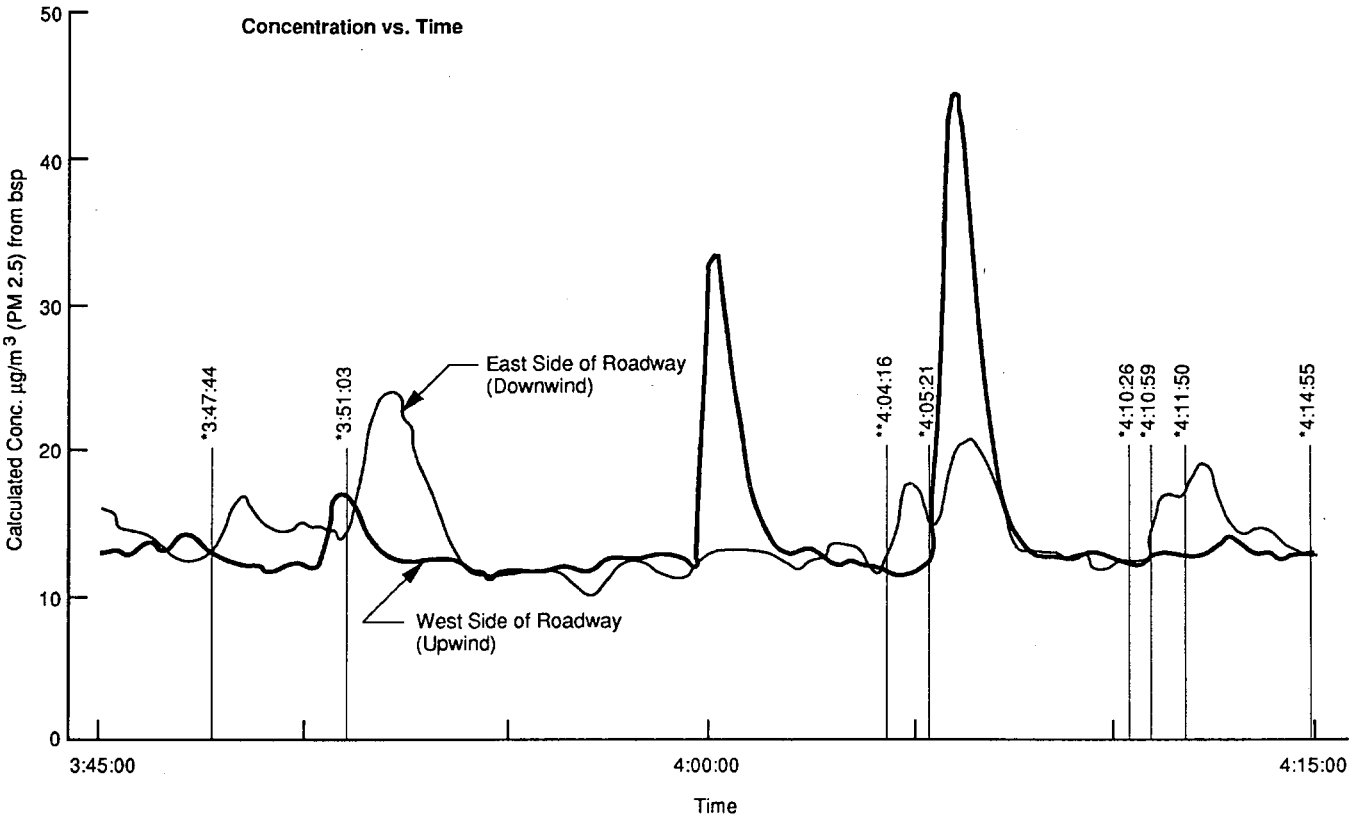
Emission Factors

The AP-42 emission factors predict the emission rate of fine particles via resuspension from the road surface due to passing vehicles as a function of road surface silt loading, SL (g/m^2). For $PM_{2.5}$ the functional relationship takes the following form:

$$e = 1.02(SL/0.5)^{0.6} \quad (1)$$

where e is the $PM_{2.5}$ emission factor for resuspended particles (g/vehicle/km).

For paved urban roads, the emission factor for $PM_{2.5}$ ranges from 0.7 to 2.4 g/vehicle/km (*3*). We can compare these values with the emission factor for fine particles directly emitted from the tail pipe of various vehicles. Black et al. (*5*) give a value of 0.01 to 0.02 g/veh/km for automobiles and values of 0.9, 0.38, and 0.16 g/veh/km for pre-1987, 1987, and 1988–1991 vintage heavy-duty diesels. Therefore, according to EPA's AP-42 emission factor, the emission of resuspended $PM_{2.5}$ from any vehicle, including cars, is approximately 4 to 15 times



* Time bus passed sampling location
** Time diesel truck passed sampling location

FIGURE 3 $PM_{2.5}$, July 29, 1991, 3:45 to 4:15 p.m.

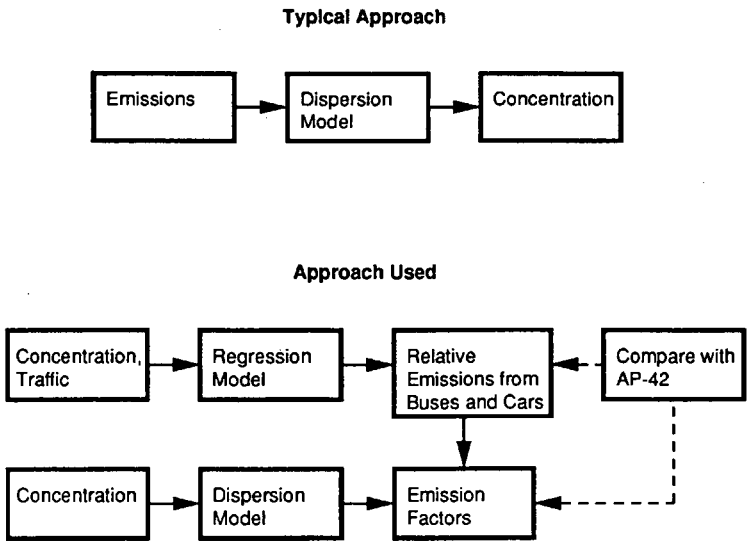


FIGURE 4 Modeling approaches.

greater than from the tail pipe of a heavy-duty diesel and 30 to 120 times greater than from the tail pipe of a car. If this were true, it would be very difficult to observe the effects of individual buses near a road. In fact, we have observed such effects and have also estimated the relative contributions from diesel buses versus automobiles using logistic regression models.

Regression Models

Five-min average $PM_{2.5}$ concentrations were calculated from the ½-sec b_{sp} readings. We then sought to statistically define the determinants of these 5-min average concentrations. To do so, two regression models were specified: one for downwind concentrations and one for upwind concentrations. The estimation of these regression coefficients allows us to determine separate $PM_{2.5}$ emission rates for cars and diesel buses, because we consider each of these two sources in the model.

The downwind 5-min average $PM_{2.5}$ concentration was regressed against the corresponding 5-min diesel bus and automobile volumes. With $PM_{2.5}$ measurements 2 m from the roadway, wind speeds of 1 to 5 kph, and dry conditions, 5-min average $PM_{2.5}$ concentrations ($\mu g/m^3$) are given by the following linear equation:

$$PM_{2.5} = 10.65 + 2.74(\text{bottom exhaust bus volume, vph}) + 1.60(\text{top exhaust bus volume, vph}) + 0.05(\text{automobile volume, vph}) \quad (2)$$

The traffic volumes are expressed in units of vehicles per hour but are computed as 5-min averages. The regression results of this equation are presented in Table 2. We attempted to separate out the effects of wind speed and wind direction, but we had too few observations and too small of a variance in our data to arrive at statistically significant variable coefficients in a regression equation.

Table 2 indicates a significant difference in $PM_{2.5}$ emissions between buses with top exhausts (standard public transit buses) and buses with bottom exhausts (tour buses). This is expected since our nephelometers are set at near-pedestrian height levels, and lower exhausts can potentially provide a much more dramatic effect.

Equation 2 can be written in general form as

$$C = a_1 + a_2[\text{vph}]_2 + \ldots + a_i[\text{vph}]_i + \ldots \quad (3)$$

where

$a_i = e_i/D_i$ for $i \neq 1$;
D_i = the atmospheric dilution factor, expressed in units of m^3/hr for a moving point source where mixing and dilution occurs in the crosswind direction (along the roadway) and in units of m^2/hr for an infinite line source where only the dilution occurs in the vertical direction; and
e_i = the emission factor for the ith source, expressed in units of g/vehicle when D_i has units of m^3/hr or in units of g/vehicle/meter when D_i has units of m^2/hr.

For $D_i = D_j$, $i \neq j \neq 1$, the dilution effect from each vehicle type is the same, and

$$a_i/a_j = (e_i/e_j)\,(D_j/D_i) = e_i/e_j \quad (4)$$

From Equation 2, $a_2 = 2.74$ for bottom exhaust buses, and $a_4 = 0.05$ for automobiles. Assuming $D_2 \approx D_4$ [i.e., the dilution effects are the same for buses and automobiles (the tailpipe emissions are of equivalent height and the resuspension emissions are both at ground level)], we deduce that $e_2/e_4 \approx 50$. This implies that a diesel bus with bottom exhaust emits approximately 50 times more $PM_{2.5}$ from the road than an automobile. In contrast, AP-42 emission factors imply that resuspension from the road is the main source of $PM_{2.5}$ and that all vehicles should contribute approximately equally.

To explore wake effects, a regression model was also run to arrive at a predictive model of upwind effects. The upwind model was of the form

$$PM_{2.5} = 16.08 + 3.98(\text{bottom exhaust bus volume, vph}) - 0.615(\text{top exhaust bus volume, vph}) - 0.614(1/\text{automobile volume, vph}) \quad (5)$$

The regression results of this equation are presented in Table 3.

In this model, buses with exhausts above the vehicle actually lowered the upwind concentration. This is because the wake

TABLE 2 Least Squares Regression of Measured Downwind 5-min Averaged $PM_{2.5}$ Concentration Calculated from b_{sp}

Independent Variable	Estimated Coefficient	Standard Error	t-Statistic
Bus volume with bottom exhaust	2.73747	0.45577	6.00629
Bus volume with top exhaust	1.60366	0.19854	8.07724
Automobile volume	0.049991	0.029113	1.71710
Constant	10.65253	1.22219	8.71596

Number of Observations	38
R-squared	0.80279

TABLE 3 Least Squares Regression of Measured Upwind $PM_{2.5}$ Concentration Calculated from b_{sp}

Independent Variable	Estimated Coefficient	Standard Error	t-Statistic
Bus volume with bottom exhausts	3.978	0.658	6.04
Bus volume with top exhausts	-0.615	0.287	-2.14
Inverse of automobile volume	-0.614	0.287	-2.140
Constant	16.08	1.589	10.11
Number of Observations		38	
R-squared		0.52355	

of the bus dispersed the pollution, lowering the $PM_{2.5}$ concentration, and its emissions were released high enough that they did not increase the $PM_{2.5}$ concentration. On the other hand, if the exhaust was below the bus, the emissions were carried by its wake, traveling along the ground and registering on the nephelometer, thus increasing the upwind $PM_{2.5}$ concentration. Aside from this, the findings were consistent with the downwind model.

Dispersion Models

In considering dispersion model alternatives (to arrive at vehicle emission rates), several model options are available. These include Gaussian line source, wake theory, box, street canyon, and intersection models. A summary of the characteristics of these models is presented in Table 4 [see Balogh and Mannering (*7*) for a complete review]. The primary dispersion model used in this research is the Gaussian line source model, although a comparison with the street canyon model (*8*) will also be made. Descriptions of the Gaussian line source and street canyon models are provided below.

The Gaussian model was developed by applying a Gaussian distribution to Fick's turbulent diffusion equation. Therefore, for the Gaussian model to hold true, the basic assumptions of the Fickian diffusion equation must be satisfied. These assumptions include spatial homogeneity (invariance in space), stationarity (invariance in time), and a large diffusion time (*9*).

Most models currently used in practice for assessing near-roadway effects are modified forms of the Gaussian model. What differentiates models are the formulation and choice of parameters. The finite line source model described below is the Gaussian dispersion model modified for highways (*10*).

$$\frac{C}{Q} = \frac{1}{\text{sigma}_z u} \times \left\{ \exp[-(z-h)^2]/2\text{sigma}_z^2 + \exp[-(z+h)^2]/2\text{sigma}_z^2 \right\} \quad (6)$$

where

C = the concentration ($\mu g/m^3$);
Q = the emission rate ($\mu g/sec$);
h = the effective height of emission release (m);
u = wind speed (m/sec);
z = the height above the ground (m);
sigma_z = the standard deviation of the distribution C in the z axis (m), adjusted for on-road vehicle wake effects = $1.5 + t_R/10$; and
t_R = the residence time of air passing over the mixing zone (sec).

In the 1970s, several models based on the Gaussian equation were developed to predict concentrations of gaseous air pollutants. Evidence has shown that there are definite differences in the dispersion of particulate matter and gases, such as gravitational settlement and coagulation (*11*). However, because the models are used here on a microscale and the focus of our measurements is on particulate matter emissions less than 2 μm, these differences are not important. When the early models were tested with gaseous tracers, they proved accurate when the wind was perpendicular to the road and the atmospheric boundary layer was neutrally bouyant. However, when winds were nearly parallel to the roadway, the concentrations predicted by the models were higher than the actual measured concentrations (*12*).

A street canyon is any roadway sheltered on both sides by complex topographical features, such as buildings, walls, earth banks, and trees. In street canyons pollutants can be trapped and concentrations elevated. Exposure to pollutants is short term for pedestrians passing through the area and long term for people working or living in adjacent buildings.

Because of the many complex street canyons in urban areas, accurate modeling is necessary. Most successful street canyon models are based on a modification of the box model. The

TABLE 4 Dispersion Model Alternatives

Type of Model	Use	Relative Accuracy	Relative Difficulty
Gaussian line source	flat open highway	good	little difficulty
wake theory	flat open highway	excellent	difficult
box	highway network	low	little difficulty
street canyon	areas sheltered by trees, buildings, walls, etc.	good	little difficulty
intersection	highway intersections	good	some difficulty

model described below assumes circular air patterns over the street (*8*). The background $PM_{2.5}$ concentration, plus C_L or C_w, is the total concentration for that respective side of the roadway.

$$C_L = \frac{7 * 10^6 * Q'}{(u + 0.5)[(x^2 + z^2)\ 0.5 + 2]} \quad (7)$$

$$C_w = \frac{(7 * 10^6)\ Q'(Hb - z)}{W(u + 0.5)\ Hb} \quad (8)$$

where

Q' = emission rate (g/m/sec),
C_L = concentration contributed by vehicle emissions for the downwind or leeward side ($\mu g/m^3$),
C_w = concentration contributed by vehicle emissions for the upwind or windward side ($\mu g/m^3$),
u = the average wind speed above the canyon (m/sec),
x = the horizontal distance to the receptor from the emissions source (m),
z = the vertical distance to the receptor (m),
Hb = the leeward side average building height, and
W = the width of the canyon (m).

As an adjustment factor for vehicle wake turbulence, 0.5 is added to the wind speed.

With the coefficients of Equation 2, the estimated $sigma_z$ (2.5 m), the average wind speed (0.6 m/sec), and the Gaussian line source model (see earlier discussion), an emission factor could be calculated. We used 1-hr average fine particle concentrations in the calculation. This averaging time is more consistent with the assumption of the line source model than are the 5-min average values used in the regression analysis. The emission factors were estimated to be 0.02 g/veh/km for automobiles and 0.8 g/bus/km for buses.

Table 5 gives a comparison of our emission rates with those of Black et al. (*5*) as well as the EPA's AP-42 computation (*3*). The AP-42 computation for paved urban roads produces emission factors that range from 0.73 to 2.42 g/veh/km. Putting these factors into the Gaussian line source model with an average of 480 vehicles/hr resulted in $PM_{2.5}$ concentrations of 108 and 358 $\mu g/m^3$ for 0.73 and 2.42 g/veh/km, respectively. The range of observed hourly average $PM_{2.5}$ concentrations was 15 to 17 $\mu g/m^3$. Therefore, the paved urban road computations resulted in $PM_{2.5}$ concentrations from 6 to 24 times higher than those actually measured. Even if every vehicle on the road was a heavy-duty diesel bus (i.e., our calculated value of 0.8 g/bus/km), the maximum concentration would not have exceeded concentrations calculated with the AP-42 recommended factor of 0.73 g/veh/km.

The regression models combined with the Gaussian line source model produced emission factors close to those used in the studies of Black et al. but much lower than those that resulted from the AP-42 study. Table 5 compares the emission factors calculated in this study with those of Black et al. (*5*) and EPA's AP-42 (*3*).

Finally, to compare the prediction of the Gaussian line source model, emission factors of 0.012 g/veh/hr and 0.51 g/bus/hr were put into the street canyon model [described by Dabbert and Sandys (*8*)]. The resulting downwind concentrations were from 0.95 to 1 times those calculated by the Gaussian line source model and from 0.9 to 1.15 times those actually measured. The resulting upwind concentrations ranged from 0.40 to 0.95 times those actually measured. This difference probably occurred because the street canyon model does not take into account the exhaust release location, which was shown to have a strong correlation with measured concentration.

STUDY LIMITATIONS

The critical limitation of this study is that $PM_{2.5}$ measurements were only taken at a single elevation on each side of the road. The Gaussian line source model used to uncover vehicle emission rates suggests that $PM_{2.5}$ concentrations should be taken at a number of points (heights above the ground surface) so the particulate plume profile can be accurately determined. This may affect our conclusions about release height as a determinant of downwind concentrations. However, the potential error introduced into our dispersion model calculations by our single point measurement is at most a factor of 2 or 3, which is not sufficient to nullify our primary finding. That is, the confidence intervals of our vehicle $PM_{2.5}$ emission estimates do not cross the AP-42 estimates, and, consequently, AP-42 clearly overestimates $PM_{2.5}$ emissions. In addition, our estimates of the relative emission rates derived from the regression model are not as sensitive to this limitation.

The other concern is our data limitations, both in terms of quantity and variability defined by meteorology, traffic volume, engine type, fuel types, exhaust controls, and different road types. There is clearly a need for an elaborate and extensive study that will precisely establish $PM_{2.5}$ emission rates. Such a study could use the same approach adopted in this study, but emphasis should be placed on extensive data collection with high variability in road and meteorological conditions. Particle size distributions should be measured in real time as a function of height above the ground, and these measurements should be correlated with short-term fluctuations in wind speed as an additional measure of mass flux from the road surface.

TABLE 5 Emission Factor Comparison by Study (g/veh/km)

Type of Vehicle	Black et al. (5)	EPA's AP-42 (3)	This Report
Automobiles	0.01 - 0.02	None	0.02
Heavy Duty Diesels/Buses	0.9 (before 87) 0.38 (1987) 0.16 (88-91)	None	0.8
Entire Roadway	0.06	0.73 - 2.42	0.08

CONCLUSIONS

Integrating nephelometers are excellent tools for examining particulate matter along the roadway. The nephelometers used in this study took b_{sp} measurements that resulted in the accurate calculation of particle concentrations with diameters between 2.5 and 0.1 μm. The measurements' accuracy and sensitivity allowed the measurement of subtle canyon effects.

Although the equipment used in this study could not measure particles smaller than 0.1 μm, such particles exist and may be significant. Near highways, nuclei-mode-sized aerosols, particles between 0.1 and 0.01 μm, can contribute an additional particulate mass equal to 30 to 50 percent of that measured between 2.5 and 0.1 μm (*13*). These particles are created by the rapid cooling of many hot, supersaturated vapors. Nuclei-mode-sized aerosols are typically created by catalyst-equipped cars. These particles tend to coagulate quickly, approximately 1 to 2 min, into and onto particles larger than 0.1 μm (*13*). Placing integrating nephelometers close to the road may cause the effects of nuclei-mode-sized aerosols to be overlooked. This problem could indicate that integrating nephelometers placed next to roads are better for application to diesel than gasoline vehicles. Nonetheless, this does not negate the fact that essentially all of the $PM_{2.5}$ we observed comes from tail pipe emissions, not resuspension. Using EPA's AP-42 emission factors for $PM_{2.5}$ results in the opposite, erroneous conclusion.

The health risks associated with $PM_{2.5}$ make it the greatest concern of particulate matter. For example, in the Puget Sound area, the Washington State Department of Ecology claims that motor vehicles annually emit 3,000 tons of combustion particles into the air and are responsible for another 177,000 tons of fine particulates from road dust. Particles resulting from combustion are clearly on the order of $PM_{2.5}$. However, when $PM_{2.5}$ is measured along paved urban roads, there is little or no contribution from road dust. Therefore, when the health effects associated with particulate matter near roads are discussed, combustion particles, not road dust, are of primary concern.

Because of the high, short-term rises in particulate matter concentrations that result from passing diesels, more real-time studies are necessary. Since major particulate matter polluters usually pass at varying intervals, modeling them as continuous sources can be erroneous.

Bus exhausts are sometimes put under the vehicle to reduce the noise associated with the bus. However, they tend to increase the particulate matter concentrations close to the roadway. In fact, our regression model results suggest that buses with exhausts below the vehicle can have nearly twice the effect on $PM_{2.5}$ that buses with exhausts above the vehicle have.

The procedure in AP-42 for calculating particulate matter concentrations along paved urban roadways is inappropriate for calculating $PM_{2.5}$. It produces values that are from 6 to 24 times higher than those actually observed.

The use of emission standards as emission factors in line source models seems to be a valid approach for determining PM concentrations near roadways. However, adjustments must be made to account for poorly maintained vehicles. The determination of emission factors necessary for calculating $PM_{2.5}$ concentrations close to those measured in the field resulted in factors close to those reported by Black et al. (*5*) as well as recent tail pipe emission standards.

Highways with complex terrain can have both line source and street canyon characteristics. Whereas buildings are predominantly responsible for canyon characteristics, gaps between buildings and perpendicular roads can produce line source characteristics. These effects were seen during our study. The buildings on the upwind side of the road and the trees, shrubs, and sloped ground on the downwind side produced both canyon and line source effects.

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