Analysis of Particulate Matter Dispersion Near Urban Roadways

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16. ABSTRACT

Particulate matter and its dispersion near urban roadways has become an issue of increasing concern due to the possible health risks associated with the human 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. This research focuses on particulate matter smaller than 2.5 micrometers (microns), and typically denoted $PM_{2.5}$, due to the high potential health risks of such small particles. Data was collected along roadways on the University of Washington. The results of the data collection and subsequent statistical analysis revealed, as expected, that urban buses were far and away the major source of particulate emissions and that buses with low exhaust pipes are more of a threat to pedestrian traffic. More interestingly, our findings suggest that procedure AP-42 for calculating particulate matter near urban roadways is grossly inaccurate, producing values that are one to two orders of magnitude higher than actually observed $PM_{2.5}$ values.

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Final Technical Report

Research Project GC8719, Task 35
Particulate Matter Dispersion Near Urban Freeways

ANALYSIS OF PARTICULATE MATTER DISPERSION NEAR URBAN ROADWAYS

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DISCLAIMER

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SUMMARY

This report focuses on the collection and statistical analysis of particulate emissions near urban roadways. Specifically, we collected data on particulate matter smaller than 2.5 microns ($PM_{2.5}$) on roadways near the University of Washington, Seattle, campus. $PM_{2.5}$ was selected because of its high potential health risk. It is important to note that $PM_{2.5}$ can be attributed almost entirely to the combustion process; existing road dust should not contribute significantly to subsequent $PM_{2.5}$ measurements.

The statistical analysis of the data indicated that the determinants of $PM_{2.5}$ concentrations near urban roadways were a function of wind direction, number of cars, and number of buses. As expected, buses were the primary source of $PM_{2.5}$ emissions, and buses with exhausts below the bus contributed much more to the $PM_{2.5}$ levels likely to be encountered by pedestrians than buses with exhausts above the bus.

Another important finding resulted from the use of observed $PM_{2.5}$ concentrations in a dispersion model to arrive at vehicle emission rates. This procedure produced vehicle emission rates that were one to two orders of magnitude smaller than EPA estimated rates determined by procedure AP-42. This result suggests that procedure AP-42 is woefully inappropriate to forecast $PM_{2.5}$ levels near urban roadways. Although the EPA does not have compliance criteria that target $PM_{2.5}$ specifically, such criteria may be developed in the future. This possibility underscores the importance of the issue of $PM_{2.5}$ measurement — for Seattle and other metropolitan areas.

INTRODUCTION

Highway traffic has long been identified as one of the most significant sources of air pollution in urban areas. Of the numerous pollutants associated with highway vehicles, particulate emissions are becoming a focus of attention, particularly because the characteristics of urban areas make them susceptible to high airborne concentrations. Low vehicle speeds, high traffic volumes, and complex topographical features (i.e., tall buildings and closely spaced streets) all contribute to high concentrations of particulate matter (1). In Washington state, fine particulate pollution contributes to an estimated 100 deaths each year (2). The many health and environmental impacts associated with particulate matter underscore the importance of understanding particulate emissions and concentrations.

To determine particulate matter concentrations near roadways, engineers must currently rely on generic emission factors that are highly uncertain. This research will enable a more realistic assessment of the environmental impacts of traffic on particulate air pollution.

Historically, the concentration and behavior of particulate matter has not been well characterized near roadways because sampling is difficult and has traditionally required a filtration method that operates for long periods (6 to 24 hours). This project studied the characteristics of particulate matter near paved roadways by using integrating nephelometers, which measure the portion of the integrated light scattering coefficient (defined by the variable b_{sp}) that is produced by the particulate matter. At a University of Washington campus site, the researchers established a relationship between the measurement of b_{sp} and the portion of particulate matter smaller than 2.5 micrometers (μm, or microns), typically referred to as $PM_{2.5}$. This relationship allowed $PM_{2.5}$ to be calculated near roadways. Weather and traffic data were gathered concurrently with $PM_{2.5}$ data, and these data were used to establish statistical relationships between $PM_{2.5}$, emissions, traffic, and weather conditions. Fourier transforms were run, and the characteristics of $PM_{2.5}$ were analyzed in the frequency domain.

BACKGROUND

There are typically four classes of airborne particulate matter.

1. total suspended particulate (TSP) is defined as the particle size fraction of airborne particulate matter that would be collected by a standard high volume sampler (Hi Vol) (3). The size of TSP varies, but the shelter design and filter of a high volume sampler limit the particle collection to sizes less than 40 *um* (4).

2. PM_{15} is defined as airborne particulate matter smaller than or equal to 15 *um* in aerodynamic diameter.

3. PM_{10} is defined as airborne particulate matter smaller than or equal to 10 *um* in aerodynamic diameter.

4. $PM_{2.5}$ is defined as particulate matter smaller than or equal 2.5 *um* in size (3).

Many factors are responsible for generating particulate matter near paved roadways. These can be listed as follows (3).

- mineral matter tracked onto the roadway,
- vehicle related deposition from engine exhaust and tire and brake wear,
- wind erosion,
- pavement wear and decomposition,
- litter,
- ice control compounds, and
- dust fall.

Each contributing factor generates particulate matter with specific characteristics. In general, the process of combustion generates $PM_{2.5}$, while material larger than 10 *um* is of mineral origin (e.g., ice compounds, soil, and pavement). Microscopic analysis has indicated TSP mass to be approximately 40 percent combustion and 60 percent mineral matter, with traces of biological matter and tire particles (5). Different types of fuels, engine control technologies, and vehicle types influence the characteristics of particulate emissions. For example, emission characteristics differ between diesel, leaded, and unleaded gasoline vehicles. 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. A similar breakdown has also been made for brake wear (Table 2) (3).

Many negative impacts are associated with airborne particulate matter. Airborne particulate matter smaller than 2.5 *um* can impair visibility. Larger particles can settle on surrounding buildings and plant life, obscuring their aesthetic value and harming the building surfaces and plants. Particulate matter can adhere to highway signing, reducing their reflectivity. In tunnels it can adhere to tiling and lighting, diminishing illumination. A recent study in France produced evidence that 70 to 80 percent of the soiling of facades along roadways is due to transportation (6).

Airborne particulate matter is also associated with health risks to human and animal life. Some possible health effects include acute respiratory symptoms, cancer, pulmonary fibrosis, emphysema, silicosis, and neurological disorders.

Table 1. Particle Size Distribution by Type of Fuel

Diameter	0.2 *um*	1.0 *um*	1.0 *um*	2.5 *um*	10 *um*
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

Table 2. Particle Size Distribution for Brake Wear

Diameter	0.43 *um*	1.1 *um*	4.7 *um*	7 *um*	10 *um*
Brake Wear	0.09	0.16	0.82	0.90	0.98

Particulate matter can contain soluble organic components that can be extracted from the particles. These components, when inhaled or respired, can dissolve in the body fluids and enter the blood stream. These soluble organic components have been shown to be mutagenic and possibly carcinogenic in laboratory assays. PM_{15}, PM_{10}, and $PM_{2.5}$ are of major concern because of their inhalable characteristics. PM_{15} and PM_{10} are small enough to be directly inhaled and can travel as far as the bronchi, the two major branches of the windpipe. $PM_{2.5}$ is small enough to easily pass the upper airways. They travel and deposit deep into the lung, where they can be retained for long periods. TSP contains particles larger than 15 *u*m, which cannot be deposited deep in the lungs. These particles are instead deposited in the nose or mouth (7).

ORGANIZATION OF THIS REPORT

This report begins with a description of the study approach and data preparation and modelling approach. A statistical assessment of the results is then presented and, finally, the conclusions of the study are given. Appendices include descriptions of the air pollution equipment tests, graphs of test results, details of statistical results, and literature related to particulate matter emission modelling.

STUDY APPROACH

EQUIPMENT

For this research, particulate matter concentrations, wind speed and direction, temperature, and pressure were measured. To determine particulate matter concentrations, two M 901 integrating nephelometers, a MINIRAM Model PDM-3 aerosol monitor, and Harvard samplers were used. A portable weather station measured wind speed and direction. A thermometer at the sampling site measure temperature. A barometer on the University of Washington campus in More Hall measured barometric pressure.

M 901 Integrating Nephelometer

Particulate matter in the air scatters light. Integrating nephelometers measure the optical scattering coefficient from light in a sensing volume, integrated over essentially all scattering angles. Many studies have shown high correlations between the scattering coefficient and particulate matter concentrations less than or equal to 3 *um*. Waggoner and Weiss (8) showed that these two measures are a constant ratio and are nearly equivalent with a correlation coefficient (r) greater than 0.95.

The integrating nephelometers used in this study were designed and built by Radiance Research, Seattle, Washington. Figure 1 is a schematic of the nephelometers provided by Radiance Research . The instruments 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 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 *um*. Data can be stored internally in intervals of 5 minutes or read directly in 1/2-second or 1/15-second intervals (9). Researchers used portable computers to record real-time data from these nephelometers.

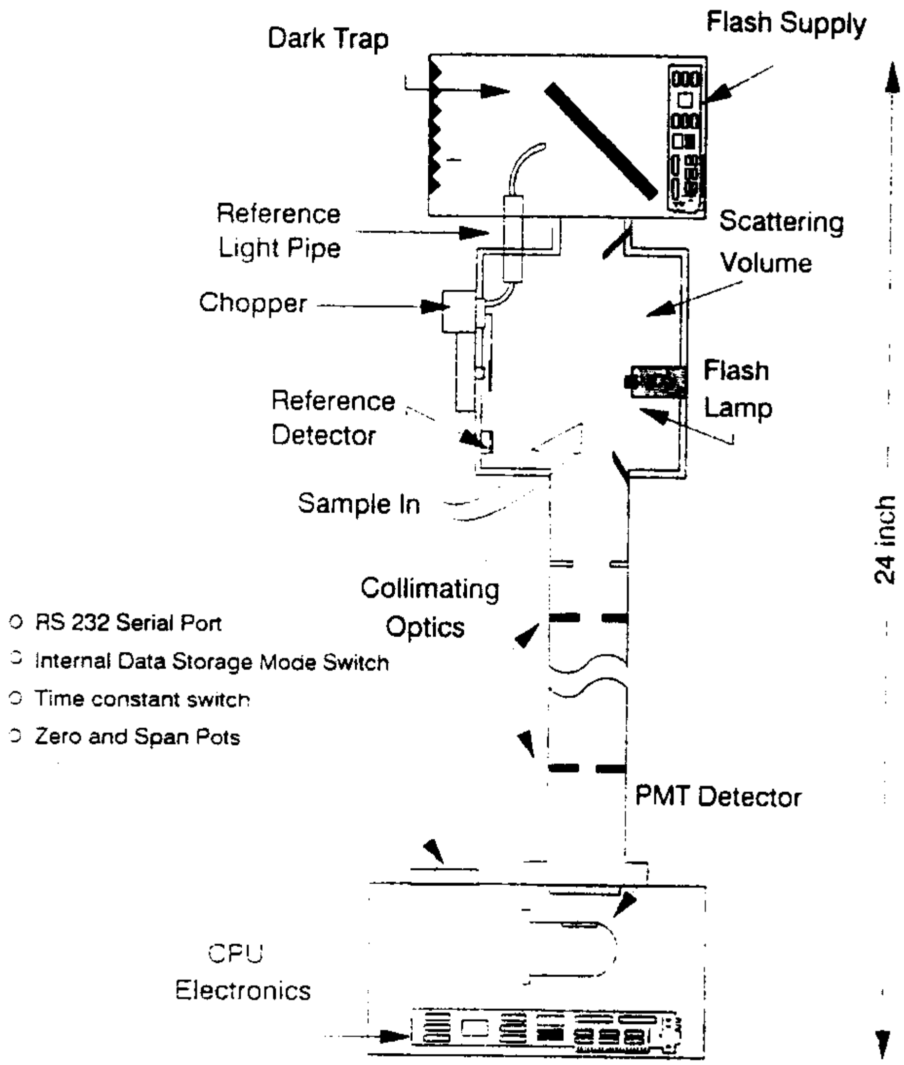


Figure 1. Schematic of Portable Nephelometer

Each nephelometer was calibrated according to specified procedures (9). First the nephelometer was filled with filtered, particle-free air and the bottom scale was set to zero. The researchers set the upper scale by filling the instrument with a gas with a known scattering coefficient and setting the upper scale to this coefficient. Gases such as refrigerant$_{12}$ (CCl_2F_2) (with a b_{sp} of 3.15 * $10^{-4}m^{-1}$) and refrigerant$_{24}$ ($CHClF_2$) (with a b_{sp} of 1.43 * $10^{-4}m^{-1}$) are normally used. The researchers calibrated the nephelometer with refrigerant$_{12}$ for all studies discussed in this report.

MINIRAM Model PDM-3

The MINIRAM (Miniature Real-time Aerosol Monitor) Model PDM-3 is a light scattering aerosol monitor of the nephelometric type. Concentration measurement ranges can be set from 10 *u*g/m^3 to 10,000 *u*g/m^3, or 100 *u*g/m^3 to 100,000 *u*g/m^3. Because this study required concentration measurements with a precision of 1 or 2 *u*g/m^3, the MINIRAM was not used for data collection near the highway. However, it is useful for finding "hot spots," areas with high particulate concentrations. It is designed for a maximum response of PM_{10}. It operates at a wavelength of 880 nm, which is higher than the typically recommended wavelength of between 475 nm and 525 nm. Time averaged data can be stored internally or read off the display (10).

Harvard Sampler

Particulate sampling is usually performed with filter collection techniques. The Harvard sampler is a filter sampler that uses impaction characteristics to segregate particles by size. When air is pumped through the sampler, airborne particles enter the sampler chambers. As they pass from chamber to chamber they collide with specially designed impactor plates. The plates are coated with a thin oil that retains particles upon impact, thus removing particles of unwanted sizes before they are collected in the filter. By knowing the weight of particles on the filter and the volume of air pumped through the filter, researchers can calculate a concentration.

For this report, filtration methods were used to measure TSP, PM_{15}, PM_{10}, and $PM_{2.5}$. But because of filter limitations, these concentrations did not include any particles less than 1*um*.

Weather Station

Wind speed and wind direction were collected on a Weather Pro Model TWR-3 weather station, and factory calibrations were assumed. The weather station anemometer is accurate from 3 mph to 120 mph in 1 mph increments, or 5 kph to 190 kph in 1 kph increments. Wind direction is measured in 10 degree increments. In future studies of this nature, a weather station that measures wind speed down to 1 kph should be used to achieve best results.

Comparison of Equipment

For purposes of comparison, the nephelometers, Harvard samplers, and MINIRAM were run side by side, and 8-hour particulate matter concentrations were measured. These tests produced the results listed in Table 3. (See Appendix A for further information.)

These data showed that PM_{10} is approximately 2.3 times greater than the concentration that was determined by the nephelometer. This result was consistent with past studies that have calculated PM_{10} concentrations near local streets to be approximately 2.7 times greater than $PM_{2.5}$, and PM_{10} concentrations near collector streets to be approximately 2.4 times greater than $PM_{2.5}$ (6). The MINIRAM produced a result approximately 4.5 times greater than the concentration determined by the nephelometer. However, the MINIRAM approximation did not include two days that the MINIRAM recorded an 8-hour average concentration of zero. Although the MINIRAM was designed for a maximum response to PM_{10}, these tests indicated that it measures concentrations closer to those of the TSP. Test number four resulted in a $PM_{2.5}$ Harvard sampler concentration that was almost the same as that calculated with the nephelometer,

Table 3. Equipment Comparison Results

Test #	Harvard Sampler			Nephelometer	MINIRAM
	$PM_{2.5}$	PM_{10}	TSP*	$PM_{2.5}$	PM_{10}+
1	—	21 ug/m^3	—	14 ug/m^3	40 ug/m^3
2	—	18 ug/m^3	—	6 ug/m^3	0 ug/m^3
3	—	19 ug/m^3	—	8 ug/m^3	0 ug/m^3
4	15 ug/m^3	—	34 ug/m^3	16 ug/m^3	30 ug/m^3
5	—	16 ug/m^3	—	9 ug/m^3	40 ug/m^3
6	—	21 ug/m^3	—	8 ug/m^3	50 ug/m^3

* TSP was calculated by using a Harvard sampler with no impactor plates.

15 ug/m^3 and 16 ug/m^3, respectively. This result reinforced the assumption that there is a high correlation between b_{sp} and the concentration of airborne particulate matter. The fact that the Harvard sampler could not collect particles less than 1 um, while the integrating nephelometer could not measure particles less 0.1 um may explain why the $PM_{2.5}$ measured by the nephelometer was slightly higher than that measured by the Harvard sampler.

To determine $PM_{2.5}$ concentrations along the roadways, only the M 901 integrating nephelometers were used. The MINIRAM was not used because its accuracy was only to the nearest 10 ug/m^3, which was not satisfactory for measuring roadway levels near 20 ug/m^3. The Harvard sampler could not be used near roadways because it required a sampling interval of approximately 8 hours to accurately determine the particulate matter concentration.

DATA COLLECTION

Site Selection

Two locations were chosen for this study. The first location, Lake Washington Boulevard in the University of Washington Arboretum, was chosen because restrictions on the roadway resulted in predominantly automobile traffic. The second location, Stevens

Way on the Seattle campus of the University of Washington, was chosen because of its high ratio of buses to automobile traffic.

Lake Washington Boulevard

The study on Lake Washington Boulevard analyzed the effects automobiles have on particulate matter concentrations near a paved roadway. The sampling occurred just north of Boyer Avenue East. Lake Washington Boulevard passes through the University of Washington Arboretum in Seattle, Washington. It is a flat, two-lane road with curb, gutter, and grass shoulders. There are no buildings in this area, and trees and shrubs are approximately 10 to 13 meters from the roadway centerline. During peak periods the roadway's traffic volumes can exceed 2,200 vehicles per hour (vph), and the traffic consists almost entirely of automobiles. During off-peak periods approximately 1 percent of the traffic consists of small trucks. Because of area bridge height restrictions, large trucks and buses avoid this portion of roadway.

At this location, data were collected on three days, June 27, 1991, from 12:00 PM to 2:00 PM; July 3, 1991, from 1:20 PM to 3:20 PM; and July 7, 1991, from 1:25 PM to 5:55 PM. Site data included distance from the edge of the pavement to the nephelometers and distances to trees and shrubs. To measure b_{sp}, 102 5-minute readings were taken and converted to 5-minute averages. Traffic counts were taken in similar, 5-minute periods for both diesels and other vehicles. Average wind speed and wind directions were obtained from the weather station. (See Figure 2 for the setup procedure.) Temperature readings were taken periodically throughout the tests, and beginning and ending barometer readings were obtained. In general, all testing was performed on clear days with temperatures of 70 to 80 degrees, wind speeds of 1 to 5 kilometers per hour (kph), wind gusts of 6 to 15 kph, and barometric pressures of 760 to 766 millimeters mercury (mm Hg).

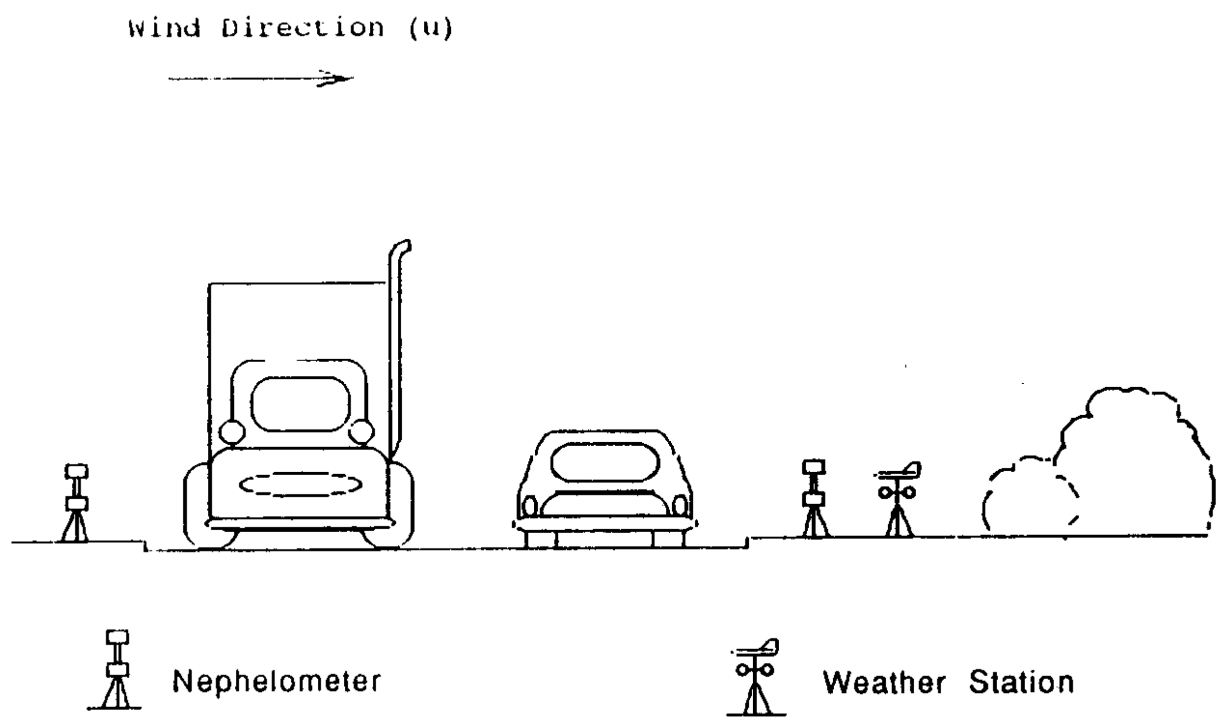


Figure 2. Equipment Setup Procedure

Stevens Way

The Stevens Way study analyzed the effects that buses and automobiles have on particulate matter concentrations in areas with complex terrain. The samples were taken on Stevens Way just south of Grant Lane. 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 meters from the centerline, on the upwind, or west, side of the roadway. Trees and shrubs were from 5 to 7 meters from the centerline of the roadway. On the downwind, or east, side of the roadway was an incline at about 7 meters from the roadway. This incline rose about 2 meters above the road and leveled off. During peak periods automobile traffic volumes can exceed 500 vph, and transit and tour bus volumes can exceed 30 buses per hour (bph). Traffic counts during this study indicated that approximately 97 percent of the buses traveled upgrade.

At this location data were collected on two days, July 11, 1991, from 3:50 PM to 5:05 PM; and July 29, 1991, from 3:35 PM to 5:30 PM. Site data included distance from the edge of pavement to the nephelometers (2 meters) and distances to trees and shrubs. b_{sp} readings were taken in half-second intervals for 190 minutes, or approximately 23,000 half-second periods. b_{sp} data were also converted to 5-minute averages to total 38 periods. Automobile counts were taken in 5-minute 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. Temperature readings were taken periodically throughout the tests, and beginning and ending barometer readings were obtained. In general, all testing was performed on partly cloudy days with temperatures of 75 to

85 degrees, wind speeds of 1 to 2 kph, wind gusts to 5 kph, and barometric pressures of 760 to 766 mm Hg.

DATA PREPARATION AND MODELLING

Appendix D discusses models that can be used to calculate emission factors and air pollution dispersion near roadways. The typical approach for analyzing air pollution along the roadway is to estimate emission factors and put these factors in dispersion models to calculate air pollution concentrations near roadways. However, one approach used for this project was to take 5-minute average $PM_{2.5}$ concentrations measured near roadways and regress them with traffic data to determine the relative $PM_{2.5}$ contributed by specific types of vehicles. These $PM_{2.5}$ contributions were put into dispersion models to determine an emission factor. These emission factors were then compared to those typically used in the analysis of $PM_{2.5}$ near paved roadways. Figure 3 compares the typical procedure with that used in this analysis.

Fourier transforms are typically performed to identify the frequency components that make up a continuous wave form. Fourier analysis was used to describe the rapid fluctuations in $PM_{2.5}$ concentrations that occurred on time scales of less than 5 minutes. The consideration of more rapid fluctuations allowed the researchers to determine averaging times that would better describe the data than the 5 minutes used in the regression analysis.

The "fast" Fourier transform was used for this analysis. This is a finite, discrete version of the Fourier transform. The half-second b_{sp} coefficients taken at Stevens Way on July 29, 1991, were smoothed and differenced. Next, the fast Fourier transform was performed. The transform's output consisted of real and imaginary components of the discrete Fourier transform. The sum of the squares of these complex Fourier components were used to develop a periodogram. The periodogram was then used to study the power spectrum of the data.

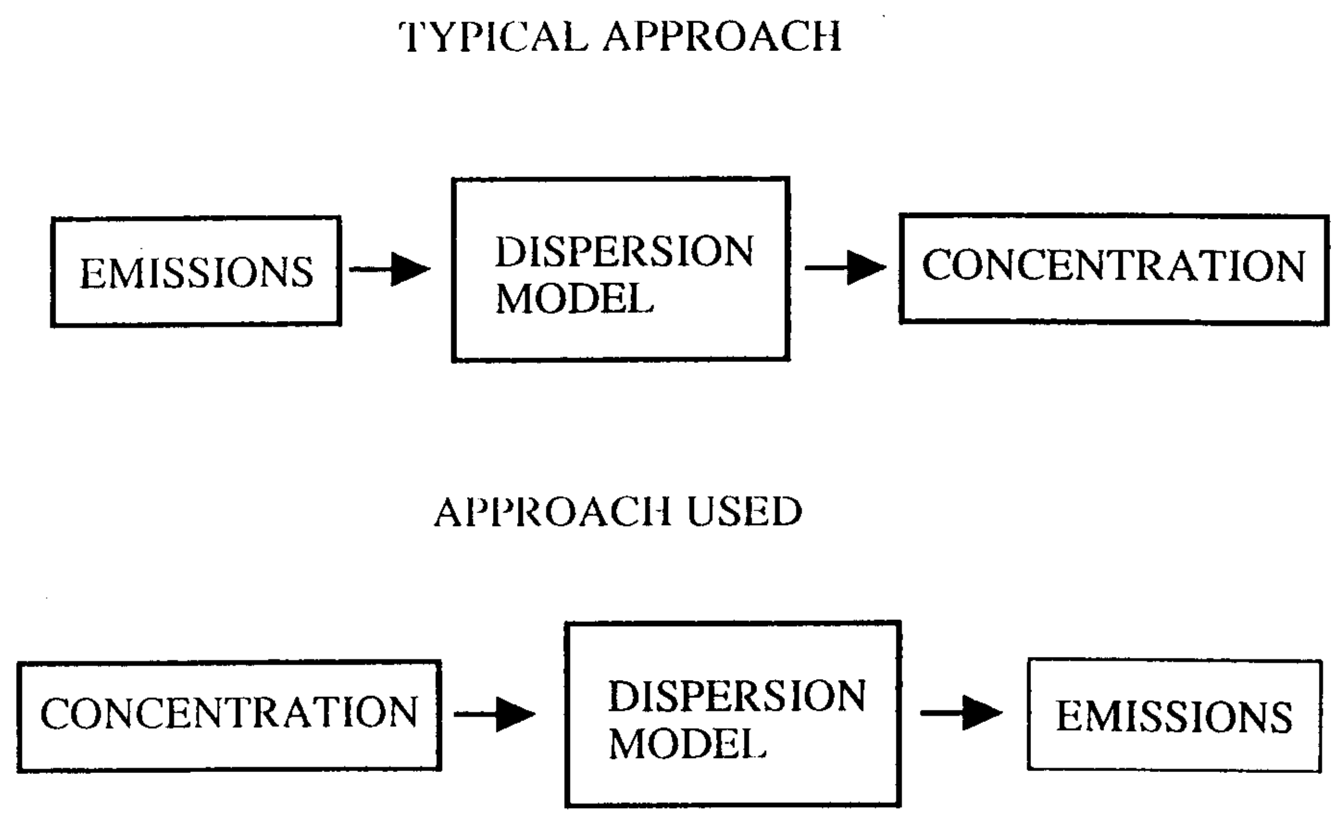


Figure 3. Modelling Approaches

RESULTS

LAKE WASHINGTON BOULEVARD

The Lake Washington Boulevard data were put into a statistics program called StatView II created by Abacus Concepts Inc., 1988. This is a Macintosh based statistics program. With this program, multinomial regressions were attempted. Theoretically, regression models could be developed with these data to predict $PM_{2.5}$, but because of the limits in the range of $PM_{2.5}$, the models were inconsequential. $PM_{2.5}$ ranged from approximately 2 ug/m^3 to 18 ug/m^3, which was not a large enough range of concentration to yield satisfactory models.

As an example of such models, a simple regression with a downwind concentration and the total number of vehicles per 5-minute interval was performed. Trucks and buses were not observed because of overpass restrictions. The 5-minute average $PM_{2.5}$ concentration (C), in ug/m^3, is produced by the following formula:

$$C = 8.75 + 0.048\ (\#cars) \qquad \text{(Eq. 1)}$$

The t-statistic for cars per 5-minute period was satisfactory at 2.15, but the R^2 value of .05 made this model unsatisfactory. This model would not even be mentioned except for the following interesting relationship. The model yielded a rise in $PM_{2.5}$ of .048 ug/m^3 per vehicle per 5 minutes. The combination of this concentration, a $sigma_z$ of 1.52 m for 2 m from the roadway (see Figure 2), an average wind speed of 0.6 m/s, and the line source model yielded an emission factor of 0.013 g/veh/km. This emission factor was very close to those that were used in previous studies (11) that found that light duty, gasoline powered vehicles (e.g., cars and trucks of less than 3 tons gross vehicle weight) between 1976 and 1981 and 1981 and 2000 produced particulate emission factors of 0.02 g/veh/km and 0.01 g/veh/km, respectively.

The particulate matter emission factor procedure for paved urban roads, mentioned in the AP-42, was also used to calculate an emission factor for this area (8). The equation for calculating an emission factor (e) for the roadway is

$$\text{g/veh/km is } e = k\ (sL/.5)^{P} \qquad \text{(Eq. 2)}$$

As defined, the base emission factor (k) is 1.02 g/VKT, and the exponent p is 0.6 for $PM_{2.5}$ (6). The silt loadings for a collector street range from 0.29 to 2.11 (3). With this information, the emission factor ranged from 0.73 g/veh/km to 2.42 g/veh/km. This factor produced $PM_{2.5}$ concentrations that were from 56 to 186 times higher than those measured in the field, indicating that the particulate matter emission factor procedure for paved urban roads was inappropriate for calculating $PM_{2.5}$. In fact, the silt loading resulting from soil on the roadway should have had very little effect on $PM_{2.5}$ because the majority of soil particles are larger than 2.5 *u*m.

Even though a model was not developed, some interesting relationships between upwind and downwind concentrations, and parallel and crosswind wind directions were revealed (see the Lake Washington Boulevard box plots presented in Figure 4). For these box plots, the $PM_{2.5}$ concentration was estimated from b_{sp}. The average $PM_{2.5}$ concentration was approximately 6 *u*g/m^3. "Parallel" was defined as wind blowing at 0° to 45°, 135° to 225°, or 315° to 0° to the roadway, and "crosswind" was defined as wind blowing at 45° to 135° and 225° to 315° to the roadway. Most interesting was that the fourth spread, the difference between the upper fourth and the lower fourth of the data, was much smaller when the wind was parallel to the road than when it was crosswind. The fourth spread for the parallel situation was approximately half of what it was for the crosswind case. This difference can probably be explained by the fact that when the wind blows parallel to the road, car wakes have a greater effect on dispersion than when it blows perpendicularly (12). During this study the flow of traffic was steady while the wind would often gust. Thus, perpendicular wind gusts affected the concentrations, but the effects of parallel wind gusts were negated by the vehicles.

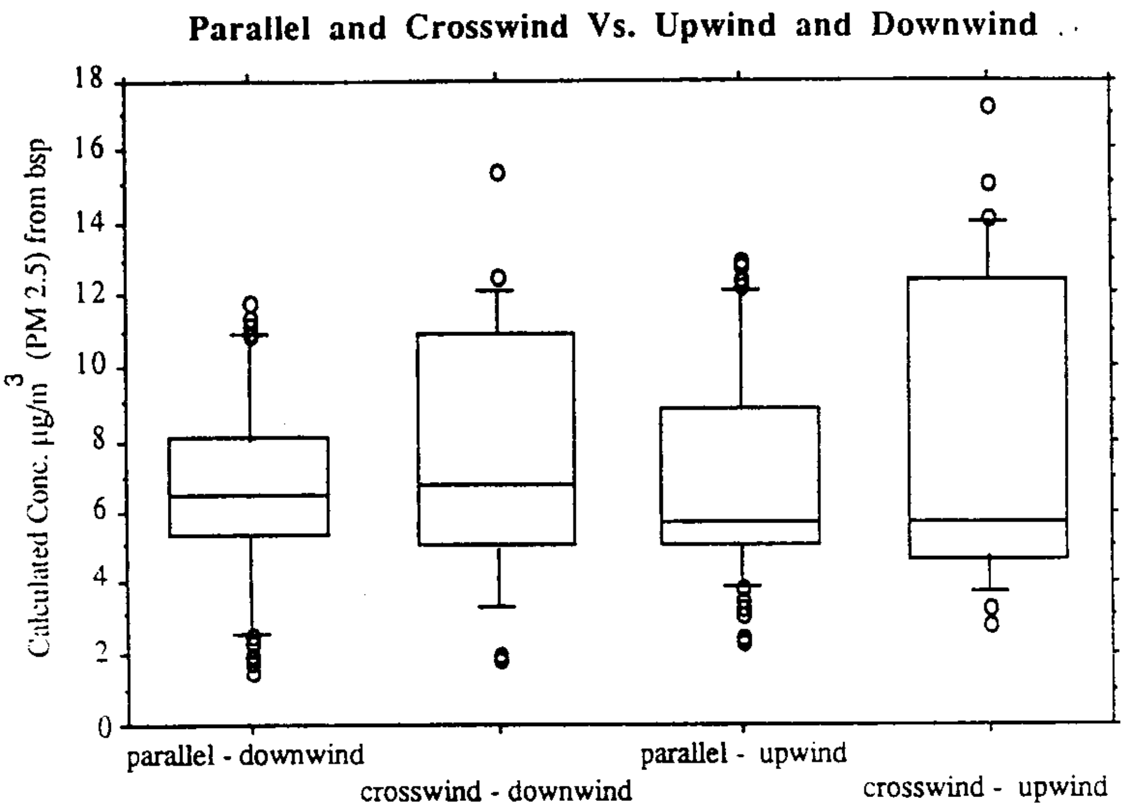


Figure 4. Box Plots for Lake Washington Blvd.

STEVENS WAY

At the Stevens Way location, nephelometers were placed 2 meters from the roadway, and upwind and downwind concentrations were estimated in approximately half-second intervals. On July 11, 1991, half-second concentrations were calculated only for the downwind location, while on July 29, 1991, half-second concentrations were calculated for both the downwind and upwind locations. Concentration versus time graphs were developed for these data (see Appendix B for a complete set of graphs). These graphs (see Figure 5) show that each time a bus passed the sampling location, the downwind $PM_{2.5}$ concentration rose from approximately 5 ug/m^3 to 15 ug/m^3 and then returned to its initial level over a period of 1 to 1.5 minutes. 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 ug/m^3 and 65 ug/m^3. Two spikes could be traced to Gray Line Tour buses with exhaust below the bus. However, the third spike was questionable because of an interruption in the bus data collection process (see Figures 6, 7, and 8).

These data show that the effects of one bus were not always additive to that of a previous bus. Depending upon the frequency of the buses, their emissions could be additive or their wakes could be deleterious. Automobiles tended to have very little impact 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, if a larger vehicle passed by slowly in the

Concentration Vs Time

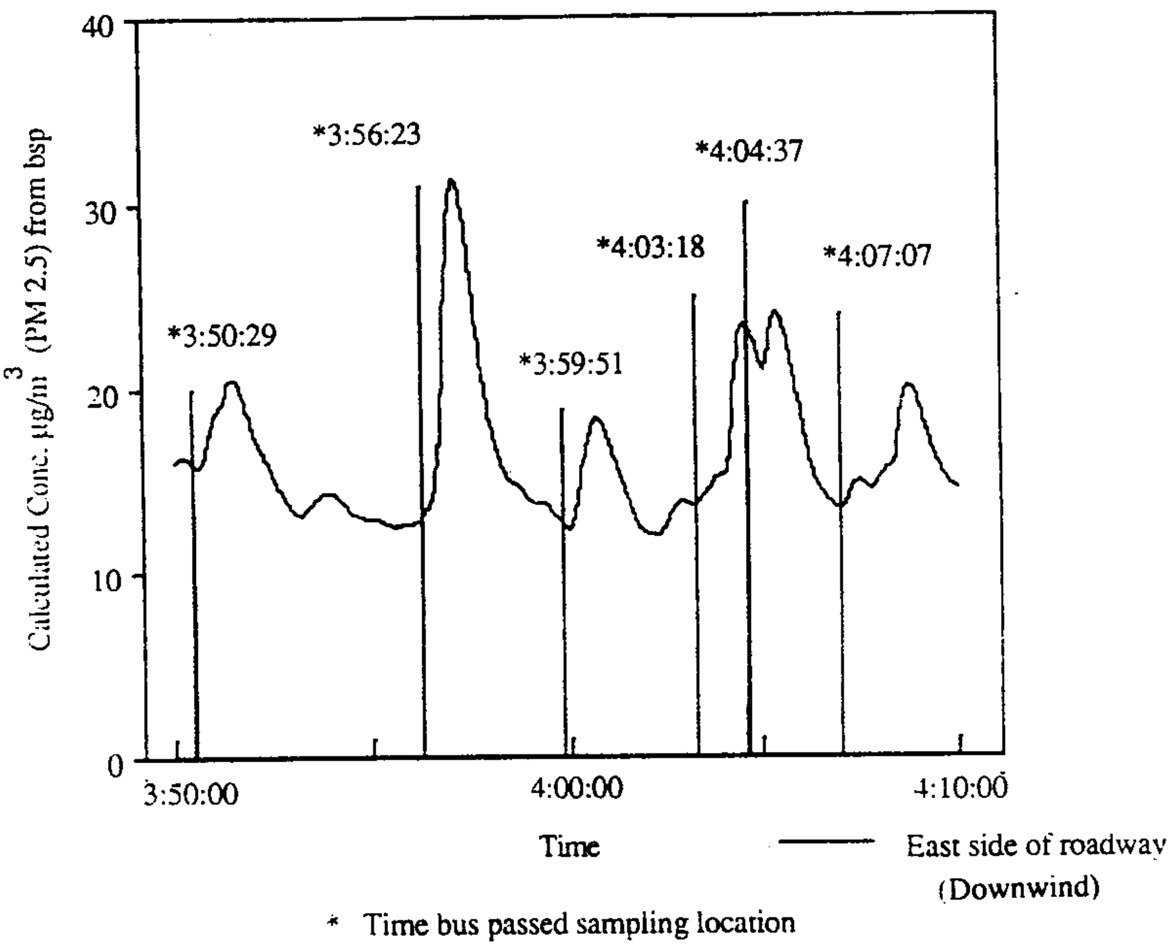


Figure 5. $PM_{2.5}$ Graph, 7/11/91, 3:50 PM to 4:10 PM

Concentration Vs Time

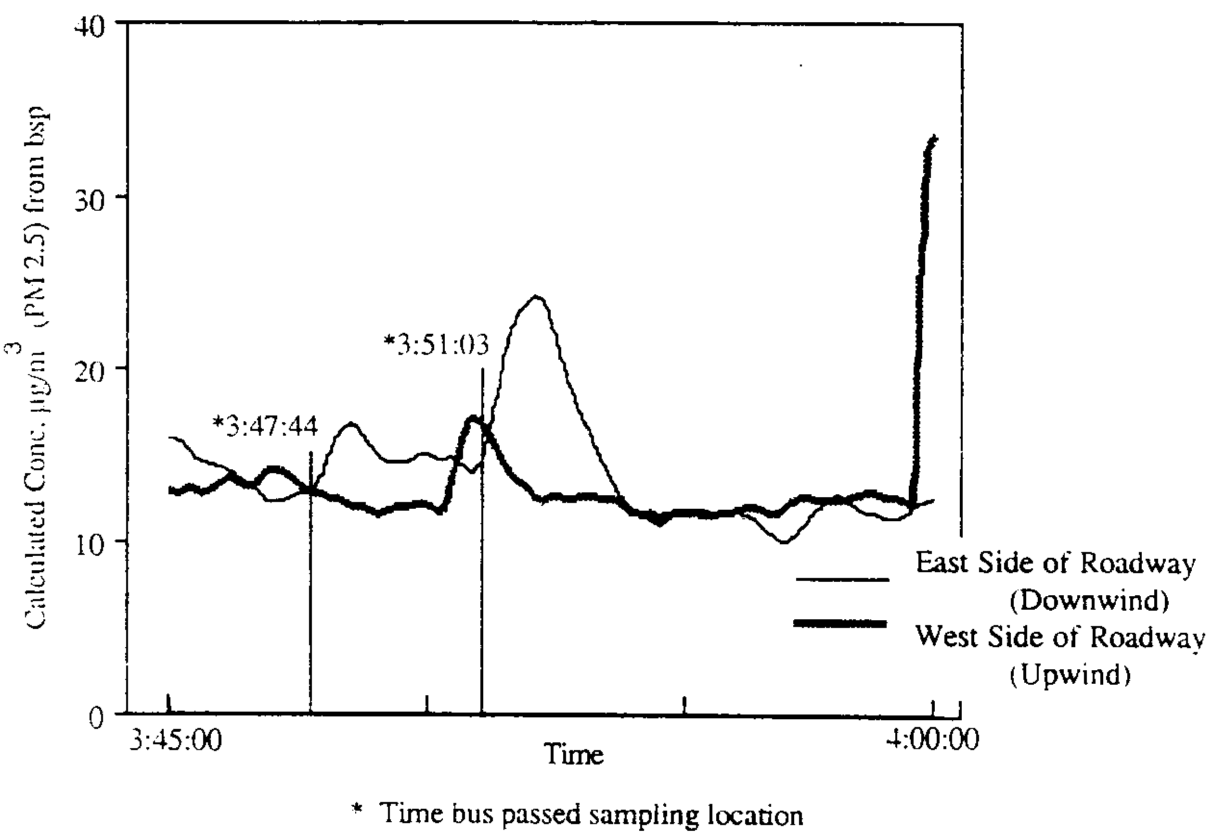
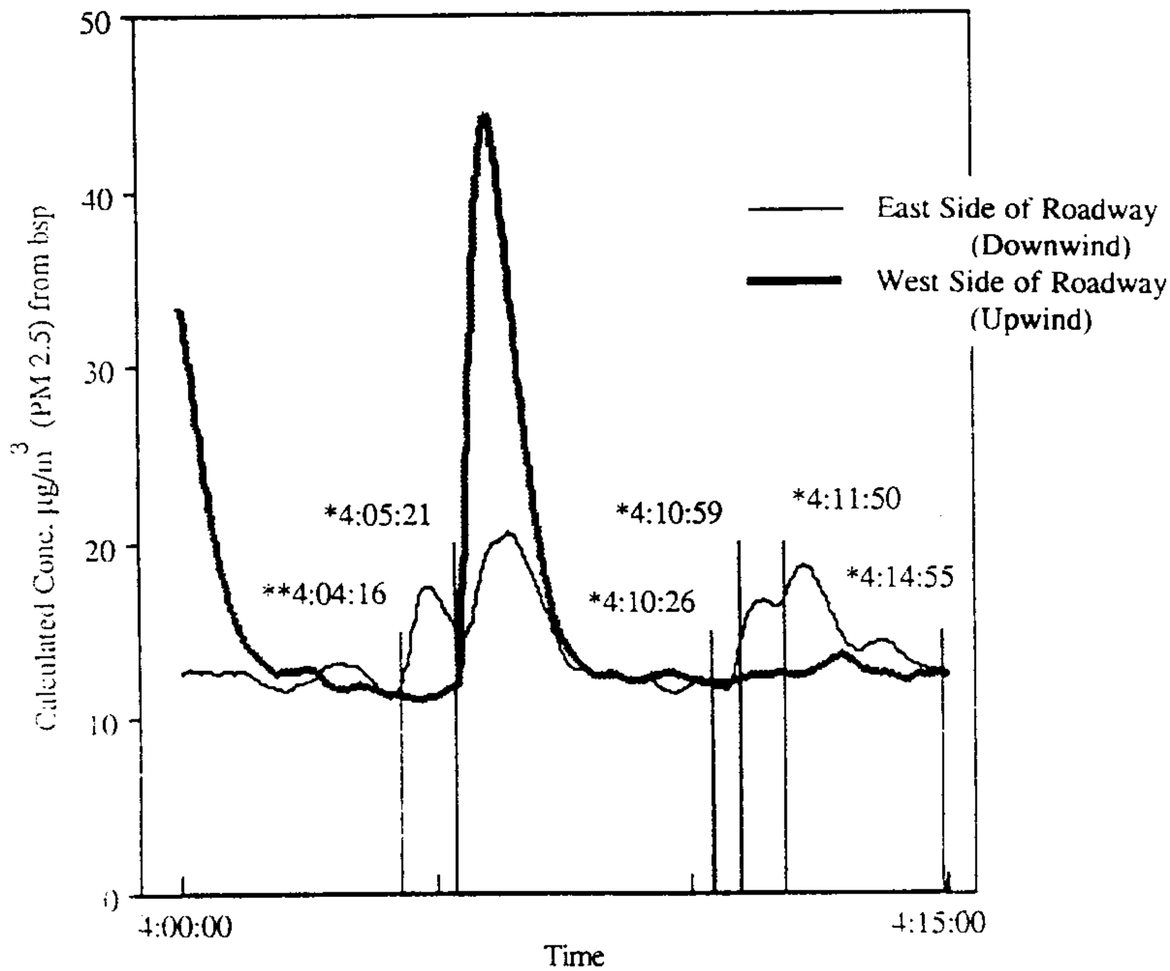


Figure 6. $PM_{2.5}$ Graph, 7/29/91, 3:45 PM to 4:00 PM

Concentration Vs Time



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

Figure 7. $PM_{2.5}$ Graph, 7/29/91, 4:00 PM to 4:15 PM

Concentration Vs Time

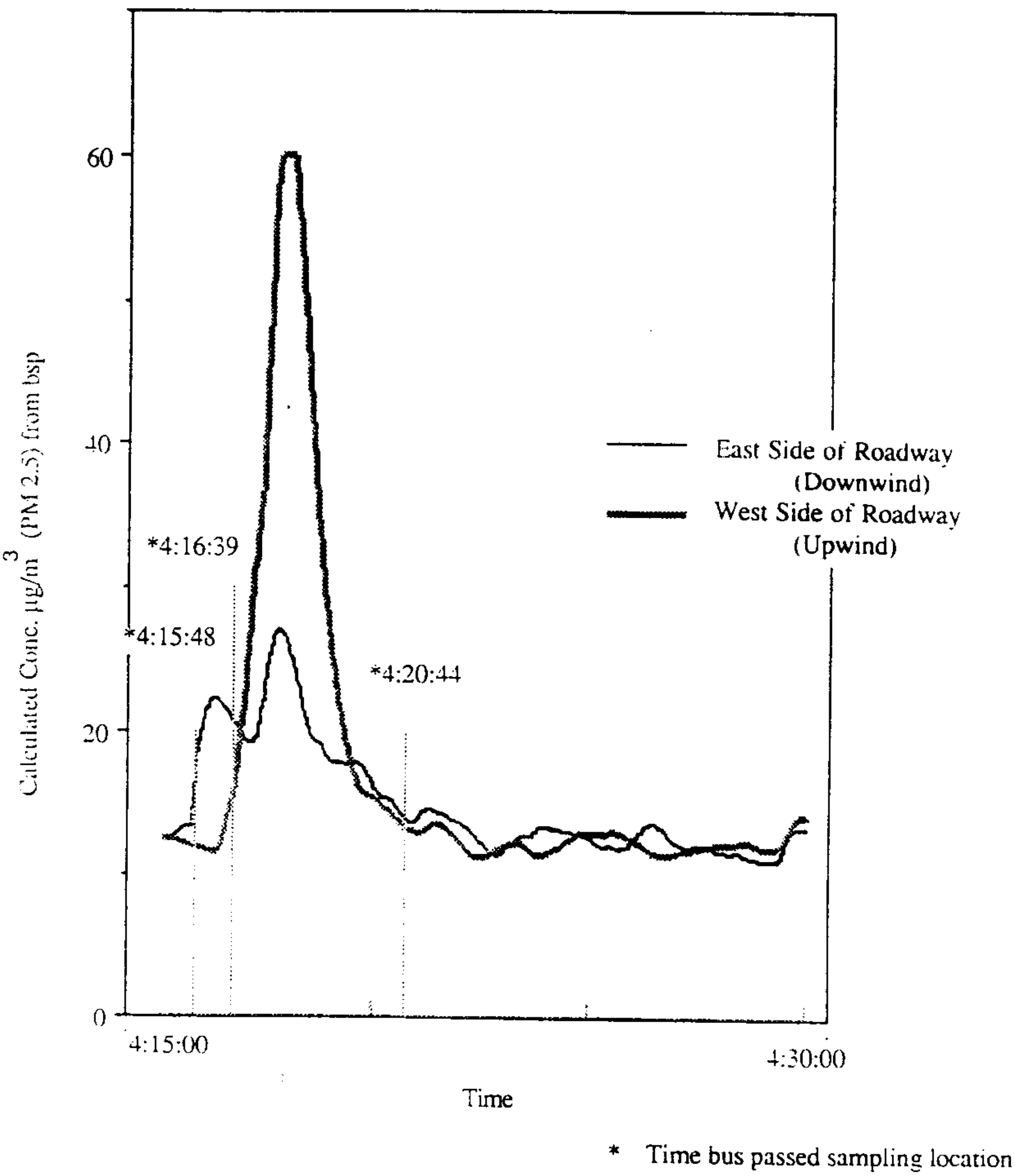


Figure 8. $PM_{2.5}$ Graph, 7/29/91, 4:15 PM to 4:30 PM

congested lane, it could still cause enough turbulence to lower the $PM_{2.5}$ concentration. This seems to have been the case at 4:14:26 in Figure 9.

Stevens Way Frequency Study

For this analysis, a Fourier transform was performed on the smoothed time series data collected by the nephelometers, and a periodogram was developed. The periodogram for the west (upwind) and east (downwind) side of the street during the same time interval can be seen in figures 10 and 11.

A comparison of the power spectral density (psd) of both sides of the roadway revealed several interesting observations. Figures 10 and 11 show that the downwind psd was stronger than the upwind psd. This was expected, since the downwind concentration was more responsive to vehicle traffic.

The psd seemed to be offset so that when one peak was at a maximum the corresponding peak was at a minimum. For example, at the frequency 0.0045, or about 220 seconds, information was observed on the upwind side of the roadway but not the downwind side. Just the opposite was observed at a frequency of 0.0035, or about 285 seconds. This effect was probably the result of incomplete mixing and the canyon effects associated with this location. When a bus passed, it emitted a puff of pollution. This puff would become caught in the wind currents and be transported in a circular motion above the roadway. As the puff moved, it was sampled on one side of the road and not at the other. With the street canyon approximately 13 m wide and a wind speed of .5 m/s, the time for the puff to travel from the upwind nephelometer to the downwind nephelometer was about 60 seconds, approximately equal to the lag in the periodograms.

Stevens Way Regression Modelling

The Stevens Way data were put into a statistics program called Statistical Software Tools (SST), 1986. This is a DOS based statistic program and was used to perform multiple regressions. All data used in the following regressions were taken in 5-minute intervals. Details on each regression can be seen in Appendix C.

Concentration Vs Time

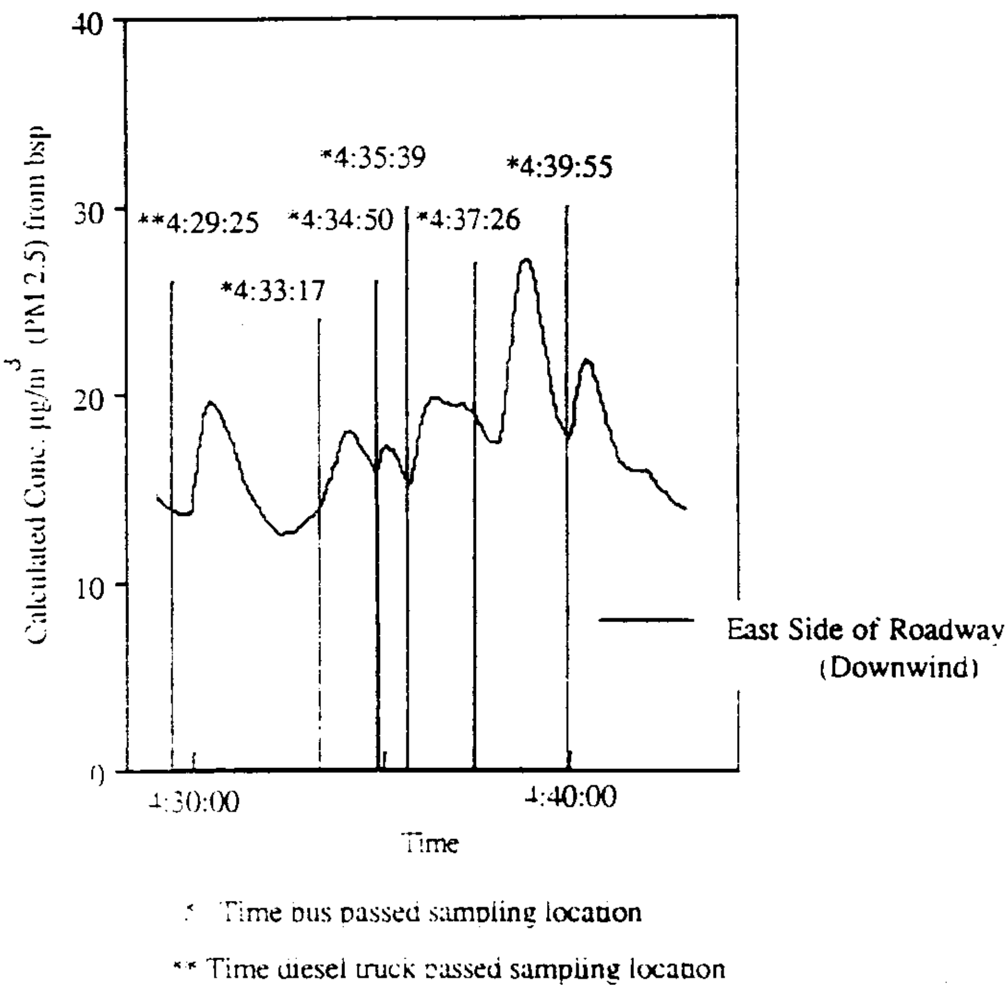


Figure 9. $PM_{2.5}$ Graph, 7/11/91, 4:10 PM to 4:29 PM

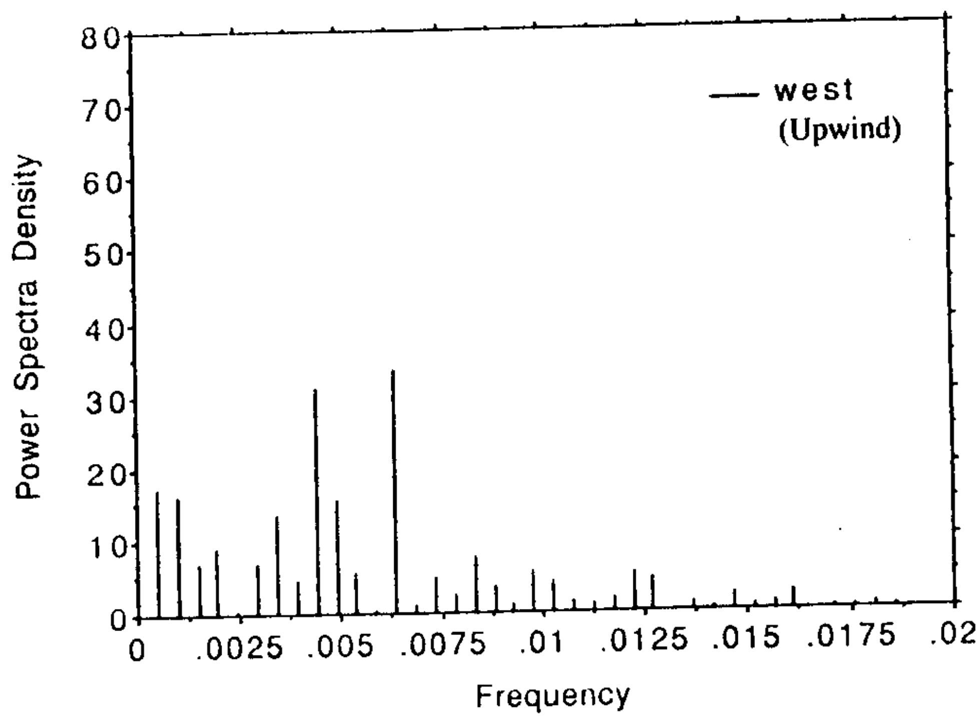


Figure 10. Periodogram for Upwind or West Side of Street

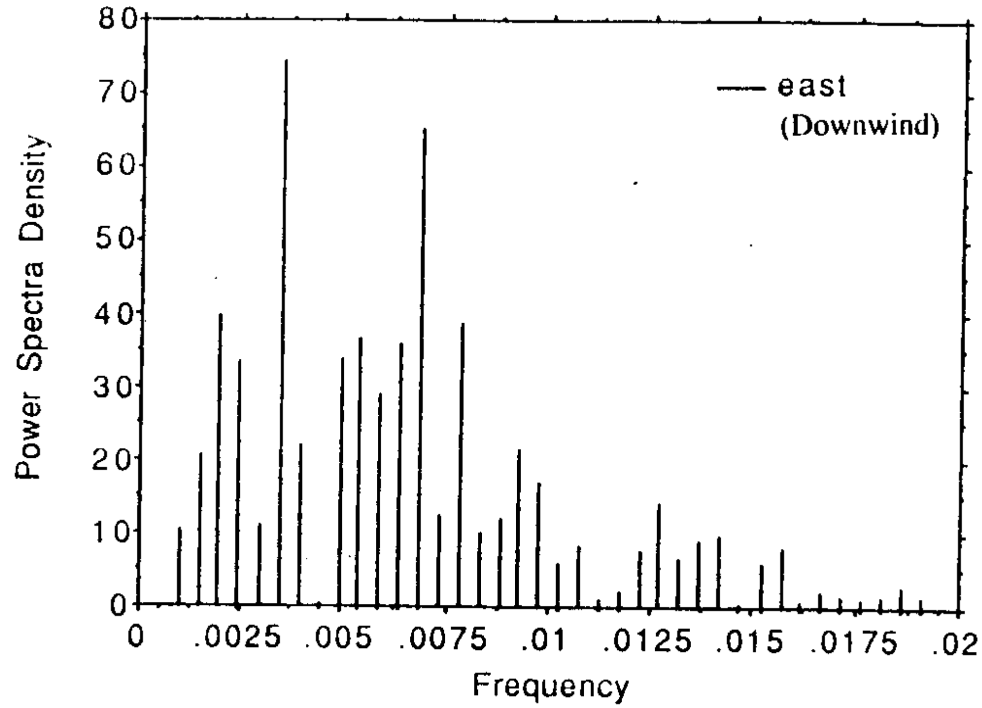


Figure 11. Periodogram for Downwind or East Side of Street

Downwind Study. The downwind $PM_{2.5}$ concentration was regressed with the total number of diesels (i.e., buses) and the total number of cars. In this model, when measurement occurs 2 meters from the roadway and wind speeds are approximately .5 m/s, the 5-minute average $PM_{2.5}$ concentration can be determined by the following equation:

$$C_{down} = 10.77 + 0.04505(\#cars) + 1.849(\#buses) \qquad \text{(Eq. 3)}$$

where 10.77 is the background concentration, which is close to the actual measured concentration of 10 ug/m^3.

The t-statistic for the total number of buses (11.042) was considered excellent, but the t-statistic for the total number of cars (1.483) was considered marginal. The corrected R^2 value of 0.76493 was satisfactory for this model.

Because of the marginal t-statistic for the variable, number of cars, it was removed, and a new regression was performed. The new model is represented by the following equation:

$$C_{down} = 12.56 + 1.831(\#buses) \qquad \text{(Eq. 4)}$$

Both t-statistics for the total number of buses (10.78) and for the constant (29.62) were considered excellent. The corrected R^2 value dropped insubstantially from the previous model to 0.75709.

Another interesting relationship was obtained when the bus category was separated by the location of the exhaust. The downwind, 5-minute $PM_{2.5}$ concentration was regressed with the total number of buses with exhausts below the bus, the total number of buses with exhausts above the bus, and the total number of cars. The following equation was determined for downwind, 5-minute average $PM_{2.5}$ concentrations:

$$C_{down} = 10.65 + 2.73(\#bottom) + 1.60(\#top) + .050(\#cars) \qquad \text{(Eq. 5)}$$

The t-statistic for the constant was 8.71, 6.00 for the total number of buses with exhausts below the bus, 8.08 for buses with exhausts above the bus, and 1.71 for the total number of cars. The corrected R^2 value was 0.78539. The most interesting result was that the emissions released from under the bus had an impact on the concentration that was 1.71 times greater than the impact from the exhaust at the top of the bus. This finding was intuitively correct, since there was no height differential between the exhaust and the nephelometer.

After reviewing the models, the researchers decided that, to make comparisons between studies, the model that used both the total number of cars and the total number of buses, or Equation 3, was most appropriate. Even though automobile traffic contributed very little to downwind $PM_{2.5}$ concentrations, it had to be considered. $Sigma_z$ could be calculated as 1.52 at a distance of 1.5 meters from the roadway with the equation

$$1.5 + 0.8 \times (1 + .0002x)^{-0.5} \qquad \text{(Eq. 6)}$$

An average wind speed of 0.6 m/s was determined from the data. With the coefficients of Equation 3, the estimated $sigma_z$, the average wind speed, and the finite line source model (see Appendix D), an emission factor could be calculated. The emission factors were calculated at 0.012 g/veh/km for automobiles and 0.51 g/bus/km for buses.

These emission factors were within the range of those used in previous studies (11). As discussed previously, it is appropriate to assume a particulate emission factor for light duty, gasoline powered vehicles for 1976 to 1981 of 0.02 g/veh/km, and for 1981 to 2000 of 0.01 g/veh/km. Also, heavy-duty diesels have possible emission factors of 1.0 g/veh/km for the years 1980 to 1985, 0.9 for 1986, 0.38 g/veh/km for 1987, and 0.16 g/veh/km for 1987 to 1991. These findings reinforce the possible emission factor of 0.51 g/bus/km calculated by this study.

The AP-42 computation for paved urban roads produces emission factors that range from 0.73 g/veh/km to 2.42 g/veh/km (5). Putting these factors into the finite line source model resulted in $PM_{2.5}$ concentrations of 108 ug/m^3 for 40 vehicles at 0.73 g/veh/km and 530 ug/m^3 for 60 vehicles at 2.42 g/veh/km. The range of observed 5-minute average $PM_{2.5}$ concentrations was 12 ug/m^3 to 26 ug/m^3. Therefore, the paved urban roadway computations resulted in $PM_{2.5}$ concentrations from 9 to 20 times higher than those actually measured. Even if every vehicle on the roadway was a heavy-duty diesel bus (i.e., 0.51 g/bus/km) the maximum concentration would not have exceeded concentrations calculated with the AP-42 recommended factor of 0.73 g/veh/km.

In summary, the regression models and the line source model produced emission factors close to those used in Black's studies but much lower than those that resulted from the AP-42 study. Table 4 compares the emission factors calculated in this study with those of Black and AP-42.

Upwind. Multiple regression was also performed on the upwind $PM_{2.5}$ concentrations. To get any reasonable model, the variable "number of buses with exhausts below the bus" had to be used. All model variables were 5-minute averages. The first model resulted from a regression of upwind $PM_{2.5}$ concentrations with the total number of buses with exhaust above the vehicle, the total number of buses with exhaust below the vehicle, and the total number of cars. In this model, when measurements were taken 2 meters from the roadway and wind speeds were approximately .5 m/s, the $PM_{2.5}$ concentration in 5- minute averages could be determined by the following equation:

$$C_{up} = 16.08 - 0.615(\#top) + 3.98(\#bottom) + (-71.84/\#cars) \quad \text{(Eq. 7)}$$

The t-statistic for the constant was 10.11, -2.140 for the total number of buses with exhaust above, 6.043 for the total number of buses with emissions from under the bus, and -1.261 for the inverse number of cars. The only questionable t-statistic was that of the inverse number of cars. The corrected R^2 value of 0.482 was marginal.

Table 4. Emission Factor Comparison by Study

Type of Vehicle	Black's Study	AP - 42 Study	This Report
Cars	0.01 - 0.02	None	0.013
Heavy Duty Diesels/Buses	0.9 (before 87) 0.38 (1987) 0.16 (88-91)	None	0.51
Entire Roadway	0.06	0.73 - 2.42	0.05

In this model, the location of the exhaust was very important to upwind concentration. Buses with exhausts above the vehicle actually lowered the upwind concentration. This is because the wake of the bus dispersed the pollution, lowering the $PM_{2.5}$ concentration, while its emissions were released high enough that they did not increase the $PM_{2.5}$ concentration (see Figure 12). On the other hand, if the exhaust was below the bus, the emissions were carried by its wake, travelling along the ground and registering on the nephelometer, thus increasing the upwind $PM_{2.5}$ concentration (see Figure 13).

The second model resulted from a regression of upwind $PM_{2.5}$ concentrations with the total number of buses with exhaust above the vehicle and the total number of buses with exhaust below the vehicle. In this model, when measurements were taken 2 meters from the roadway and wind speeds were approximately .5 m/s, the $PM_{2.5}$ concentration in 5-minute averages could be determined by the following equation:

$$C_{up} = 14.21 - 0.635(\#top) + 3.90(\#bottom) \qquad \text{(Eq. 8)}$$

The t-statistic for the constant was 24.35, -2.195 for the total number of buses with exhausts above, and 5.900 for the total number of buses with exhausts below. The corrected R^2 value of 0.47277 was marginal. This model showed the same negating effects that buses with exhausts above the vehicle had on upwind concentrations. A comparison of this model with the previous model indicated once again that the effects of automobiles were virtually negligible.

In summary, these models related very interesting characteristics of particulate matter near roadways. However, they were severely restricted. Both low wind speed and a constant sampling distance caused the limitations.

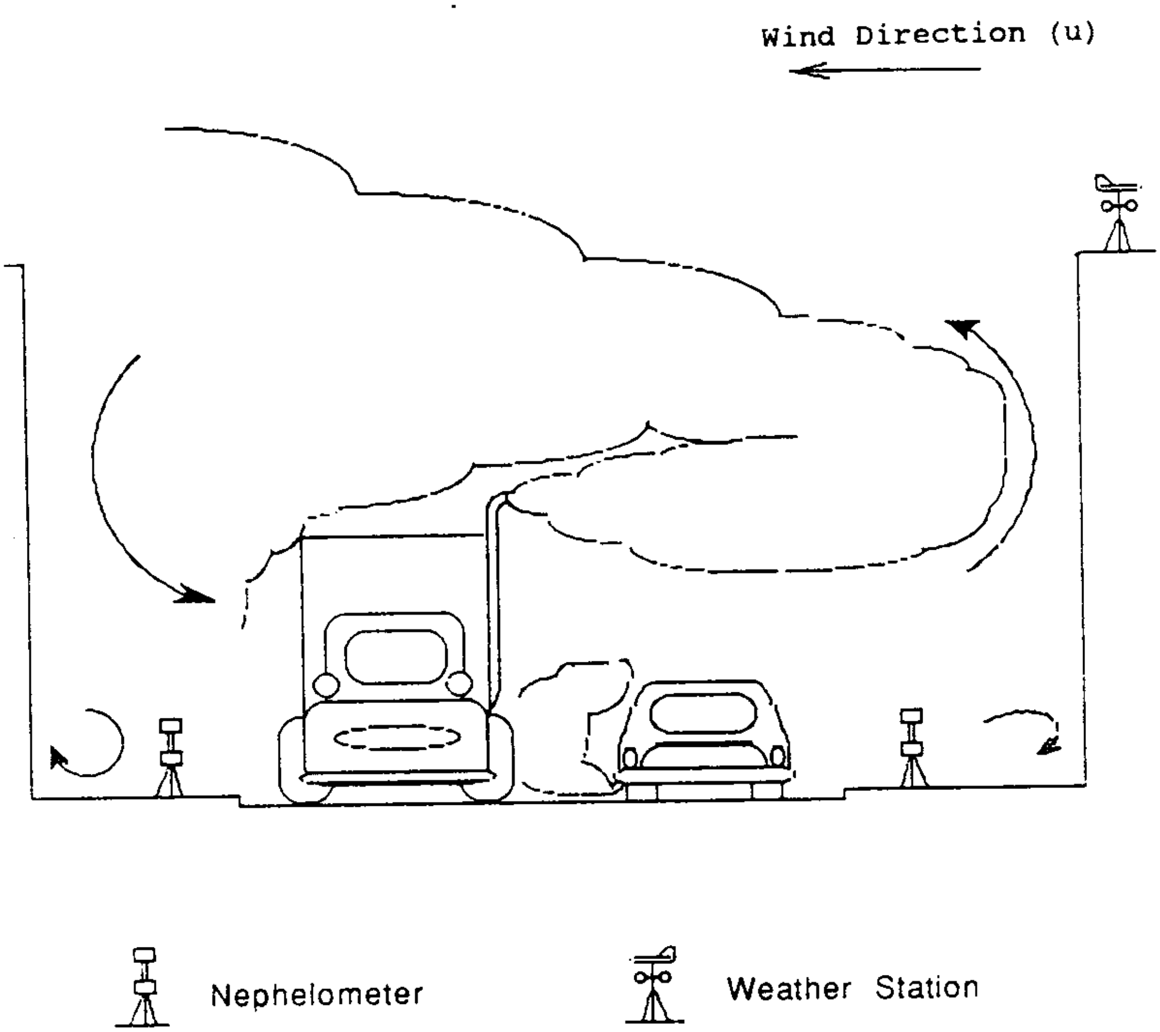


Figure 12. Street Canyon Turbulence for Exhaust Above Vehicles

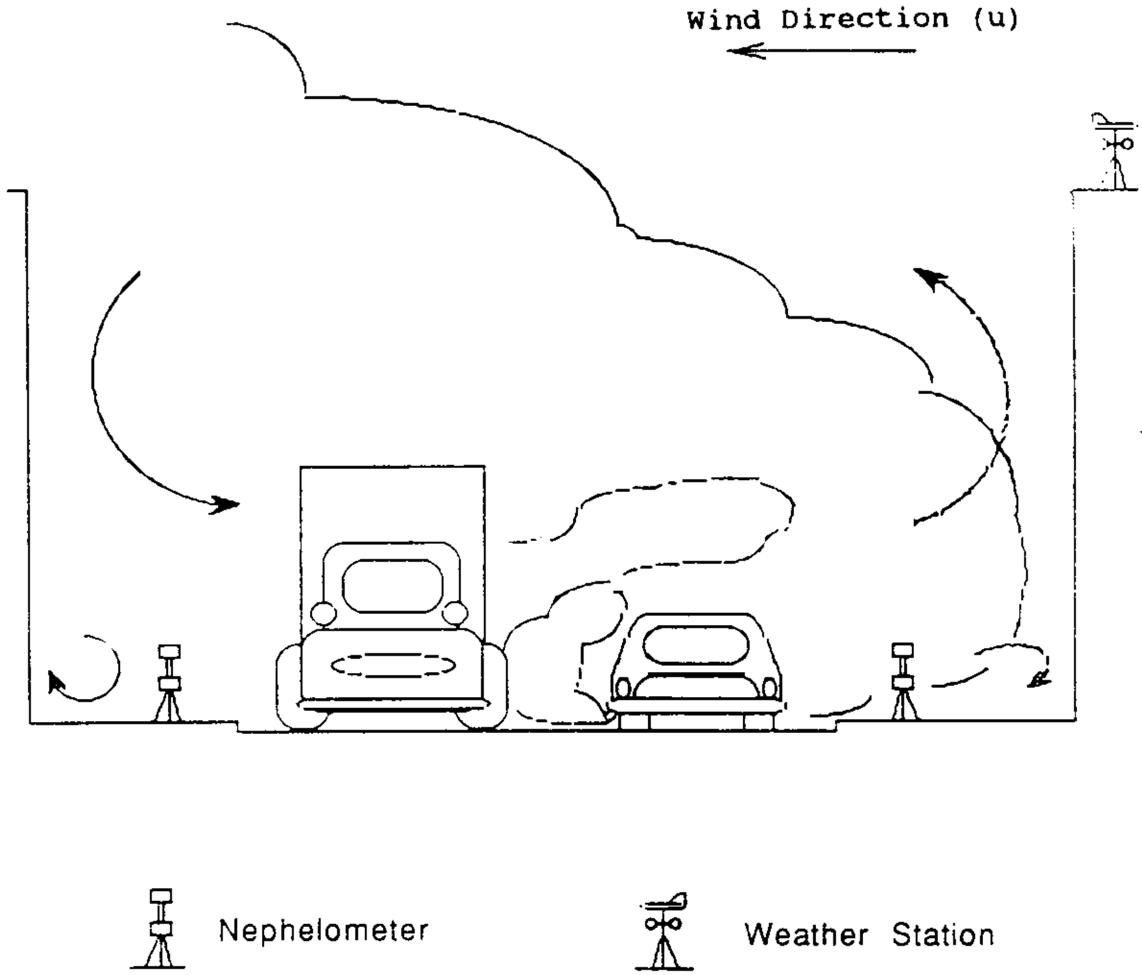


Figure 13. Street Canyon Turbulence for Exhaust Below Vehicles

THREE STAGE LEAST SQUARES MODEL

Because of the relationship between the upwind concentration and the downwind concentration indicated by the Fourier transform, a three stage least squares regression was performed. In this model, both the upwind (C_{up}) and the downwind (C_{down}) 5-minute average $PM_{2.5}$ concentrations were the dependent variables. The independent variables were

- the number of buses per 5 minutes with exhausts above the bus (#top),
- the number of buses per 5 minutes with exhausts below the bus (#bottom),
- the number of cars (#cars) per 5-minutes, C_{up},
- the inverse number of cars (inv #cars) per 5 minutes, C_{down}, and
- a constant.

In this model, when measurements are taken 2 meters from the roadway and wind speeds are approximately .5 m/s, the downwind and upwind 5-minute average $PM_{2.5}$ concentration can be determined by Equations 9 and 10, respectively.

$$C_{down} = 5.836 + 0.393(C_{up}) + 1.172(\#top)+1.852(\#bottom) + 0.0304(\#cars) \qquad \text{(Eq. 9)}$$

$$C_{up} = 4.531 + 0.812(C_{down}) + 1.766(\#bottom) + -1.928(\#top) + -22.268(\text{inv } \#cars) \qquad \text{(Eq. 10)}$$

The t-statistics for all variables in this model were greater than 2, except for "#cars" and "inv #cars." In Equation 9 the t-statistic for "#cars" was 1.1587, while in Equation 10 for "inv #cars" it was -0.428. Thus, while the effects of automobiles on the downwind side of the roadway were significant, they were not significant on the upwind side of the roadway. The t-statistic for C_{up} and C_{down}, when used as independent variables, were 3.887 and 3.905, respectively. This result reinforces the earlier statement that there is a relationship between upwind and downwind concentrations. The corrected R^2 value for Equation 9 was 0.788, the corrected R^2 value for Equation 10 was 0.487, and the corrected system R^2 was 0.687. A more in-depth review of the three stage least squares regression is given in Appendix C.

The variable "inv #cars" was used in Equation 10 because it produced a higher t-statistic than "#cars." However, the t-statistic for "inv #cars" was not statistically significant at -0.428. This was a very interesting variable because it showed that as the number of cars increased, the detrimental effects of the vehicles decreased. In other words, on the upwind side of the roadway, if only a few cars were on the road, the vehicle wake dispersed more $PM_{2.5}$ than the $PM_{2.5}$ contributed by the vehicles. On the other hand, as the number of cars increased, the $PM_{2.5}$ contributed by the vehicles increased (through the increase in C_{down} in Equation 10) at a greater rate than the increased dispersion that resulted from the greater number of cars, thus producing a relative increase in $PM_{2.5}$.

STEVENS WAY STREET CANYON MODEL

Emission factors of 0.012 g/veh/hr and 0.50 g/bus/hr were put into the street canyon model (described in Appendix D). The resulting downwind concentrations were from 0.95 to 1 times those calculated by the 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.

CONCLUSIONS

Integrating nephelometers are an excellent tool for examining particulate matter along the roadway. The nephelometers used in this study took b_{sp} measurements that resulted in the accurate calculation of particles between the ranges of 2.5 *u*m and 0.1 *u*m. 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 *u*m, it does not mean they do not exist or are insignificant. Near highways, nuclei-mode-sized aerosols, particles between 0.1 *u*m and 0.01 *u*m, can contribute an additional particulates mass equal to approximately 30 percent to 50 percent of that measured between 2.5 and 0.1*u*m. 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 minutes, into and onto particles larger than 0.1 *u*m (10). Placing integrating nephelometers close to the roadway may cause the effects of nuclei-mode-sized aerosols to be overlooked. However, this problem could indicate that integrating nephelometers placed next to roadways are better for application to diesel vehicles than gasoline vehicles.

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

Twice spikes in concentrations of 46 ug/m^3 and 60 ug/m^3 were traced to Gray Lines Tour buses with exhaust below the bus. This result indicates that these tour buses were the most polluting vehicles travelling through campus. Their emissions were often four to six times higher than those of public transit buses. The clean-up of these buses could reduce short-term exposure to pedestrians substantially.

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, modelling them as continuous sources can be erroneous. The health effects of high, short-term exposures should be studied and standards should be determined.

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, buses with exhausts below the vehicle can have roughly twice the effect on $PM_{2.5}$ that buses with exhausts above the vehicle do.

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 one to two orders of magnitude 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 of Black et al. (7) and recent emission standards.

Highways with complex terrain can have both line source and street canyon characteristics. While buildings are predominantly responsible for canyon characteristics, gaps between buildings and perpendicular roads can produce line source characteristics. These effects were seen at the Stevens Way location. 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.

ACKNOWLEDGMENTS

Special thanks is given to Mr. Quentin Hanley and Ray Weiss for help with the equipment used in this project. Without the financial support of the Washington State Department of Transportation this work could not have been performed.

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APPENDIX A

AIR POLLUTION EQUIPMENT TESTS

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AIR POLLUTION EQUIPMENT TESTS

Location: U.W. Campus/Roof of More Hall
Tester: Morgan Balogh

Date: 4/18/91	Test Number 1
Tare Weight of Filter	0.109631 grams
Weight of Filter After Test	0.109672 grams
Weight of Material on Filter	0.000041 grams
Avg. Pressure	773.8208 mm Hg
Avg. Temperature	14.99218 Deg. Cel.
Corrected Air Displacement	1.944373 Cubic Meters
Harvard Sampler Concentration	21.09 ug/m^3 8hr avg.
Miniram Concentration	40 ug/m^3 8hr avg.
Nephelometer 901.15 Conc.	14.21 ug/m^3 8hr avg.
Nephelometer 901.16 Conc.	14.00 ug/m^3 8hr avg.

Date: 4/26/91	Test Number 2
Tare Weight of Filter	0.111284 grams
Weight of Filter After Test	0.111318 grams
Weight of Material on Filter	0.000034 grams
Avg. Pressure	760.7 mm Hg
Avg. Temperature	13.9 Deg. Cel.
Corrected Air Displacement	1.896 Cubic Meters
Harvard Sampler Concentration	17.94 ug/m^3 8hr avg.
Miniram Concentration	0 ug/m^3 8hr avg.
Nephelometer 901.15 Conc.	6.54 mg/m^3 8hr avg.
Nephelometer 901.16 Conc.	5.19 mg/m^3 8hr avg.

Date: 4/30/91	Test Number 3
Tare Weight of Filter	0.109298 grams
Weight of Filter After Test	0.109334 grams
Weight of Material on Filter	0.000036 grams
Avg. Pressure	758.9 mm Hg
Avg. Temperature	23.6 Deg. Cel.
Corrected Air Displacement	1.904 Cubic Meters
Harvard Sampler Concentration	18.91 ug/m^3 8hr avg.
Miniram Concentration	0 ug/m^3 8hr avg.
Nephelometer 901.15 Conc.	9.13 ug/m^3 8hr avg.
Nephelometer 901.16 Conc.	7.15 ug/m^3 8hr avg.

Date: 5/4/91	Test Number 4	
	2.5 ug/m^3	>10 ug/m^3
Tare Weight of Filter	0.112377 grams	0.113617 grams
Weight of Filter After Test	0.112404 grams	0.113679 grams
Weight of Material on Filter	0.000027 grams	0.000062 grams
Avg. Pressure	765.3 mm Hg	
Avg. Temperature	23.4 °C	
Corrected Air Displacement	1.835m^3	1.849 m^3
Harvard Sampler Conc.	14.71 ug/m^3	33.79 ug/m^3
Miniram Concentration	30 ug/m^3 8hr avg.	
Nephelometer 901.15 Conc.	15.76 ug/m^3 8hr avg.	
Nephelometer 901.16 Conc.	16.32 ug/m^3 8hr avg.	

Date: 5/15/91	Test Number 5
Tare Weight of Filter	0.110585 grams
Weight of Filter After Test	0.110614 grams
Weight of Material on Filter	0.000029 grams
Avg. Pressure	763.0 mm Hg
Avg. Temperature	21.6 Deg. Cel.
Corrected Air Displacement	1.775 Cubic Meters
Harvard Sampler Concentration	16.34 ug/m^3 8hr avg.
Miniram Concentration	40 ug/m^3 8hr avg.
Nephelometer 901.15 Conc.	8.91 ug/m^3 8hr avg.
Nephelometer 901.16 Conc.	N/A ug/m^3 8hr avg.

Date: 5/16/91	Test Number 6
Tare Weight of Filter	0.112773 grams
Weight of Filter After Test	0.112807 grams
Weight of Material on Filter	0.000034 grams
Avg. Pressure	762.2 mm Hg
Avg. Temperature	15.9 Deg. Cel.
Corrected Air Displacement	1.668 Cubic Meters
Harvard Sampler Concentration	20.56 ug/m^3 8hr avg.
Miniram Concentration	50 ug/m^3 8hr avg.
Nephelometer 901.15 Conc.	8.12 ug/m^3 8hr avg.
Nephelometer 901.16 Conc.	N/A ug/m^3 8hr avg.

APPENDIX B

TIME/CONCENTRATION FIGURES

APPENDIX B

TIME/CONCENTRATION FIGURES

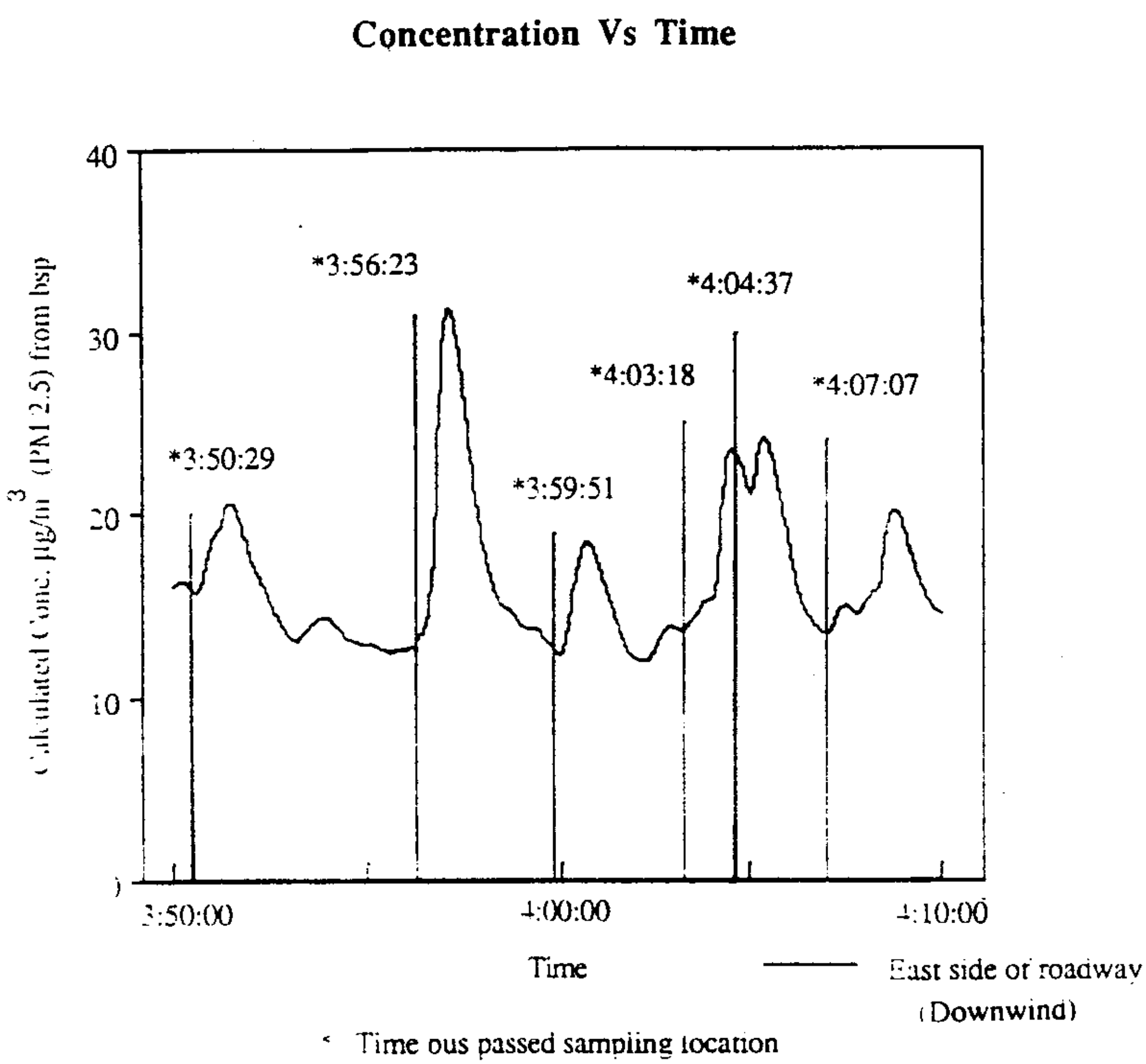
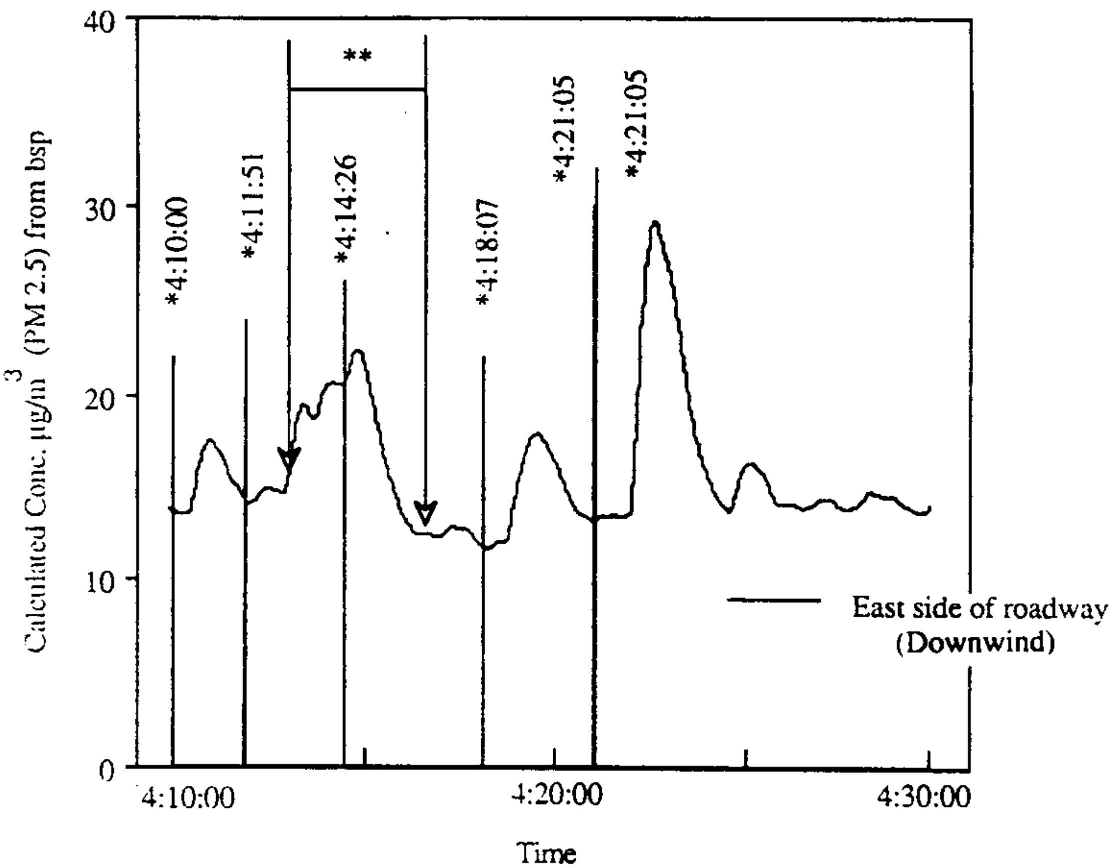


Figure B1. $PM_{2.5}$ Graph, 7/11/91, 3:50 PM to 4:10 PM

Concentration Vs Time



* Time bus passed sampling location

** Period from 4:12:50 to 4:16:20 where traffic was backed up through sampling location

Figure B2. $PM_{2.5}$ Graph, 7/11/91, 4:10 PM to 4:29 PM

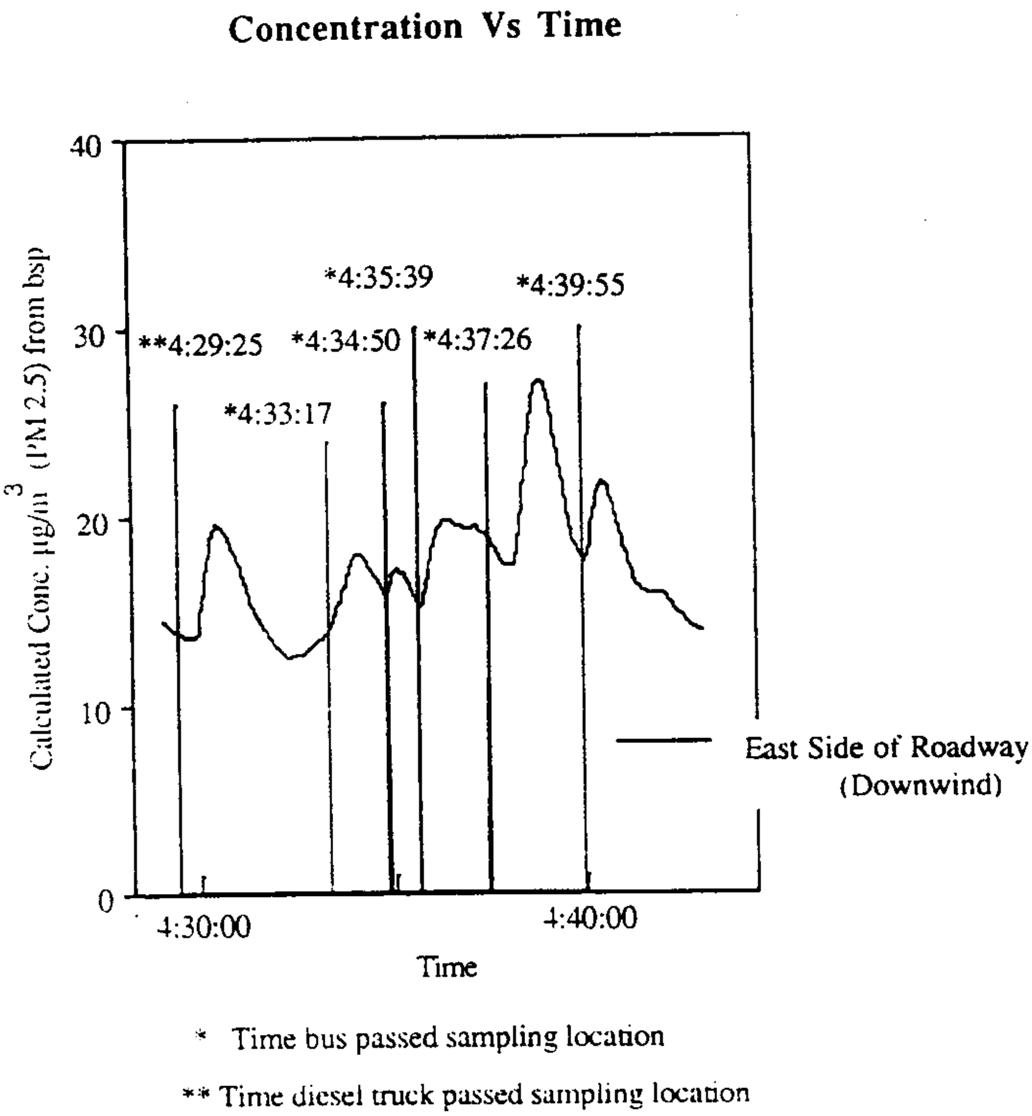
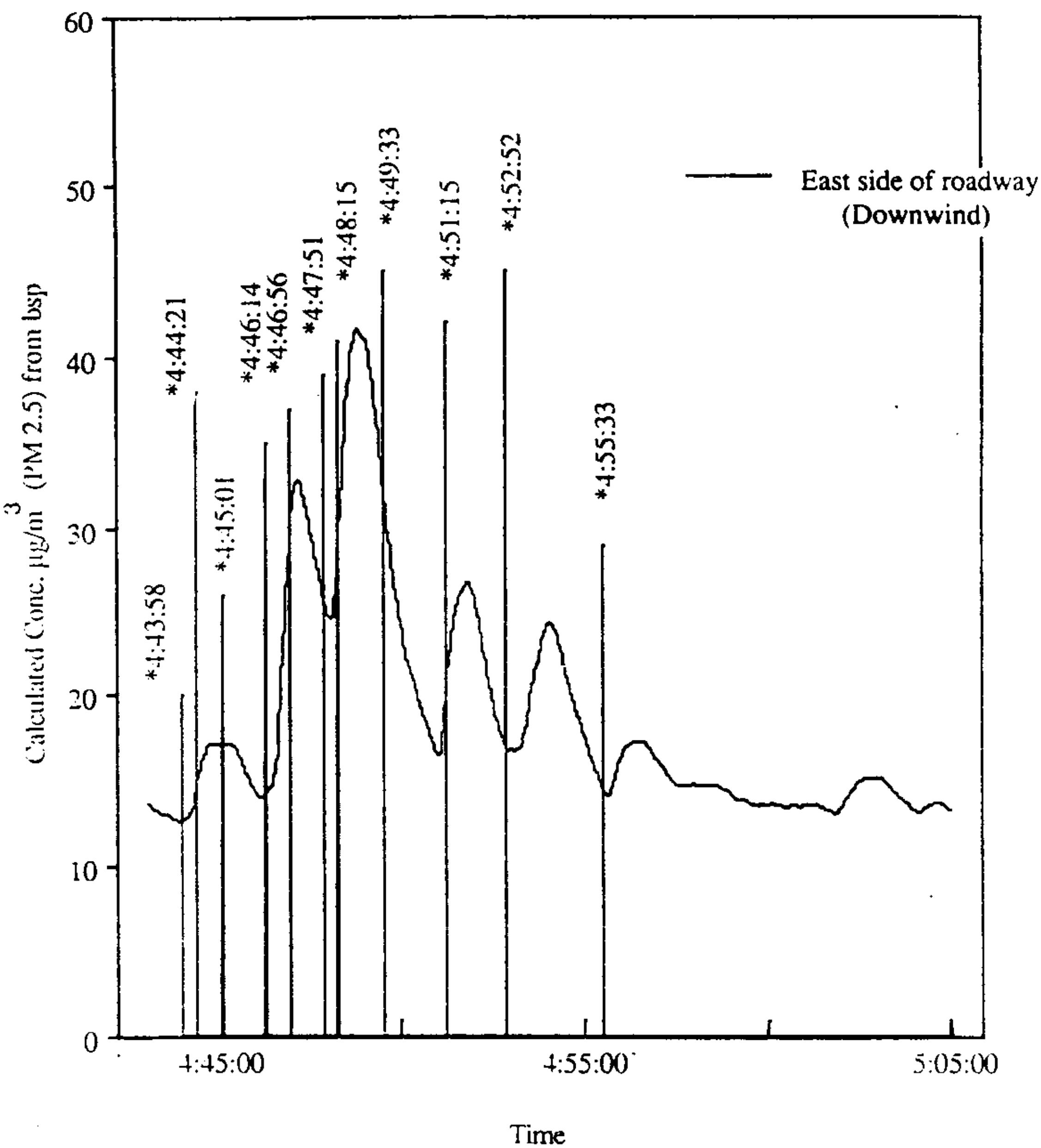


Figure B3. $PM_{2.5}$ Graph, 7/11/91, 4:29 PM to 4:43 PM

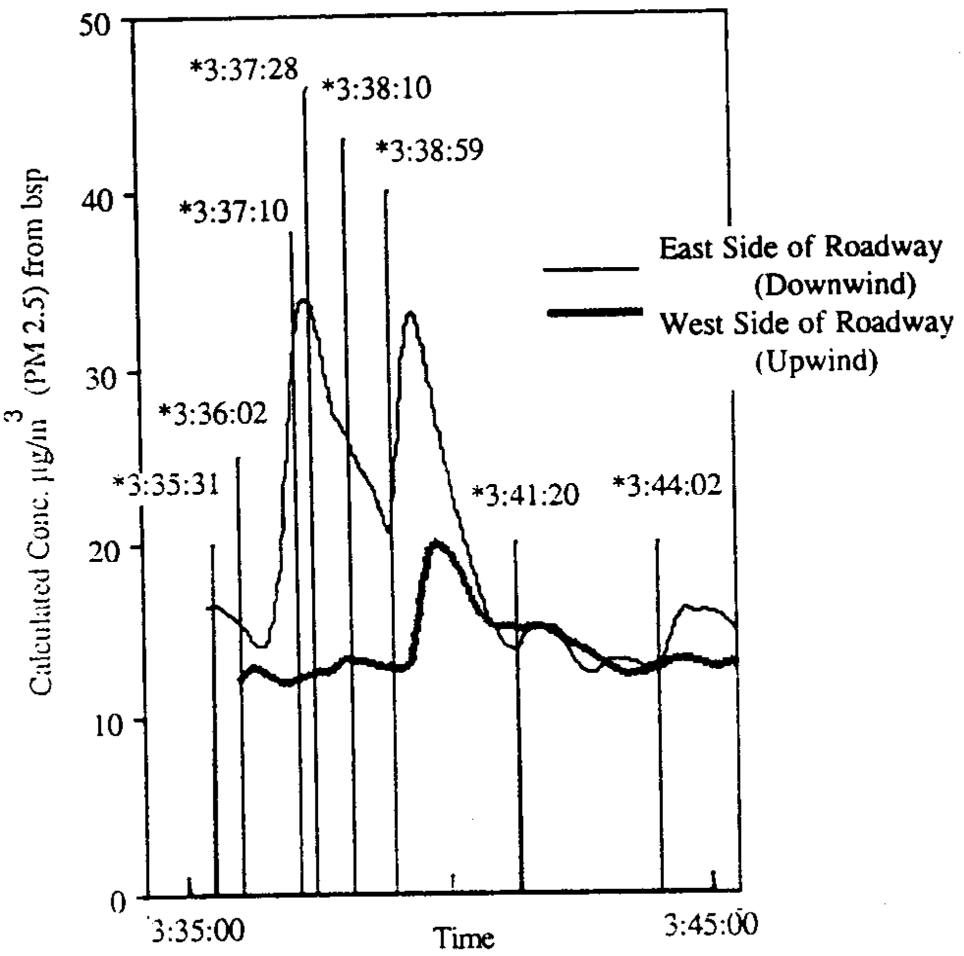
Concentration Vs Time



* Time bus passed sampling location

Figure B4. $PM_{2.5}$ Graph, 7/11/91, 4:43 PM to 5:05 PM

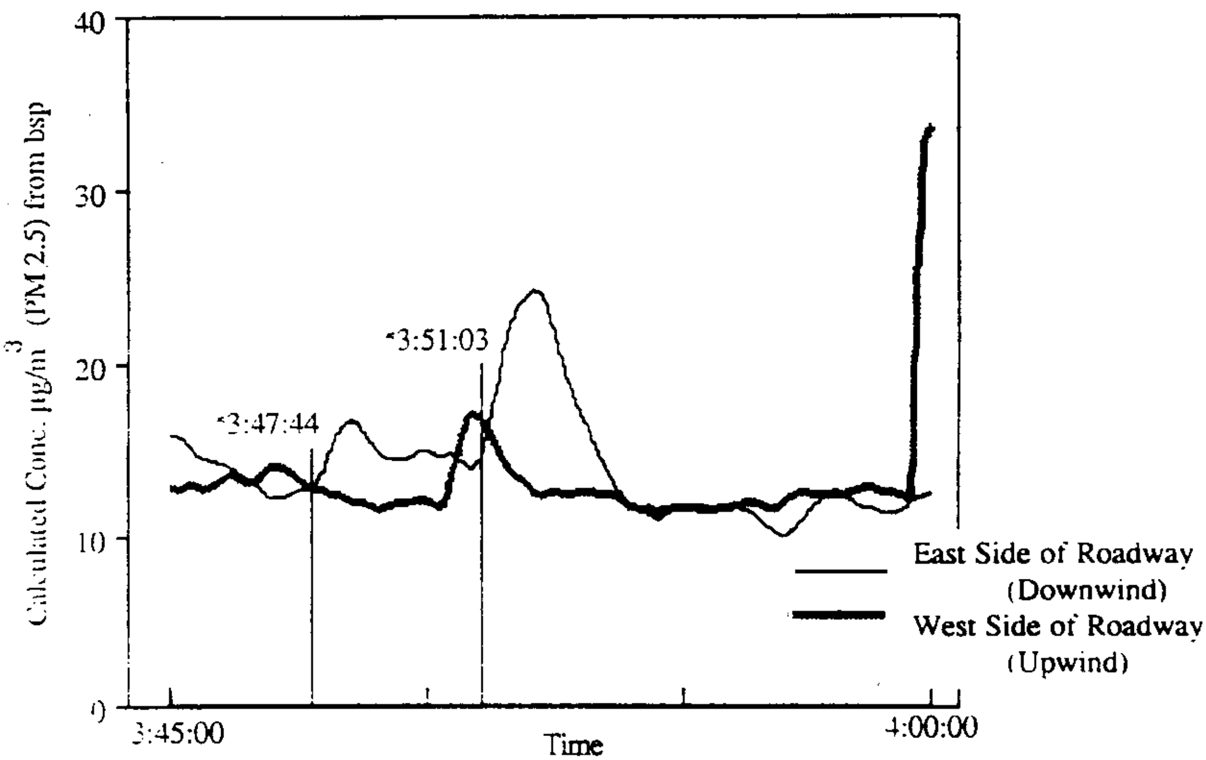
Concentration Vs Time



* Time bus passed sampling location

Figure B5. $PM_{2.5}$ Graph, 7/29/91, 3:35 PM to 3:45 PM

Concentration Vs Time



* Time bus passed sampling location

Figure B6. $PM_{2.5}$ Graph, 7/29/91

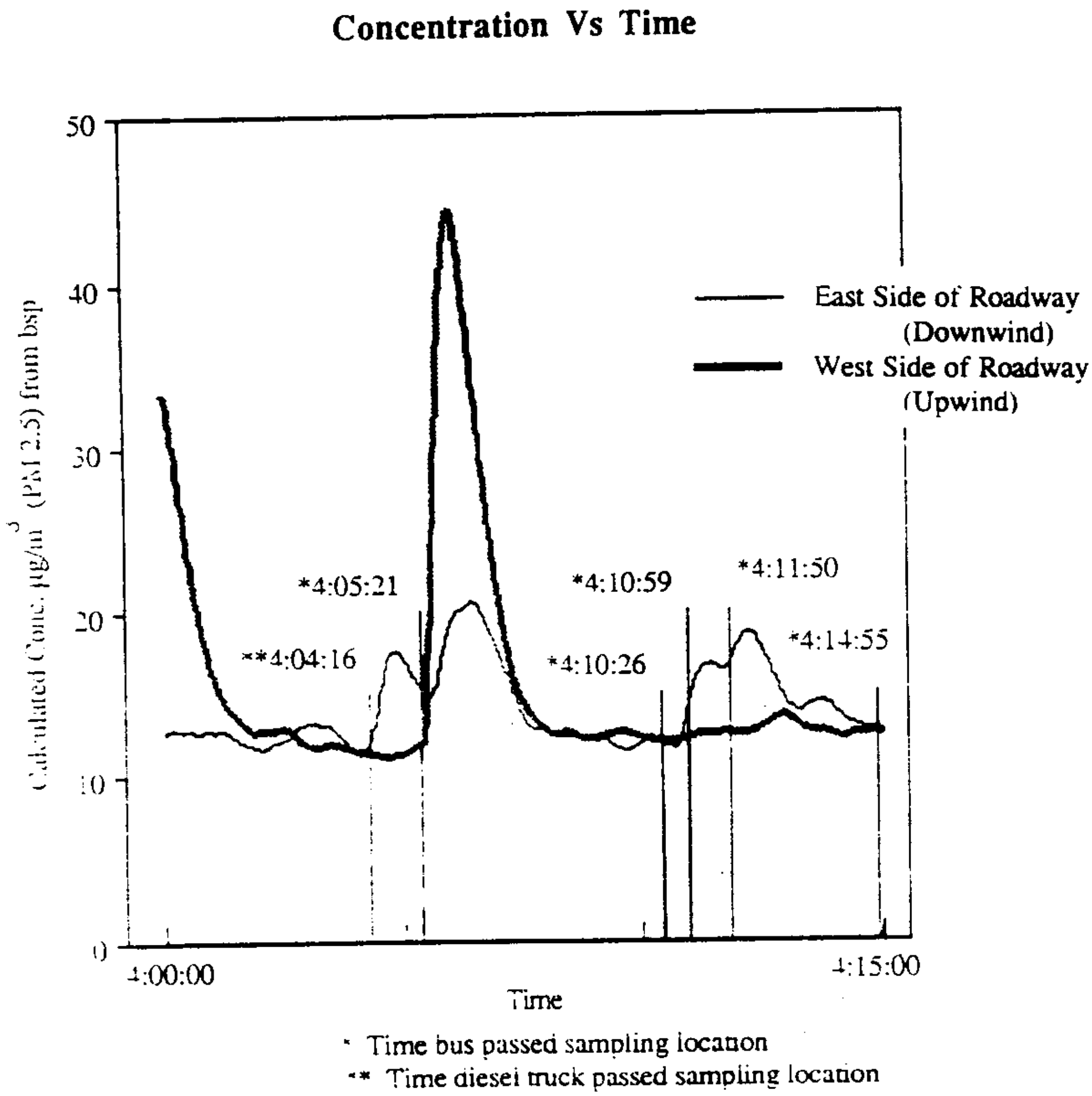
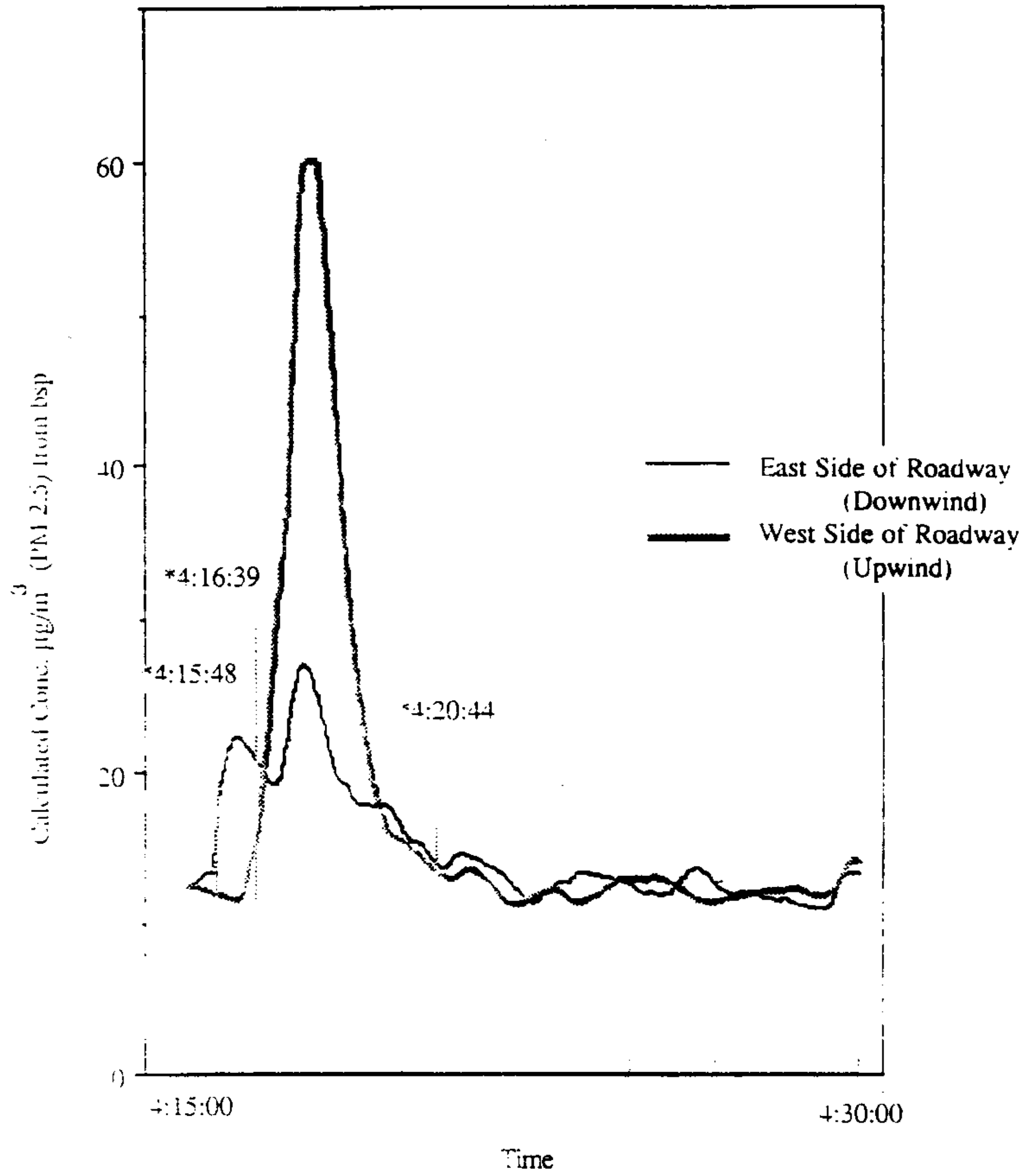


Figure B7. $PM_{2.5}$ Graph, 7/29/91, 4:00 PM to 4:15 PM

Concentration Vs Time



* Time bus passed sampling location

Figure B8. $PM_{2.5}$ Graph, 7/29/91, 4:15 PM to 4:30 PM

Concentration Vs Time

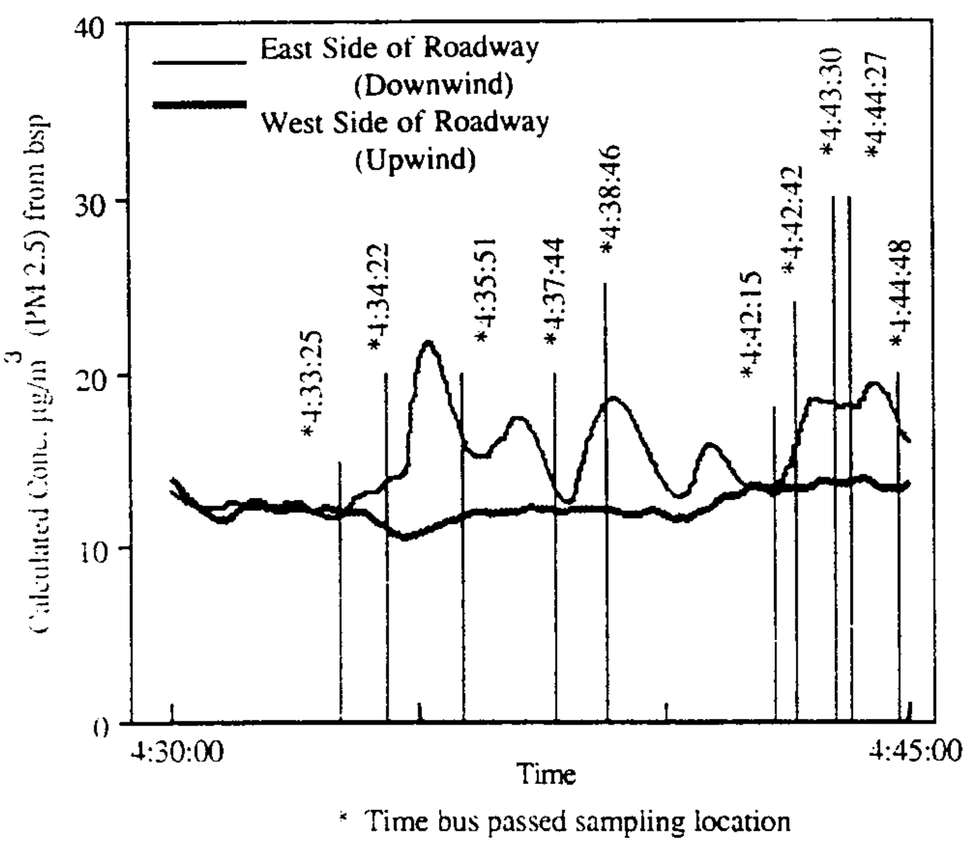


Figure B9. $PM_{2.5}$ Graph, 7/29/91, 4:30 PM to 4:45 PM

Concentration Vs Time

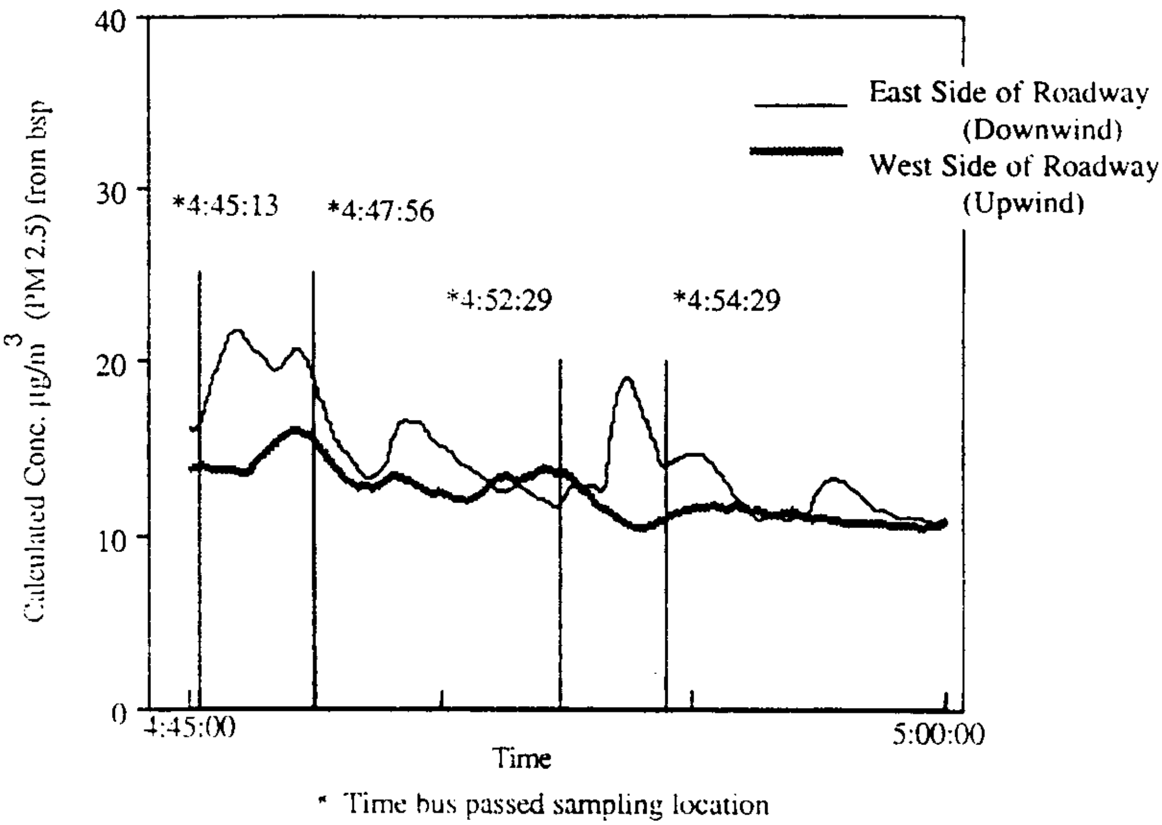


Figure B10. $PM_{2.5}$ Graph, 7/29/91, 4:45 PM to 5:00 PM

APPENDIX C

RESULTS OF REGRESSION ANALYSIS

APPENDIX C

RESULTS OF REGRESSION ANALYSIS

The downwind 5 minute $PM_{2.5}$ concentration was regressed with the total number of diesels (i.e., buses) per 5 minute interval, and the total number of cars per 5 minute interval. The following results were obtained.

ORDINARY LEAST SQUARES ESTIMATION

Dependent Variable: measured downwind $PM_{2.5}$ concentration calculated from bsp.

Independent Variable	Estimated Coefficient	Standard Error	t-Statistic
#buses	1.849	0.167	11.042
#cars	0.04505	0.03037	1.483
constant	10.77	1.278	8.425

Number of Observations	38
R-squared	0.77763
Corrected R-squared	0.76493
Sum of Squared Residuals	79.03185
Standard Error of the Regression	1.50268
Durbin-Watson Statistic	1.89647
Mean of Dependent Variable	16.26811

The downwind 5 minute $PM_{2.5}$ concentration was regressed with the total number of diesels (i.e., buses) per 5 minute interval. The following results were obtained.

ORDINARY LEAST SQUARES ESTIMATION

Dependent Variable: measured downwind $PM_{2.5}$ concentration calculated from bsp.

Independent Variable	Estimated Coefficient	Standard Error	t-Statistic
#buses	1.831	0.170	10.78
constant	12.56	0.424	29.62

Number of Observations	38
R-squared	0.76365
Corrected R-squared	0.75709
Sum of Squared Residuals	84.00078
Standard Error of the Regression	1.52753
Durbin-Watson Statistic	1.65258
Mean of Dependent Variable	16.26811

The downwind 5 minute $PM_{2.5}$ concentration was regressed with the total number of buses with exhausts below the bus, the total number of buses with exhausts above the bus, and the total number of cars.

ORDINARY LEAST SQUARES ESTIMATION

Dependent Variable: measured downwind $PM_{2.5}$ concentration calculated from bsp.

Independent Variable	Estimated Coefficient	Standard Error	t-Statistic
#bottom	2.73747	0.45577	6.00629
#top	1.60366	0.19854	8.07724
#cars	0.049991	0.029113	1.71710
constant	10.65253	1.22219	8.71596

Number of Observations	38
R-squared	0.80279
Corrected R-squared	0.78539
Sum of Squared Residuals	70.08921
Standard Error of the Regression	1.43577
Durbin-Watson Statistic	1.44937
Mean of Dependent Variable	16.26811

The upwind 5 minute $PM_{2.5}$ concentration was regressed with the total number of buses with exhausts above the vehicle, the total number of buses with exhausts below the vehicle, and the inverse number of cars. The following results were obtained.

ORDINARY LEAST SQUARES ESTIMATION

Dependent Variable: measured upwind $PM_{2.5}$ concentration calculated from bsp.

Independent Variable	Estimated Coefficient	Standard Error	t-Statistic
#top	-0.615	0.287	-2.14
#bottom	3.978	0.658	6.04
inv car	-0.614	0.287	-2.140
constant	16.08	1.589	10.11

Number of Observations	38
R-squared	0.52355
Corrected R-squared	0.48151
Sum of Squared Residuals	1.46363e+002
Standard Error of the Regression	2.07480
Durbin-Watson Statistic	1.77677
Mean of Dependent Variable	14.59396

The upwind 5 minute $PM_{2.5}$ concentration was regressed with the total number of buses with exhausts above the vehicle, and the total number of buses with exhausts below the vehicle. The following results were obtained.

ORDINARY LEAST SQUARES ESTIMATION

Dependent Variable: measured upwind $PM_{2.5}$ concentration calculated from bsp.

Independent Variable	Estimated Coefficient	Standard Error	t-Statistic
#top	-0.635	0.289	-2.195
#bottom	3.899	0.660	5.900
constant	14.211	0.583	24.35

Number of Observations	38
R-squared	0.50127
Corrected R-squared	0.47277
Sum of Squared Residuals	1.53207e+002
Standard Error of the Regression	2.09221
Durbin-Watson Statistic	1.58325
Mean of Dependent Variable	14.59396

The upwind 5 minute $PM_{2.5}$ concentration was regressed with the total number of buses, the total number of buses with exhausts below the vehicle, and the inverse of the total number of cars. The following results were obtained.

ORDINARY LEAST SQUARES ESTIMATION

Dependent Variable: measured upwind $PM_{2.5}$ concentration calculated from bsp.

Independent Variable	Estimated Coefficient	Standard Error	t-Statistic
#top	-0.615	0.287	-2.14
#bottom	4.59	1.58	10.12
inv car	-71.84	56.97	-1.261
constant	16.08	0.785	5.847

Number of Observations	38
R-squared	0.52355
Corrected R-squared	0.48151
Sum of Squared Residuals	0.0146
Standard Error of the Regression	2.07480
Durbin-Watson Statistic	1.77677
Mean of Dependent Variable	14.59396

The upwind 5 minute $PM_{2.5}$ concentration was regressed with the total number of buses and the total number of buses with exhausts below the vehicle. The following results were obtained.

ORDINARY LEAST SQUARES ESTIMATION

Dependent Variable: measured upwind $PM_{2.5}$ concentration calculated from bsp.

Independent Variable	Estimated Coefficient	Standard Error	t-Statistic
#top	-0.635	0.289	-2.20
#bottom	4.53	0.791	5.73
constant	14.2	0.584	24.4

Number of Observations	38
R-squared	0.50127
Corrected R-squared	0.47277
Sum of Squared Residuals	1.53207e+002
Standard Error of the Regression	2.09221
Durbin-Watson Statistic	1.58325
Mean of Dependent Variable	14.59396

THREE STAGE LEAST SQUARES ESTIMATION

Dependent Variable: measured downwind $PM_{2.5}$ concentration calculated from bsp.

Independent Variable	Estimated Coefficient	Standard Error	t-Statistic
Downwind Concentration	0.393	0.101	3.905
# top	1.172	0.571	2.054
# bottom	1.852	0.189	9.811
inv cars	0.0304	0.0263	1.1587
Constant	5.836	1.676	3.482

R-squared	0.81101
Corrected R-squared	0.78810
Sum of Squared Residuals	67.17048
Standard Error of the Regression	2.03547
Mean of Dependent Variable	16.26811

Dependent Variable: measured upwind $PM_{2.5}$ concentration calculated from bsp.

Independent Variable	Estimated Coefficient	Standard Error	t-Statistic
Upwind Concentration	0.812	0.209	3.887
# top	-1.1928	0.818	-4.545
# bottom	1.766	0.424	2.159
inv cars	-22.268	51.973	-0.428
Constant	4.531	3.241	1.398

R-squared	0.54285
Corrected R-squared	0.48743
Sum of Squared Residuals	1.40435e+002
Standard Error of the Regression	4.25561
Mean of Dependent Variable	14.59396
Number of observations	38
System R-squared	0.68668

APPENDIX D

LITERATURE REVIEW

APPENDIX D

LITERATURE REVIEW

MODELS

Since the 1970s, several highway dispersion models have been developed. The majority of these models utilize the Gaussian dispersion equation, modified for different highway situations. The models discussed in this literature review include the Gaussian line source model, wake theory, box model, street canyon model, and intersection model (see Table D1 for model usage).

Many approaches are used to model the diffusion of air pollution. There are two basic ways to characterize how emissions are released. One approach is to treat the emission as a continuous release of pollution over a period of time. The other approach is to treat the emissions as a series of instantaneous releases, or puffs. Because the worst emissions accompany high traffic volumes, in highway models emissions are assumed to be continuously released. Three basic shapes are related to the emission source. First, point sources, such as smoke stacks, have small release areas and high concentrations at release. Second, area sources, such as landfills and parking lots, have large release areas and low concentrations. Finally, line sources, such as highways, are a string of point sources that are much longer than they are wide.

Table D1. Models

Type of Model	Use	Accuracy	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	low	some difficulty

Gaussian Line Source Model

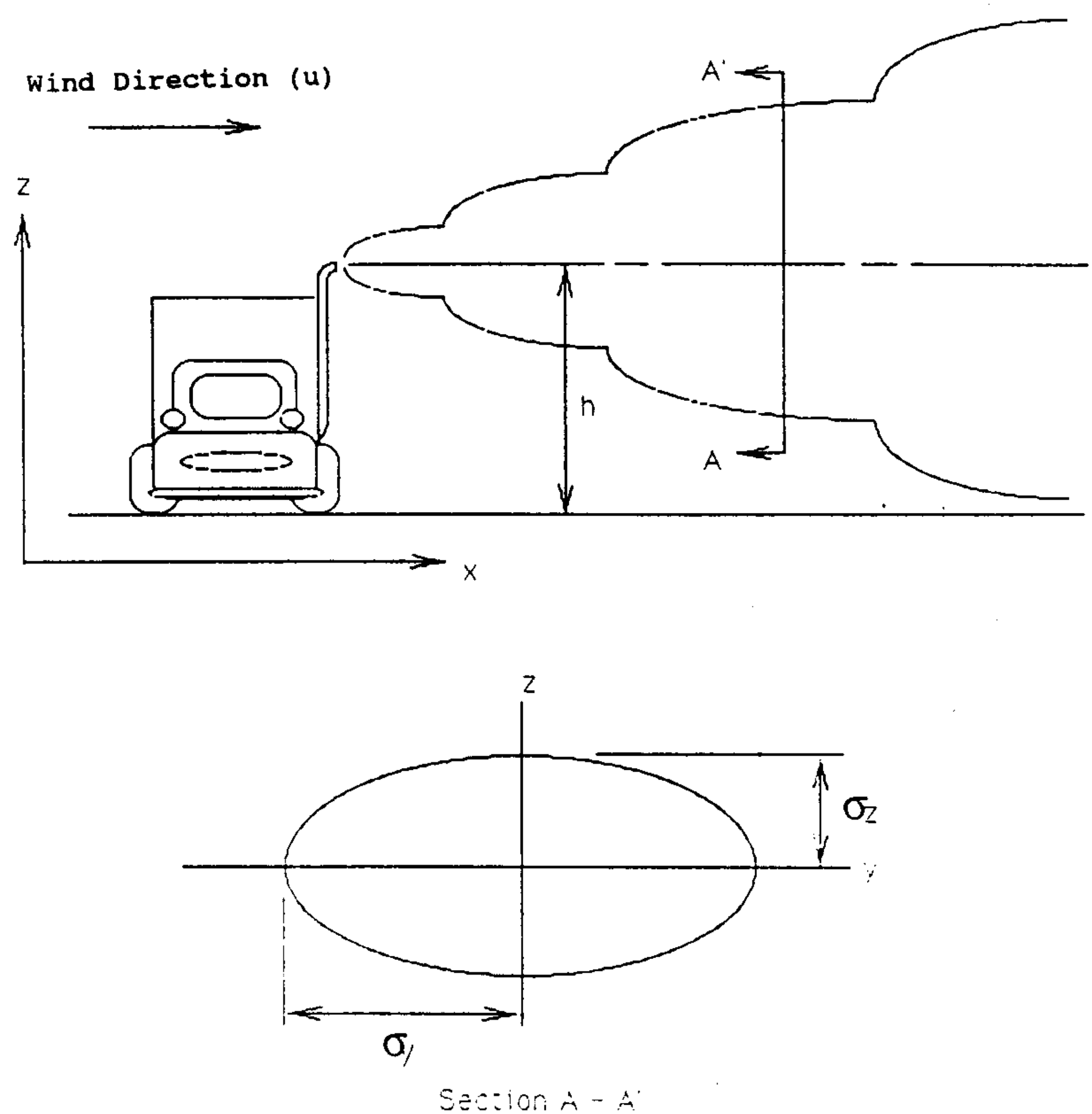
Atmospheric flows, or advection, strongly affect the concentration of air pollution. Advection is caused by the movement of air (warm air rising and cold air descending) and the drag of the earth's surface. The atmospheric boundary layer is the layer of the atmosphere where advection takes place. When particulate matter is released into the atmosphere it usually is trapped in the atmospheric boundary layer, where it is mixed with surrounding air (i.e., diluted) through air turbulence. When the atmospheric boundary layer is stable, little advection occurs and thus little mixing of air pollution. However, when the atmospheric boundary layer is unstable, advection is strong, and air pollution is substantially mixed.

Turbulence is in fact three-dimensional; therefore, the change in diffusion is a function of the turbulence in the horizontal (Ky), vertical (Kz), and downwind direction (Kx). Fick's law of turbulent diffusion sets the turbulence in the x, y, and z directions equal to a constant. This assumption allows for much simpler mathematical modelling of dispersion.

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 (does not vary in space), stationarity (does not vary in time) and a large diffusion time (1). Equation 1 is a Gaussian model for a continuous line source. Figure D1 illustrates some general concepts of the Gaussian model.

$$\frac{C}{Q} = \frac{1}{2\,\mathrm{pi}\ \mathrm{sigma}_y\ \mathrm{sigma}_z\ u} \exp -y^2/2\,\mathrm{sigma}^2{}_y \qquad \text{(Eq. 1)}$$

$$X[\exp - (z-h)^2/2\mathrm{sigma}^2{}_z + \exp - (z+h)^2/2\mathrm{sigma}^2{}_z]$$



u is the wind speed.
h is the effective height
σ_y & σ_z are standard deviations of the
crosswind concentration distributions.

Figure D1. Gaussian Concepts

where C = the concentration in micrograms per cubic meter (ug/m^3).

Q = the emission rate in micrograms per second (ug/s).

h = the effective height of emission release in meters (m).

u = the wind speed (m/s).

y = horizontal direction at right angles to the plume axis, with y equal to zero on the axis.

z = the height above the ground (m).

$sigma_y$, $sigma_z$ = standard deviations of the distribution C in the y and z axis, respectively (m).

Most models currently in use 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 (2).

$$\frac{C}{Q} = \frac{1}{sigma_z\, u}\left[\exp - (z-h)^2/2sigma^2_z + \exp - (z+h)^2/2sigma^2_z\right] \qquad \text{(Eq. 2)}$$

where C = the concentration (ug/m^3).

Q = the emission rate (ug/s).

h = the effective height of emission release (m).

u = the wind speed (m/s).

z = the height above the ground (m).

$sigma_z$ = a standard deviation of the distribution C in the z axis (m).

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 (3). However, because the models are on a microscale and a high percentage of particulate matter emissions are less than 2 um, there is little reason not to use these models. When the early models were tested with gaseous tracers, they proved accurate when the wind was perpendicular to the roadway and the atmospheric boundary layer was near neutral stability. Neutral stability occurs when the air temperature rises 1° for each 100-m rise in height. However, when winds were nearly parallel to the roadway, the concentrations predicted by the models were higher than the actual measured concentrations (4).

Wake Theory

Wind speed and sigma$_Z$ (a representation of the vertical dispersion of the pollution plume) inversely affect air pollution concentration (see Figure D1). The main reason that Gaussian models often over-predicted concentrations is that the parameters sigma$_Z$ and wind speed were not sensitive to the effects of vehicle turbulence. In the General Motors Sulfate Dispersion experiment, sigma$_Z$ was calculated for several downwind distances from the center of the roadway. The experiment found little variation in sigma$_Z$, despite changes in the stability of the atmospheric boundary layer. From this the researchers concluded that vehicle induced turbulent mixing near roadways actually dominates boundary layer mixing (5).

Chock (6) used the General Motors Sulfate Dispersion data to develop relationships between sigma$_Z$ and wind speed to vehicle wake turbulence. By modifying sigma$_Z$ and wind speed, one can account for vehicle wake turbulence and buoyancy effects from the heated exhaust.

Other, more simplistic approaches have been taken to account for vehicle wake turbulence. Often only one variable in the Gaussian model has been modified. For an example, see the street canyon model later in this appendix. In this model, a factor of 0.5 m/s, has been added to the wind speed to account for vehicle wake turbulence.

Often only sigma$_Z$ has been modified to account for vehicle wake turbulence. This has usually been done by artificially moving the existing emission source upwind so that diffusion is increased at the actual emission point. This creates an initial value for sigma$_Z$ at the roadway. This sigma$_Z$ accounts for the increased diffusion turbulence that results from vehicle wakes. The typical value for sigma$_Z$ is generally the height of the vehicles passing the sample location, or approximately 1.5 m (5). Rao and Keenam (7) showed that $sigma_Z$ could be calculated by adding Brigg's (8) rural estimation of $sigma_Z$ for weakly unstable atmospheric conditions to the initial $sigma_Z$ (1.5) for up to 30 m from the roadway, as defined by Equation 3.

$$1.5 + 0.8x(1 + .0002x)^{-0.5} \qquad \text{(Eq. 3)}$$

where x = the downwind distance form the source to the receptor.

This is a simplistic way to account for the wake effects of vehicles, but because all sampling performed in this study occurred within 3 m from the roadway, and the atmospheric stability was weakly unstable or unstable, it was satisfactory. When these $sigma_Z$ modifications are used, it is important to remember that they were developed with vehicle speeds of 80 km/hr and a vehicle distribution that is not typical of highway traffic.

Research by Eskridge and Hunt (9) that utilized the General Motors Sulfate Dispersion Experiment data led to the development of a finite-difference model that incorporates vehicle wake theory. Wake theory was derived from a perturbation solution to the equations of motion. This model was developed to represent the fluctuation in the wind velocity caused when the vehicle wake passes the observation point. These deficits can be summed to account for several vehicles. The wind deficit allows for the estimation of diffusion constants (Ky, Kz, Kx).

Since its development, Eskridge has modified his model through the use of wind tunnel experiments and tracer gas profiles. However, because of the study conditions used to develop this model, vehicle speed must be much greater than wind speed. This condition is easy to meet when dispersion near highways is modelled.

Emission Factors

To use Gaussian dispersion models, an emission rate for the roadway must be computed. To develop the emission rate, statistical and empirical approaches are typically utilized. In 1984 a model was developed for the Environmental Protection

Agency that estimated the particulate emission factor for paved urban roads. Through multiple linear regression, the following empirical expression was obtained (10):

$$e = k\,(sL/.5)^P \text{ grams per vehicle kilometer traveled (g/veh/km)} \quad \text{(Eq. 4)}$$

$$e = k\,(sL/.7)^P \text{ pounds per vehicle mile traveled (lb/veh/km)} \quad \text{(Eq. 5)}$$

where e = the particulate emission factor, g/VKT (lb/VMT).

L = total road surface dust loading, g/m^2 ($grains/ft^2$).

s = surface silt content, fraction of particles <= 75 microns in diameter.

k = base emission factor, g/VKT (lb/VMT).

p = exponent estimable by ordinary least squares (dimensionless).

For this model, the silt loadings (sL) can be measured, calculated, or taken from various charts that are broken down by roadway category. Table D2 shows the base emission factor (k) and exponent (p) defined for various particle sizes (11).

AP-42 was an E. P. A. document that reported data on emissions of atmospheric pollutants (12). Several vehicle emission factors pertaining to particulate matter were given in this report. Information on vehicle lead emissions, and bus and truck PM_{10} emission factors were given. Because these emission factors were based on averages, caution is recommended in their use.

Similarities in inertia weight and engine type has led to the grouping of heavy duty diesels and transit buses in the past. But because transit buses have very low average speeds, make frequent stops, and have higher acceleration and deceleration rates, buses should be examined separately. Transit buses typically operate in heavily populated urban corridors, so that public exposure to bus emissions is relatively high. Because these issues are of major concern, the E. P. A. is supporting an ongoing analysis of transit bus emissions. Although the study is not complete, AP-42 gave average lifetime PM_{10} emission factors for various types of bus engines. These engines were built from 1982 to 1984. The Detroit Diesel Allison engines DDA 6V-71N and DDA

Table D2. Base Emission Factor and Exponent (E. P. A., 1988)

Particle Size Fraction	k		p
	g/VKT*	(lb/VMT)**	
TSP	5.87	(0.0208)	0.9
PM_{15}	2.54	(0.0090)	0.8
PM_{10}	2.28	(0.0081)	0.8
$PM_{2.5}$	1.02	(0.0036)	0.6

* grams per vehicle kilometer travelled
** pounds per vehicle mile travelled

8V-71N had emission factors of 6.27 grams per mile (g/mile). DDA 6V-92TA had an emission factor of 4.77 g/mile. Other engines had an average emission factor of 5.52 g/mile. At this time there are no speed or temperature correction factors. The PM_{10} emissions of heavy duty diesels (trucks) vary but are generally about 2 g/mile.

Black et al. (13), using a literature survey (14) and some simplifying assumptions, developed particulate emission factors for motor vehicles. The general approach of this study was to use emission standards as emission factors along with laboratory measured emission factors. The emission factors from Black's study were given for gasoline vehicles. In 1986 diesel passenger cars were regulated, and in 1988 diesel trucks were regulated. These emission standards are assumed to be equivalent to the actual emission rates. Black's forecasted emissions factors for light duty, gasoline powered vehicles (i.e., cars and trucks of less than 3 tons gross vehicle weight) are stated in Table D3.

Table D3. PM_{10} Emission Factors for Gasoline Vehicles (Black et al., 1985)

Model Year of Vehicle	pre - 1975	1976 to 1980	1981 to present
factor (g/veh/km)	0.03	0.02	0.01

The diesel car particulate matter emissions standard for 1986 is 0.38 g/veh/km and for 1987 through 1991 is 0.13. Light-duty diesel trucks are trucks with gross vehicle weights of up to 4.25 tons. Although light duty diesel trucks have not been regulated in the past, their emission rates should be between that of cars and heavy duty trucks, or about 0.40 g/veh/km for 1986 and 0.20 g/veh/km for 1987 through 1991. Heavy duty diesels are trucks and buses with gross vehicle weights greater than 4.25 tons. Pre-standard emission rates can be estimated at 1.0 g/veh/km for the years 1980 through 1985 and 0.9 for 1986. Heavy duty diesel emission standards were 0.38 g/veh/km for 1987 and 0.16 g/veh/km for 1988 through 1991.

Box Model

Most study areas contain many individual emission sources. Consequently, modelling each source is often too complex. A box model defines the area to be studied and combines all sources. The basic assumptions in a box model are that the emissions and wind speed over a given distance are constant, and the atmosphere over the modelling region is a well-mixed box. With several assumptions, the typical box model equation is derived as follows (15):

$$C = \frac{\text{delta x } Q_a}{z_i \, u} \qquad \text{(Eq. 6)}$$

where C = the concentration (ug/m^3).

Delta x = the downwind distance (m).

Q_a = the emission rate (ug/s).

z_i = the is the distance between the ground and the mixing height (m).

U = the horizontal wind speed (m/s).

This is a very useful model because it is simple to use, the cost of data collection is minimal, and results are fairly accurate. It is also useful to determine whether concentrations in an area are high enough to justify the money for further study.

Street Canyon Models

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 modelling is necessary. Most successful street canyon models are based on a modification of the box model. The model described below assumes circular air patterns over the street (16). The background $PM_{2.5}$ concentration, plus C_L or C_w, is the total concentration for that respective side of the roadway.

$$C_L = \frac{7 * 10^6 * Q'}{[u + 0.5]\left[(x^2 + z^2)\, 0.5 + 2\right]} \qquad \text{(Eq. 7)}$$

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

where Q' = the emission rate in grams per meter per second (g/m/s).

C_L = concentration contributed by vehicle emissions for the downwind or leeward side (ug/m^3).

C_w = concentration contributed by vehicle emissions for the upwind or windward side (ug/m^3).

u = the average wind speed above the canyon (m/s).

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.

W = the width of the canyon (m).

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

Intersection Model

Intersection characteristics such as low vehicle speeds, starting and stopping, convergence of traffic, and travel delay, elevate particulate matter concentrations at intersections. The E. P. A. recommends the use of the model CAL3QHC for roadway intersection analysis. This is a Gaussian-based model that uses queue length to determine the length of the line source, and delay to calculate emissions rates for the queue (17). Because of these relationships, queue length is very important in calculating accurate concentrations. Guldberg performed a study comparing the queue length predicted by CAL3QHC and that predicted by the Highway Capacity Manual. The study found that CAL3QHC consistently predicted queue lengths approximately 11 percent shorter than the Highway Capacity Manual. The intersection study approaches outlined in the Highway Capacity Manual are accurate and nationally accepted (18). The results of this study should be taken into account when CAL3QHC is used.

SUMMARY

The models listed above have proved to be useful and fairly accurate. A recommended procedure is to collect field data and measurements, then use this information to calibrate the model. Once the model has been calibrated, future conditions can be incorporated into the model and forecasts can be made at a given location. These models are also useful in obtaining pollution concentrations that can be compared to other locations that use the same model.

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