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The U.S. Environmental Protection Agency (EPA) has designated Spokane, WA a non-attainment area for particulate matter with a diameter equal to or less than 10 micrometers (PM_{10}). Observed 24hr PM_{10} standard exceedances, neglecting windblown dust contributions from surrounding agricultural areas, stem primarily from residential wood combustion and unpaved and paved roads.

Particulate matter emission rates from roadway related activities, that are used to develop emission inventories, are currently based on highly uncertain EPA approved empirical correlations. In this work, stationary and mobile point source tracer release techniques were used to determine PM_{10} emission rates from 4-lane commercial/residential paved roads under sanded and unsanded conditions, and from unpaved roads relative to site specific vehicular and ambient parameters. Measured street (4+ lanes; ≥10,000 vehicles/day) emission factors for unsanded and sanded roads were respectively 40% and 10% lower than the EPA approved reference value. Notably the sanded condition was not four times greater than the unsanded case, as current EPA guidance suggests. Preliminary results indicate a consistent relationship between PM_{10} and relative humidity under unsanded conditions. Evidence suggests that street sweeping has a negligible effect on PM_{10} emission reduction. Considerable uncertainties were observed with the empirical emission factor equation. In fact, experimental observations were on average 90% lower than the empirical predictions.

Within the constraints imposed by the variable experimental conditions, the deduced emission factors for unpaved roads agreed reasonably well with the unpaved road empirical formula. Limited correlations were observed with ambient meteorological parameters.

The capability of the "upwind-downwind" concentration method to provide accurate emission factor predictions is also presented. Within a factor of two uncertainty limit, the SIMFLUX Gaussian dispersion model predictions agreed well with the experimentally determined emission factors. Lateral and vertical dispersion were seen to dominate roadway diffusion characteristics during periods of high and low wind speed, respectively

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PM10 Estimates for Eastern Washington

THE MEASUREMENT OF ROADWAY PM_{10} EMISSION RATES USING TRACER TECHNIQUES

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Abstract

The U.S. Environmental Protection Agency (EPA) has designated Spokane, WA a non-attainment area for particulate matter with a diameter equal to or less than 10 micrometers (PM_{10}). Observed 24hr PM_{10} standard exceedances, neglecting windblown dust contributions from surrounding agricultural areas, stem primarily from residential wood combustion, and unpaved and paved roads.

Particulate matter emission rates from roadway related activities, that are used to develop emission inventories, are currently based on highly uncertain EPA approved empirical correlations. In this work, stationary and mobile point source tracer release techniques have been used to determine PM_{10} emission rates from 4-lane commercial/residential paved roads under sanded and unsanded conditions, and from unpaved roads relative to site specific vehicular and ambient parameters. Measured street (4+ lanes; ≥10,000 vehicles/day) emission factors for unsanded and sanded roads were respectively 40% and 10% lower than the EPA approved reference value. Notably the sanded condition was not four times greater than the unsanded case, as current EPA guidance suggests. Preliminary results indicate a consistent relationship between PM_{10} and relative humidity under unsanded conditions. Evidence suggests that street sweeping has a

negligible effect on PM_{10} emission reduction. Considerable uncertainties were observed with the empirical emission factor equation. In fact, experimental observations were on average 90% lower than the empirical predictions.

Within the constraints imposed by the variable experimental conditions, the deduced emission factors for unpaved roads agreed reasonably well with the unpaved road empirical formula. Limited correlations were observed with ambient meteorological parameters.

The capability of the "upwind-downwind" concentration method to provide accurate emission factor predictions is also presented. Within a factor of two uncertainty limit, the SIMFLUX Gaussian dispersion model predictions agreed well with the experimentally determined emission factors. Lateral and vertical dispersion were seen to dominate roadway diffusion characteristics during periods of high and low wind speed, respectively.

INTRODUCTION

Under current federal law, ambient air quality standards have been established for six criteria pollutants: lead, carbon monoxide, ozone, sulfur dioxide, nitrogen dioxide, and fine particulate matter with a diameter equal to or less than 10 micrometers (PM_{10}). The major health effects associated with PM_{10} include breathing problems, and aggravation of existing cardiovascular and respiratory diseases. State Implementation Plans (SIP) are required by the Clean Air Act to ensure that air quality objectives are met. Each SIP identifies an area that violates air quality standards, and contains the strategies developed to attain those standards. The construction of a SIP requires a basic understanding of the type, size and location of various air pollution sources. This is attained through the creation of emission inventories. PM_{10} inventories are primarily categorized in terms of wood smoke, outdoor burning, industrial and road source emissions. These data bases, and subsequently the SIP's, rely heavily on source emission factors. These emission factors provide a basic quantitative description of source contributions to the overall pollutant inventory budget, are empirically deduced, and to a significant extent have a high degree of uncertainty attached to their application.

This report describes the results of research aimed at investigating and quantifying emissions from paved roads. Since a related project funded by the Washington State Department of Ecology on unpaved roads was carried out at the same time, similar results for unpaved roads are included. Through use of an improved tracer release technique, PM_{10} emission factors from

these important sources were deduced. A comparative assessment with current EPA approved emission factor models was performed. The effects of street sweeping, traction sands, and environmental conditions on particulate emissions from paved roads, were all examined in this work.

BACKGROUND

National Ambient Air Quality Standards (NAAQS) state that annually-averaged ambient PM_{10} levels should not exceed $50\mu g/m^2$, and that over a 24-hour period the PM_{10} concentration cannot exceed $150\mu g/m^2$, more than once a year. According to the 1990 Clean Air Act Amendments, the attainment status of an area for the 24-hour standard is based on the expected number of exceedances, where the expected number is classified as the estimated number of exceedances averaged over three years. During 1993 alone, eight estimated 24-hour exceedances were recorded at the Country Homes Boulevard monitoring site, Spokane (WSDOE, 1994). Accordingly, the Spokane area is currently classified by the US Environmental Protection Agency (EPA) as a non-attainment area for PM_{10}, with respect to the 24-hour standard. The area is prone to exceedances, due in part to the presence of typical characteristics conducive for high airborne particulate concentrations, including high traffic volume, low vehicular speeds, complex topographical features, and frequent meteorological inversions (Black et al., 1983). The major sources contributing to the observed 24-hour PM_{10} standard exceedances, neglecting windblown dust contributions from surrounding agricultural areas, are residential wood combustion, unpaved

roads and paved roads (WSDOE, 1993). Accordingly, within this air quality management area, there is a need to develop and improve PM_{10} emission inventories for State Implementation Plan (SIP) development.

Source Quantification Techniques

The quantification of air quality impacts from fugitive sources requires the use of air quality models. These models can be segregated into two categories: source-oriented, and receptor-oriented models. Source-oriented models primarily utilize Gaussian dispersion modeling techniques. These require intrinsic information on source emission rates and study area meteorology. The principal advantages in this system are that they can be used to directly predict the impact of either existing or proposed sources, they are widely available, and they do not require ambient air quality data. However, primary limitations include deficiencies in the quality of input data, and the inability of Gaussian models to reproduce the physical processes affecting pollutant transport. Under the ideal conditions of low-level continuous emissions, in simple terrain, with near neutral stability meteorological conditions, a "factor of two" agreement between observed and predicted concentrations is characteristic (Amer. Met. Society, 1978). Kitas and Lioy (1992) observed a typical 70% to 90% underestimation between predicted values from two Gaussian plume models and experimental data. Similar "factor of two" characteristics were also observed by Claiborn et al. (1995) while using a Gaussian line source dispersion model.

Receptor models are primarily used to determine source contributions to ambient concentrations which exceed NAAQS. Unlike dispersion models, they begin with particulate measurements at the receptors, from which source contributions are estimated by "back-calculation" using source

profiles (i.e., the chemical composition of each contributing source). Typically, general source types, such as primary geological, motor vehicle exhaust, vegetative burning, marine and industrial (e.g., copper smelting, steel production) are included (Watson et al., 1989). Uncertainties occur because (i) many sources have similar compositions; (ii) the exact number of sources contributing to a receptor are never accurately known; and (iii) components do react with each other and systems are not linear (Watson, 1984). Given the uncertainties in both approaches, it is important that both types of models are considered to be complementary, with each having strengths that compensate for the weaknesses of the other. In fact, current guidance for the development of PM_{10} State Implementation Plans recommend the application of both models with a reconciliation of their independent source apportionments (U.S. EPA, 1987).

Based on these identification techniques, evidence from urban areas in the United States suggest that re-entrained road dust and motor vehicle exhaust are major contributors to PM_{10} (Watson et al., 1989; Chow et al., 1992; Wang et al., 1992; Zimmer et al., 1992; Chow et al., 1993). Primary motor vehicle exhaust and road crustal emissions have been known to contribute between 4-40% and 30-50% of the total PM_{10} mass at urban sites, respectively (Watson et al., 1989; Chow et al., 1992; Chow et al., 1992; Kim et al., 1992). Modeling studies performed by the Washington State Department of Ecology (WSDOE) for Spokane suggest that under worst case scenario conditions paved and unpaved road contributions to overall PM_{10} emissions, excluding windblown dust, approach 50% and 30%, respectively (WSDOE, 1993). The paved road scenario (February and March) assumes lower residential wood burning and unpaved road source

emissions due to milder temperatures and increased soil moisture conditions. The increased paved road emissions are thought to be due to the action of vehicular traffic on the thawing and drying roads that have had traction materials, applied often under stagnant meteorological conditions. With respect to the unpaved road scenario (September to November) dry, stagnant atmospheric conditions along with typically low soil moisture are the contributing factors to the observed PM_{10} exceedances (WSDOE, 1993).

Paved Road Emission Factors

Particulate emission inventories from traffic related activities on paved roads are currently based on EPA approved empirical correlations found in "AP-42" (US EPA, 1985). The emission algorithm correlates total road surface dust loading and surface silt content to an overall PM_{10} emission factor, based on road type and traffic volume:

$$e = k\left(\frac{sL}{0.5}\right)^p$$

where e is the emission factor (g/Vehicle-Kilometer Traveled, or VKT), s is the silt fraction (% by mass of particles < 75μm), L is the total road dust loading (g/m^2), k is an empirical factor (for PM_{10}, k = 2.28), and p is a dimensionless exponent (for PM_{10}, p = 0.8).

The algorithm yields emissions under baseline unsanded conditions. Observed uncertainties with the equation stem from the use of only five sampling locations and a limited data base to represent urban area conditions in the US. Additionally, the AP-42 data base contains vehicle

exhaust, and brake and tire wear components that are representative of 1980, rather than current, conditions (Zimmer et al., 1992). Measured silt loadings have been found to vary by more than a factor of 20 for any respective road (Zimmer et al., 1992). Additionally, results from recent field studies in Denver (Zimmer et al., 1992) suggest that no consistent relationship exists between silt content and emission rates. In spite of these uncertainties, the use of silt loading measurements to characterize PM_{10} emission factors for paved roads is common.

Under sanded conditions a substantial increase in PM_{10} is expected to occur. AP-42 suggests that a factor of 4 is applied to unsanded PM_{10} estimates, based on the empirical equation, for the prediction of antiskid material emissions (US EPA, 1985), however no data confidently back this assertion. Another estimation technique for PM_{10} emissions from sanded roads utilizes the "gap filling" technique (Cowherd, 1988), which assumes that all the PM_{10} content of road sand is resuspended. The AP-42 document gives a default value of 0.018 pounds of PM_{10} emissions per ton of sand applied. This value does not take into account any additional particles formed through the grinding of larger material, and is itself based on data acquired from sandy soil/river sand and not from actual sanding material. Where salt is used for traction, a value of 10 pounds of PM_{10} emissions per ton of salt is suggested. This assumption is based on the notion that 10% of the applied salt is emitted as PM_{10}, however again no data are given to endorse this hypothesis. Due to the weak underlying assumptions and lack of supporting data, confidence in the predictive ability of these guidelines is poor. In fact, Zimmer et al., (1992), observe that use of the gap filling technique for sand application underpredicts actual emissions, while the

assumptions concerning road salt tend to overpredict. In a study performed in Minnesota that was aimed at clarifying correlations between source parameters and emission factors, measured factors were found to range between 1 and 11g/VKT, and did not follow any consistent relationship (Kinsey, 1993). Clearly, genuine uncertainties exist in the application of emission factors derived from these previous methods.

Unpaved Road Emission Factors

The unpaved road empirical emission model utilizes road surface and vehicle parameters to estimate PM_{10} contributions from vehicle travel. It was developed from a large data base using multiple linear regression techniques. The equation is recommended within the constraints set by the parameters from which it was formed (Fitzpatrick, 1987):

$$e = k(1.7)\left(\frac{s}{12}\right)\left(\frac{S}{48}\right)\left(\frac{W}{2.7}\right)^{0.7}\left(\frac{w}{4}\right)^{0.5}\left(\frac{365-p}{365}\right)$$

where e is the emission factor (kg/VKT); k is an empirical factor (for PM_{10}, k = 0.36); s is the silt content of the road surface material (%); S is the mean vehicle speed (km/hr); W is the mean vehicle weight (Mg); w is the mean number of wheels; and p represents the number of days per year with at least 0.254mm of precipitation.

Contrary to the general opinion that the unpaved model underpredicts emissions in western arid and semi-arid climates, Muleski and Stevens, (1992) report no evidence of systematic underprediction. In fact, they conclude that under the conditions of eastern/western regional and light/heavy-duty vehicle categories, historical data do not support a relationship between

moisture content and emission factor model performance. Both Claiborn et al. (1995) and Muleski and Stevens (1992) report relatively good agreement between the empirical equation and field results, which is somewhat surprising considering the model's limited underlying data base. Approximately 80% of the tests were conducted on industrial rather than public roads, so the data base was developed from road traffic characterized by heavier vehicles than are common on rural roads. Also, a considerable portion of the experiments were conducted at vehicle speeds lower than 35mph. Muleski and Stevens (1992) report that typical vehicle speeds are generally higher on unpaved rural roads in Arizona, and hence fall outside the range of values represented in the AP-42 data base. Considering that resuspended dust concentrations are 4% to 18% higher for trucks as compared to cars at the same speed, and higher vehicle speeds produce larger emissions, there is obvious scope for uncertainty (Kitsa and Lioy, 1992). Moreover, little is known of the relationship between emission factor and ambient temperature, relative humidity, or road moisture content during a period when roadway conditions progress from frozen, to thawed, to dry.

Tracer Release Methodology

Stationary and mobile point source tracer release techniques were used for the measurements described in this thesis. Cadle et al. (1977) and Okamoto et al. (1990) utilized mobile point source and line source tracer release schemes to simulate vehicular catalyst sulfate aerosol and NO_x tail-pipe emissions. The general pollutant emission rate model proposed by these investigators is:

$$Q_P = Q_{SF6} \times \left(\frac{C_p}{C_{SF6}} \right) \quad (1)$$

where Q_p is the pollutant emission rate, Q_{SF6} is the tracer gas release rate, and C_p, C_{SF6} are the respective net pollutant and tracer downwind concentrations (i.e., with background concentrations subtracted). A useful feature of this technique is that it provides its own quality control check; if measurements are properly conducted, if the tracer disperses in the same manner as the pollutant, and if the line source adequately models the road source, then the concentration ratios (C_p/C_{SF6}) should be constant regardless of how far downwind the samples are collected. In the studies by Cadle et al. and Okamoto et al., the observed similarity between the tracer and pollutant concentration ratios at varous sampling sites provided confirmation of their comparable dispersion characteristics.

Under a previous WSDOT sponsored project, the line source technique was extended to measure PM_{10} emissions from roads (Claiborn et al., 1995). Although the experiments of Cadle et al., related to aerosols of limited size (0.01-0.1μm), Claiborn et al. (1995) took the process a step further by describing road particulate emissions of 10μm and less. A key issue that needed to be addressed in that study was whether SF_6 disperses in the same manner as the larger particulate matter. Claiborn et al. (1995) suggest that under a "worst case" laminar flow condition (wind speeds $\leq 2ms^{-1}$), a 10μm size particle would settle 0.3m vertically for every 100m horizontally traveled. Although this could introduce some error into the analysis, typical experimental conditions were outside this general flow regime. Also, since SF_6 and particulate concentration

ratios were observed to be relatively constant, it was concluded that SF_6 and PM_{10} exhibit comparable dispersion characteristics over the distances employed in the experiments. Based on this result, equation 1 was used to calculate PM_{10} emission rates from roadways, from the measured tracer release rate, and upwind-downwind ambient tracer and PM_{10} concentrations.

SIMFLUX Dispersion Model Characteristics

Prior to the work of Claiborn et al., 1995, field measurement of fugitive mass emissions was often conducted using the "upwind/downwind" method. This requires the measurement of upwind and downwind particulate concentrations and meteorological conditions, followed by application of a Gaussian model. The dispersion equations are used to back-calculate the emission rate required to generate the observed downwind concentrations. An appraisal of this process was performed using the SIMFLUX point/line source dispersion model (Howard et al. 1992).

Many models have been devised to describe the dispersion of pollutants from roadways (Zimmerman and Thompson, 1975; Chock, 1978; Peterson, 1980). These primarily treat emissions from each lane of a road as a continuous, straight, finite line source with uniform emission rate. Similarly, the Gaussian model, SIMFLUX, characterizes the roadway source by defining it as a line source composed of a series of point sources. The SIMFLUX model employs the superposition principle, which treats the concentration at a receptor as the summation of contributions from all the point sources comprising the line source. Cadle et al. (1977) noted the importance of mechanical mixing in the dispersion of pollutants near roadways.

The SIMFLUX model accommodates mechanical mixing through use of horizontal and vertical dispersion coefficients, which are related to corresponding horizontal and vertical virtual point source dimensions,

$$\sigma_{Yo} = \frac{Y}{4.3}$$

$$\sigma_{Zo} = \frac{H}{2.15}$$

where, σ_{Y0}, σ_{Z0} are the initial horizontal and vertical coefficients, and Y and H are the horizontal and vertical source dimensions. This technique acts to simulate the lateral and vertical distortion of the plume created by vehicular turbulence, by altering initial vertical and horizontal dispersion coefficients. The model uses Pasquill-Gifford (PG) diffusion curves for horizontal and vertical dispersion. Since the PG curves are based on 10 minute averaging periods, a correction is applied to the horizontal diffusion coefficient to account for increased plume meander during longer averaging intervals (Howard et al. 1992). It is important to note that dispersion characteristics change with wind direction relative to the road. Chock (1978) observed the entrainment of pollutants in the upwind region due to shear, hence diluting the downwind concentration as the angle of wind direction relative to the road increased. Additionally, Howard et al. (1992) observed that errors in wind direction measurement could have a significant effect on downwind concentration predictions. On the whole, results presented by Howard et al. (1992) illustrate the success of the SIMFLUX model, and corresponding assumptions, in describing dispersion from area sources when modeled as a series of high density point source arrays. Similarly, Claiborn et al. (1995) used this technique to describe road emissions. Results were typically within a "factor

of two" when compared to experimental observations, which is characteristic of a good Gaussian description of plume dispersion (Amer. Met. Society, 1978).

RESULTS OF PRIOR WSDOT-SPONSORED RESEARCH

Claiborn et al. (1995) describe preliminary procedures and experiments relating to the evolution of the tracer technique. Their initial experiments used a tracer line source technique where SF_6 was released from a 100m line parallel to the road to simulate the roadway PM_{10} source. Samples were collected upwind and downwind of the road along the wind direction. Assuming the tracer gas disperses similarly to PM_{10}, then the ratios of the tracer source rate to PM_{10} source rate and tracer concentrations to PM_{10} concentrations will be equal. The emission rate of PM_{10} ($Q_{PM_{10}}$) can be calculated if the tracer line source release rate (Q_{SF_6}) and the concentrations of PM_{10} and SF_6 on both sides of the road are all known,

$$Q_{PM10} = \frac{Q_{SF6} \times \Delta C_{PM10}}{\Delta C_{SF6}} \qquad (2)$$

where $\Delta C_{PM_{10}}$ is the difference between downwind and upwind PM_{10} concentrations ($\mu g/m^3$) and ΔC_{SF_6} is the difference between downwind and upwind SF_6 concentrations ($\mu g/m^3$).

Through propagation of error analysis, uncertainties in the PM_{10} emission rate calculation from the line source experiment were approximately 43% (Claiborn et al., 1995). This error was primarily related to uncertainties in the SF_6 line flow rate measurement. Specifically, the typical

measured flow rate difference between the line source capillary vents and the flow control rotameter was 35%. In an attempt to circumvent this problem, in the current project it was proposed that a point source tracer release process might reduce the uncertainty, since the error in the SF_6 release rate measurement would be less than ± 5%. In this experiment, SF_6 would be released from a point-source immediately next to the road and samples were collected along downwind lines parallel to the road (Figure 1). SF_6 samplers are specifically positioned to capture the tracer plume across the road. To account for the difference in geometry between the tracer line source and point source, the downwind tracer plume profiles are integrated to determine the net tracer mass flux. Hence, the PM_{10} emission rate can be calculated:

$$Q_{PM10} = \frac{Q_{SF6} \times C_{PM10}}{\int_0^x C_{SF6}\, dx} \qquad (3)$$

where Q_{SF6} is the tracer point source release rate (μg/s), C_{PM10} is the difference between downwind and upwind PM_{10} concentrations (μg/m^3), $\int C_{SF6}\, dx$ is the crosswind integrated tracer concentration (μg/m^2), and x is the distance along the sampler array, typically $0m \leq x \leq 120m$.

The evaluated PM_{10} emission rate will then be converted to an emission factor (g/VKT), using known traffic volumes on the street. Error propagation analysis for the point source experiment, under a worst case scenario, predicts an overall error of 25%, which will provide increased confidence in the results obtained (Claiborn et al., 1995).

Thus, in this work, further tracer experiments were conducted to determine PM_{10} emission rates from 4-lane commercial/residential paved roads under both sanded and unsanded conditions, and from unpaved 2-lane roads relative to site specific ambient parameters (temperature, relative humidity and/or road moisture content). Stationary and mobile point source tracer release systems were used for the paved and unpaved road experiments, respectively. With a consistent traffic flow, the stationary release should provide adequate simulation of the road source. However, under a negligible traffic flow environment, a mobile release may prove to be a useful technique to simulate particulate emissions. For comparison, site specific information was collected from the paved and unpaved road sites, and used to predict PM_{10} emissions based upon the EPA approved empirical algorithms. The effects of street sweeping on the experimental and empirically calculated emission factors were also assessed. A dispersion modeling study using SIMFLUX was conducted to compare model predicted emissions, obtained from a basic upwind-downwind concentration method, to actual site generated emission factors.

In the present work, the point source tracer techniques thus developed were employed to examine 1) PM_{10} emission rates from 4-lane commercial/residential paved roads under both sanded and unsanded conditions, 2) PM_{10} emissions from 2-lane unpaved roads relative to site specific ambient parameters (temperature, relative humidity, precipitation, and wind speed), and 3) effects of street sweeping on PM_{10} emission factors. Stationary and mobile point source tracer release systems were used to describe particulate source emissions from the paved and

unpaved road environments, respectively. For comparison, site specific information was collected from the paved and unpaved road sites, and used to predict PM_{10} emissions based on the EPA approved empirical algorithms. A dispersion modeling study, using a dense point source array Gaussian plume model (SIMFLUX), was conducted to compare the performance of model predicted emissions, obtained from a basic upwind-downwind concentration method, to actual site generated emission factors. Initial horizontal and vertical dispersion coefficients, defining the vehicular mechanical mixing process, were determined by calibrating the model with tracer and meteorological data.

EXPERIMENTAL METHODS

Stationary Point Source Tracer Experiment

Studies were performed at two paved road sites, each of which was characterized as a 4-lane commercial/ residential road with traffic volumes exceeding 10,000 vehicles per day. This road class is predicted to account for nearly 90% of total annual paved road emissions in the Spokane area (WSDOE, 1993). Baseline emission experiments were performed in a residential area, on Country Homes Boulevard (Site 1) at a location near the Spokane County Air Pollution Control Authority (SCAPCA) air monitoring station. Site 2 experiments were conducted on N. Division Street, in a commercial area. At this site, traction sand was applied during the winter months. The traction material was characterized by a predominant 3/8 inch size distribution, a durability index of 25% maximum and a moisture content of 7% maximum. All material was deposited,

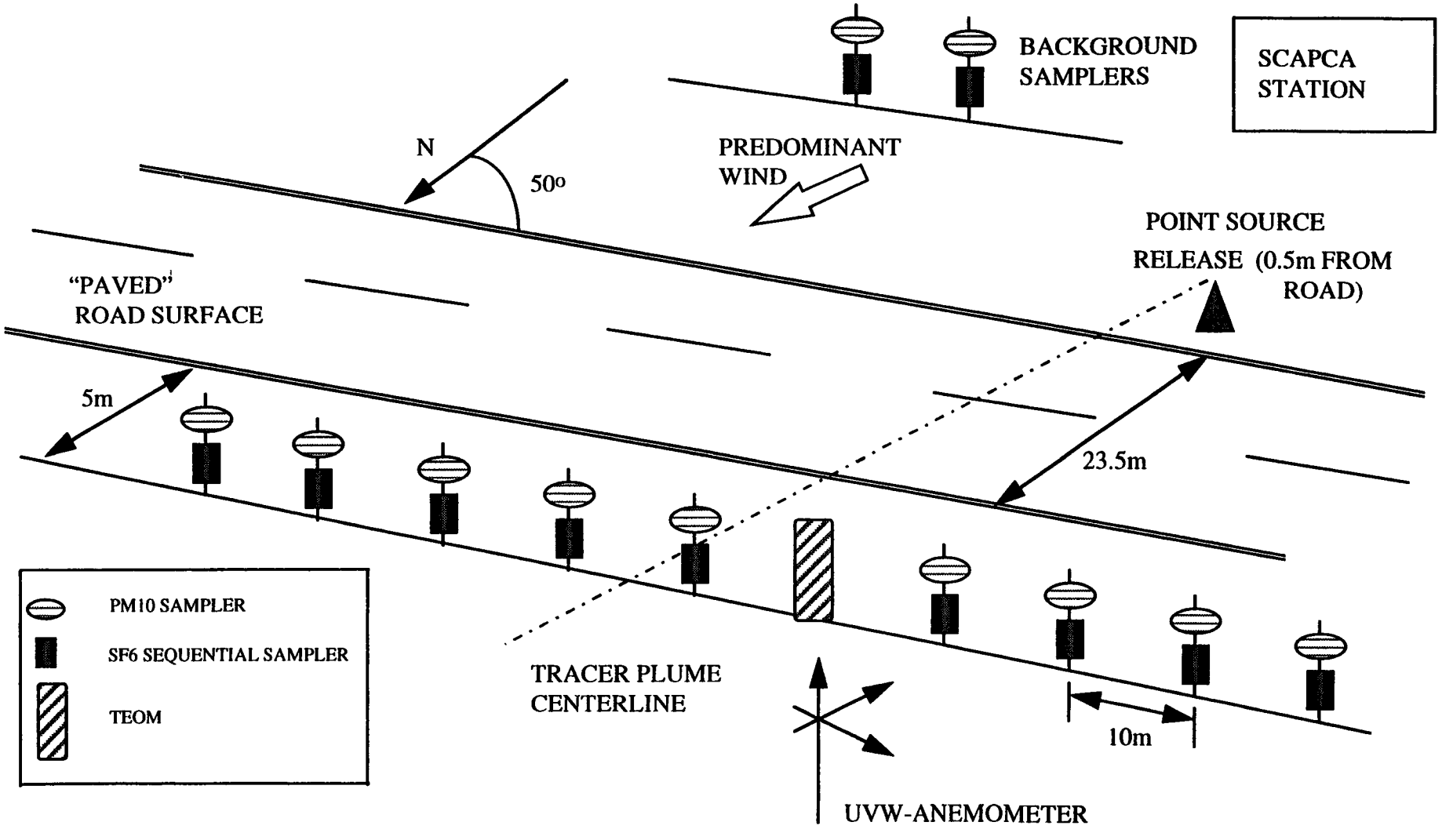

Figure 1. The Stationary Point Source Tracer Release Method: Country Homes Blvd., Spokane, WA

under prearrangement with the city road maintenance department, approximately 12 hours prior to the sampling run.

Consistent experimental procedure and layout for both sites were maintained. The inert tracer gas was released upwind of the road, and regulated to a constant rate by a 25cc/min mass flow controller. PM_{10} samples were obtained using low volume saturation samplers (Airmetrics, Springfield, OR) and Tapered Element Oscillating Microbalance (TEOM) continuous PM_{10} monitors. SF_6 samples were collected using programmable, battery operated syringe samplers. SF_6 and PM_{10} samplers were located at both upwind and downwind positions, at a height of approximately 1-2m above the ground. For background concentration determination, care was taken to position these samplers away from any potential emission source. Downwind of the road, a TEOM monitor was positioned within an array of multiple co-located SF_6 and PM_{10} crosswind samplers (Figure 1). During the Site 1 experiments, the TEOM monitor at the SCAPCA station provided either upwind or downwind PM_{10} measurements depending on the wind direction. Downwind SF_6 samplers were specifically positioned to capture the entire tracer plume emitted from the point-source. Since roadway PM_{10} emissions are assumed to be an infinite line source, the PM_{10} concentrations observed along the sampler array should be constant. Continuous wind speed, wind direction and temperature measurements were obtained via portable uvw-anemometer recordings and SCAPCA station meteorological instruments. Respective temperature measurements made at anemometer and SCAPCA station heights allowed for temperature gradient determination, and hence atmospheric stability class

assessment. Relative humidity measurements were obtained from the Spokane International Airport, approximately eight miles from Site 1. Pneumatic traffic counters were used to measure traffic volumes at 15 minute intervals. Hourly counts were determined by summing the individual 15 minute components, and then used in evaluating the emission factor per vehicle.

Mobile Point Source Tracer Experiment

Experiments were conducted at three unpaved road sites situated within non-irrigated agricultural lands near Pullman, WA. Initial experiments were conducted using a stationary point source system, however, resultant spiky tracer profiles led to the assumption that the low traffic volume did not provide sufficient mixing of the SF_6 plume as it crossed the road. Additionally, it was noted that approximately 85% of the reported traffic was due to our own vehicular movement, which occurred at a rate of one pass per minute. Hence, the experimental design was changed to a mobile point source release, where the SF_6 tracer gas was released from a vehicle driven back and forth in front of the sampler array. The tracer release system was identical to that used in the stationary point source release experiment, but the flowrate was increased to 235cc/min. Care was taken to ensure that the turnaround sites were at least 1/2 mile away from the sampling area and that a constant velocity of 35 miles per hour was maintained throughout the sampling period. SF_6 and PM_{10} samplers were positioned together along the general line of wind direction, both upwind and downwind (Figure 2), and at heights approximately 1-2m above the ground. The ratio of the concentrations observed between any co-located samplers should be

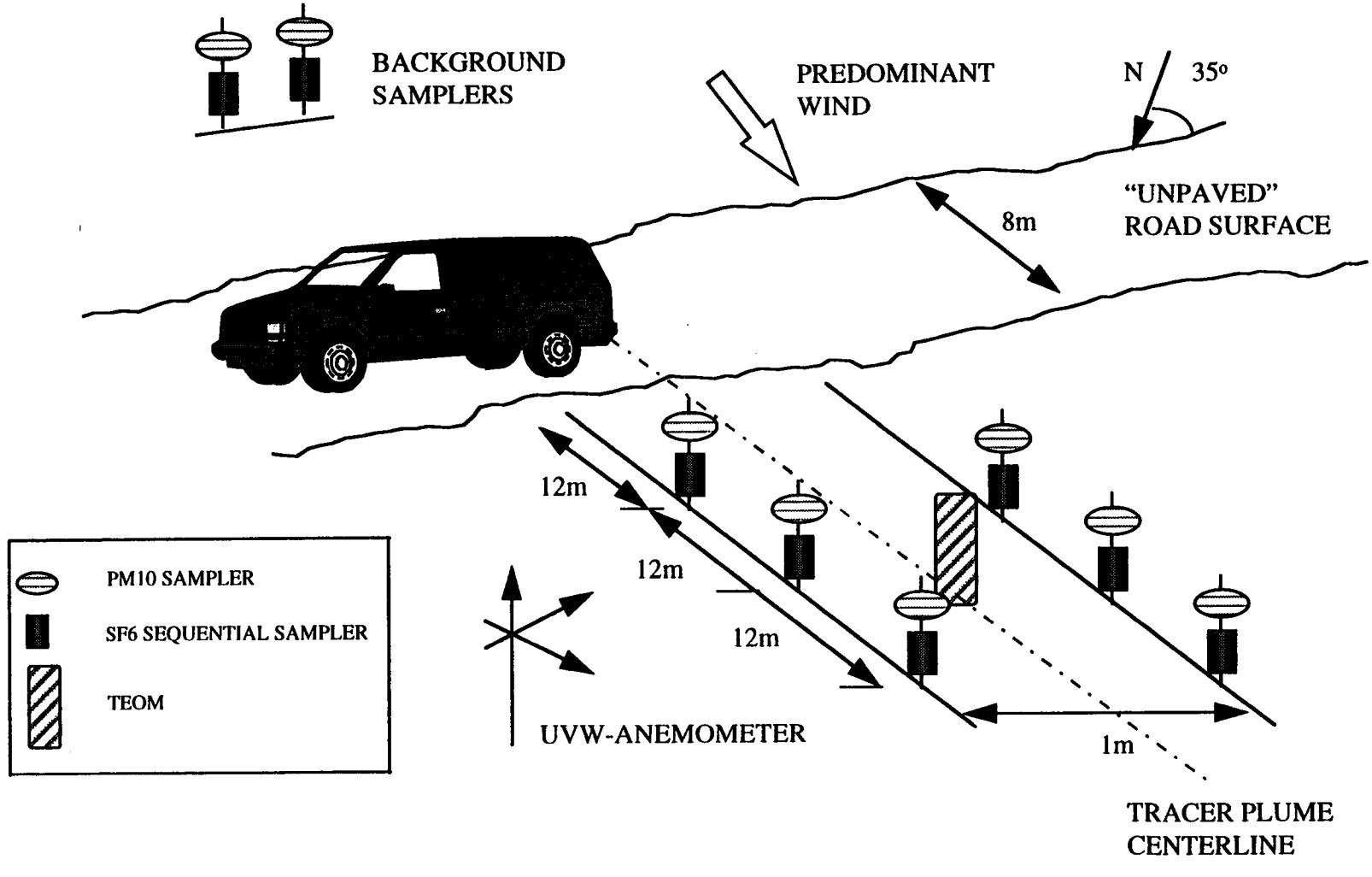

Figure 2. The Mobile Point Source Tracer Release Method: Country Club Rd. (Main).

the same, since the SF_6 and PM_{10} source emissions are occurring concurrently and dispersing analogously. The PM_{10} emission rate can be calculated using equation 2, where Q_{SF6} ($\mu gm^{-1}s^{-1}$) is calculated from the point source release rate (μgs^{-1}), the vehicle speed (ms^{-1}) and the number of vehicle passes per second (s^{-1}).

Traffic volumes and meteorological data (ambient temperature, wind speed and direction) were recorded via a pneumatic traffic counter and uvw-anemometer, respectively. Precipitation history and humidity were recorded by the meteorological station situated on the roof of the Washington State University engineering building, approximately three miles from the site.

Sample Analysis

The low volume saturation samplers operate at a flow rate of 5 l/min. Particle separation is attained through impaction, with the desired particulate sizes collected on 47mm quartz microfiber or Teflon PTFE membrane filters (Whatman International Ltd.). Although the system is characterized by a low flow rate and a small impaction inlet, extensive co-location tests have provided adequate demonstration of its effectiveness. Claiborn et al. (1995), report good agreement between samplers co-located with a TEOM. The low-volume samplers were typically $2\mu g/m^3$ higher than the TEOM, which was itself reporting values approximately 12% lower than a Selective Size Inlet (SSI) system (EPA reference method). In the Philadelphia Diesel Particulate monitoring study, low-volume samplers reported mass concentrations approximately $5\mu g/m^3$ lower than co-located dichotomous samplers (U.S. EPA, 1994). Other investigators also note

excellent agreement between these samplers and EPA reference method instruments (Parisi et al., 1993; Chow et al., 1995). During this current study, saturation samplers were co-located with SCAPCA TEOM and SSI devices during 5/94 and 4/95. The accuracy of the saturation samplers compared to these instruments in May 1994 was -5% and +13% respectively. During April 1995, the accuracy was observed to be -15% and -18% respectively. Comparison with the Washington State University TEOM over a 24hr period yielded an accuracy of -9%. The measured precision (percent standard deviation) of the saturation samplers during the co-location periods was approximately ± 6% and ± 15%, respectively.

Atmospheric PM_{10} mass concentrations were obtained gravimetrically via a 250mg capacity, 1μg sensitivity microbalance (Cahn model 33). Unexposed and exposed filters were equilibrated at 40 ± 5% relative humidity for at least 24hr prior to weighing. All filters were weighed twice before and after sample collection. 10% of the filters were reweighed and 3 blanks per 20 filters were weighed for procedure precision and accuracy determination. The unexposed blank filters had an observed standard deviation of ± 5μg over a typical 24hr weighing period.

The TEOM (RP-1400) device provides continuous particle mass concentration measurements. To prevent contraction and expansion of the tapered element, which would affect measurement precision, the sample chamber and inlet air must be kept at a constant temperature. This is typically 50°C, but due to the possibility of ammonium nitrate and organic compound volatilization, the temperature is lowered to 30°C during the wood burning season in Spokane

(10/1 - 4/1) when high concentrations of these compounds are expected. The system detection limit is 5$\mu g/m^3$ for a five minute average (Operation Manual). During a ten day co-location test against SCAPCA SSI instruments in May 1995, measurements averaged over a 24hr period indicated a precision of ± 7% and an accuracy of +5%. Similarly, the accuracy of SCAPCA TEOM concentrations, averaged over 24hr periods for the month 4/95, to SSI data was +4%.

SF_6 samples were collected each hour by sequential automated syringe samplers. Claiborn et al. (1995) identify measurement error between different sets of deployed systems to be approximately 17%. An electron capture gas chromatograph was used to analyze the SF_6 samples, with the tracer peak areas evaluated by an electronic integrator. Calibration was performed using certified standard SF_6 concentrations (Scott-Martin Inc., certified accuracy ± 5%). Reproducibility of these standards, which ranged between 73.7 parts per trillion (ppt) and 9980ppt, was consistently within ± 5%. All samples were analyzed within a day of collection, since leakage from the syringes was observed to be approximately 5%, 30% and 35% of the original concentration after 1,3 and 4 days respectively. 10% of all the samples were re-analyzed during each analytical period. The precision of the analysis was approximately ± 4%.

Based on estimated errors of 15%, 7%, 17%, 5% and 4% for measurement uncertainties for the low-volume sampler, TEOM, SF_6 measurement, flow rate measurement, and GC analysis, the estimated error was 25%. It is important to note that the PM_{10} low-volume sampler error

identification encompasses uncertainties associated with not only the programmable samplers, but also reproducibility of weighing procedures.

COMPUTATIONAL PROCEDURES

Field measurement of fugitive mass emissions has often been conducted using the "upwind/downwind" method. This requires the measurement of upwind/downwind particulate concentrations and meteorological conditions, followed by application to a Gaussian model. The dispersion equations are used to back-calculate the emission rate required to generate the observed downwind concentration pattern. For this study, the SIMFLUX dispersion model was used (Howard et al., 1992). All meteorological and concentration data used in this test were obtained from the unsanded Country Homes Blvd. experiments.

The SIMFLUX model employs the superposition principle, namely, the concentration at a receptor is the summation of contributions from all the point sources making up the line source. Cadle et al. (1977) noted the importance of mechanical mixing in the dispersion of pollutants near roadways. The SIMFLUX model accommodates this occurrence through use of horizontal and vertical concentration distributions, which are related to corresponding horizontal and vertical virtual point source dimensions,

$$\sigma_{Yo} = \frac{Y}{4.3} \tag{4}$$

$$\sigma_{Zo} = \frac{H}{2.15} \tag{5}$$

where, σ_{Y_0}, σ_{Z_0} are the initial standard deviations of the source horizontal and vertical concentration distributions, and Y and H are the horizontal and vertical source dimensions. This technique acts to simulate the lateral and vertical distortion of the plume created by vehicular turbulence, by altering the initial horizontal and vertical dispersion coefficients. The model uses Pasquill-Gifford (PG) diffusion curves for horizontal and vertical dispersion. Since the PG curves are based on 10 minute averaging periods, a correction has been applied to the horizontal diffusion coefficient to account for increased plume meander during longer averaging intervals (Howard et al., 1992).

To attain a successful correlation between predicted model and observed experimental plume concentrations, tracer and meteorological data were initially used to calibrate the model. This process allows for some of the uncertainties associated with the basic upwind/downwind method to be eliminated, since volume source dimensions can be altered to secure a best plume fit. Once calibrated, the road source was modeled as a continuous, straight, finite line source, with uniform emission rate. To ensure best simulation results, the line source was positioned centrally to the 4-lane road and its point spacings were kept within one standard deviation of the lateral dispersion (σ_y) (Howard et al., 1992). It can be shown that if the summation of the effects from a finite line source at a point is viewed as an integration problem, then the best approach is to evaluate at locations along the line source that are chosen by the appropriate Gaussian integration method (Bowman, 1976). Howener Dumbaald (1977) derived an analytical solution for a finite

line source oriented at an arbitrary angle to the wind which made all techniques using the evaluation at individual points obsolete.

RESULTS

Stationary Point Source Release Experiments

Results from separate studies, with and without a traction sand component, are summarized in Tables 1 and 2. Test 11/18/93 was previously reported and is included here for completeness. Although ten further unsanded road experiments were conducted on Country Homes Blvd., only nine are reported here since extreme wind shifts occurred in the remaining experiment. Similarly, eight sanded road experiments were performed on Division St. between 2/2/95 and 3/16/95. Five are reported here, while the other three experienced contamination and wind shift problems.

The prevailing winds during the Country Homes Blvd. experiments (Tests 2/24/94 - 5/21/95) were typically either from a direction 190-210^{o} or 330-350^{o} N, where the road itself is positioned along an axis 50^{o} N. Accordingly, the sampler array was deployed to accommodate these southerly or northerly winds. During the test periods 2/24/94 - 7/11/94, the SCAPCA monitoring station was located at a site approximately 210^{o} N, and approximately 4m from the road. For the 5/21/95 experiment the SCAPCA station was at a new position approximately 30^{o} N, and approximately 4m from the road. Experiments conducted on Division Street, which runs in a north-south direction, had prevailing 35-90^{o} or 190-210^{o} N winds. In general, wind speeds

TABLE 1: STATIONARY POINT SOURCE TRACER RELEASE PM_{10} EMISSION FACTOR EXPERIMENTS, SPOKANE, WA.

Baseline Emission Factors Without a Sand Component

Date & Experimental Duration[1]	Site[2]	WS (ms^{-1})	WD[3] (^oN)	Relative Humidity[4] (%)	Time After Street Sweep[5] (days)	Average Traffic Count (vehicles/hr)	Calc. Emission Factor (g/VKT)	Pred. Emission Factor (g/VKT)[6]	Comparison to Predicted Factor
11/18/93 (1000-1600)	1	5.3 ± 0.7	210 ± 41^o	69	21	2920	0.58	1.8	-67.8%
2/24/94 (1300-1600	2	3.8 ± 1.2	-29 ± 22^o	33.7	45	990	1.22	1.8	-32.2%
3/29/94 (1400-1700)	2	3.3 ± 0.9	10 ± 29^o	12	10	1190	1.13	1.8	-37.2%
4/18/94 (1330-1830)	2	4.9 ± 1.1	-8 ± 12^o	46	10	1270	0.56	1.8	-68.9%
5/12/94 (1200-1700)	2	6.1 ± 1.4	192 ± 19^o	24	22	1190	1.71	1.8	-5%
6/03/94 (1400-1900)	2	3.3 ± 0.6	207 ± 83^o	34.5	44	1250	0.76	1.8	-57.8%
6/10/94 (1600-1900)	2	1.8 ± 1.2	262 ± 99^o	29.5	3	1290	0.546	1.8	-69.7%

[1]Experiment 11/18/93 is from previous DOT project.
[2] Sites 1 & 2 are Major Streets, i.e. 4+ lane Commercial/Residential Road with traffic volumes exceeding 10,000 vehicles per day. Site 1: Division Street, Spokane; Site 2: Country Homes Boulevard, Spokane.
[3]Site 1 is positioned along a 0^o Northerly axis; Site 2 is positioned along a 50-230^o axis from north.
[4] Humidities represent 3 hour averages over the experimental period.
[5] The streets were swept on 11/7/93, 1/7/94, 1/10, 3/7, 3/19, 4/8, 4/20, 6/7/94 and 4/11/95.
[6] Predicted factors were determined from the US EPA, AP-42 document.

TABLE 1 (continued): STATIONARY POINT SOURCE TRACER RELEASE PM_{10} EMISSION FACTOR EXPERIMENTS, SPOKANE, WA.

Baseline Emission Factors Without a Sand Component

Date & Experimental Duration	Site[1]	WS (ms^{-1})	WD[2] (^oN)	Relative Humidity[3] (%)	Time After Street Sweep[4] (days)	Average Traffic Count (vehicles/hr)	Calc. Emission Factor (g/VKT)	Pred. Emission Factor[5] (g/VKT)	Comparison to Predicted Factor
6/21/94 (1300-1800)	2	1.3 ± 1.1	209 ± 92^o	22	22	1210	1.17	1.8	-35.0%
7/11/94 (1200-1600)	2	4.8 ± 0.7	205 ± 30^o	11	34	1200	2.325	1.8	+29.2%
5/21/95 (1300-1700)	2	4.7 ± 1.7	349 ± 33^o	—	40	1030	0.41	1.8	-77.2%
						AVERAGE	**1.04 ± 0.57**		**-42 ± 34%**

[1] Sites 1 & 2 are Major Streets, i.e. 4+ lane Commercial/Residential Road with traffic volumes exceeding 10,000 vehicles per day. Site 1: Division Street, Spokane; Site 2: Country Homes Boulevard, Spokane.

[2] Site 1 is positioned along a 0^o Northerly axis; Site 2 is positioned along a 50-230^o axis from north.

[3] Humidities represent 3 hour averages over the experimental period.

[4] The Streets were swept on 11/7/93, 1/7/94, 1/10, 3/7, 3/19, 4/8, 4/20, 6/7/94 and 4/11/95.

[5] Predicted factors were determined from the US EPA, AP-42 document.

TABLE 2: STATIONARY POINT SOURCE TRACER RELEASE PM_{10} EMISSION FACTOR EXPERIMENTS, SPOKANE.

Emission Factors With a Sand Component

Date & Experimental Duration[1]	Site	WS (ms^{-1})	WD[2] (^0N)	Relative Humidity (%)	Average Traffic Count	Calc. Emission Factor (g/VKT)	Pred. Emission Factor (g/VKT)[3]	Comparison to Predicted Factor
2/9/95 (1200-1700)	1	3.2 ± 0.6	197 ± 9°	68.5	2580	1.58	1.8	-12.2%
2/21/95 (1200-1700)	1	1.9 ± 0.7	99 ± 11°	25	2680	1.39	1.8	-22.8%
2/28/95 (1200-1700)	1	5.7 ± 1.6	42 ± 71°	27.5	3020	1.73	1.8	-3.9%
3/07/95 (1100-1700)	1	2.9 ± 0.9	81 ± 89°		2550	1.96	1.8	+9%
3/16/95 (1200-1700)	1	3.9 ± 1.2	35 ± 49°		2930	0.78[4]	1.8	-57%
					AVERAGE	**1.47 ± 0.44**		**-19 ± 25%**

[1] Site 1 is a Major Street, i.e. 4+ lane Commercial/Residential Road with traffic volumes exceeding 10,000 vehicles per day.
Site 1: Division Street, Spokane.
The experiments were performed approximately 12hr after sanding of the road.

[2] [2]Site 1 is positioned along a 0^0 Northerly axis.

[3] Predicted factors were determined from the US EPA, AP-42 document.

[4]Emission factor based on minivol results averaged over the sampling period. All other factors are determined using a 1-hr TEOM average.

throughout all these experiments were consistently higher than $1 ms^{-1}$. To account for observed variations in wind and traffic conditions as well as sampler battery problems, TEOM and SF_6 data were used to calculate hourly emission factors, which were then averaged over the test period. This technique worked favorably with the automated hourly SF_6 samplers, and permitted rejection of particular hours exhibiting undesirable wind directions. The low-volume samplers provide an overall mass concentration for the specified experimental period (typically five to seven hours). Hence, the noted difference between low-volume sampler and TEOM calculated emission factors in Tables 1 and 2 can be accounted for by errors induced through rejection of individual hours. Figure 3 illustrates typical sequential SF_6 data obtained from the experiment on 2/9/95, as a function of position along the sampler array. During hours where plume shift occurred due to changing wind conditions, the edges of the plume were extrapolated to zero and the overall area under the curve integrated for cross-wind integral determination.

For C_{PM10} calculation, the 5/12/94, 6/21/94, 7/11/94 and 5/21/95 experiments utilized upwind and downwind PM_{10} concentrations from the Washington State University and SCAPCA TEOM's. During all the other experiments, except 11/18/93 and 3/16/95 when upwind and downwind saturation samplers were used, the net downwind PM_{10} concentration was determined by subtracting the average upwind low-volume sampler concentration from the hourly downwind TEOM concentrations. To circumvent the possible error induced into the PM_{10} calculation through the use of two different instruments, a calibration curve was determined using results from co-location exercises between the two devices. Using the general

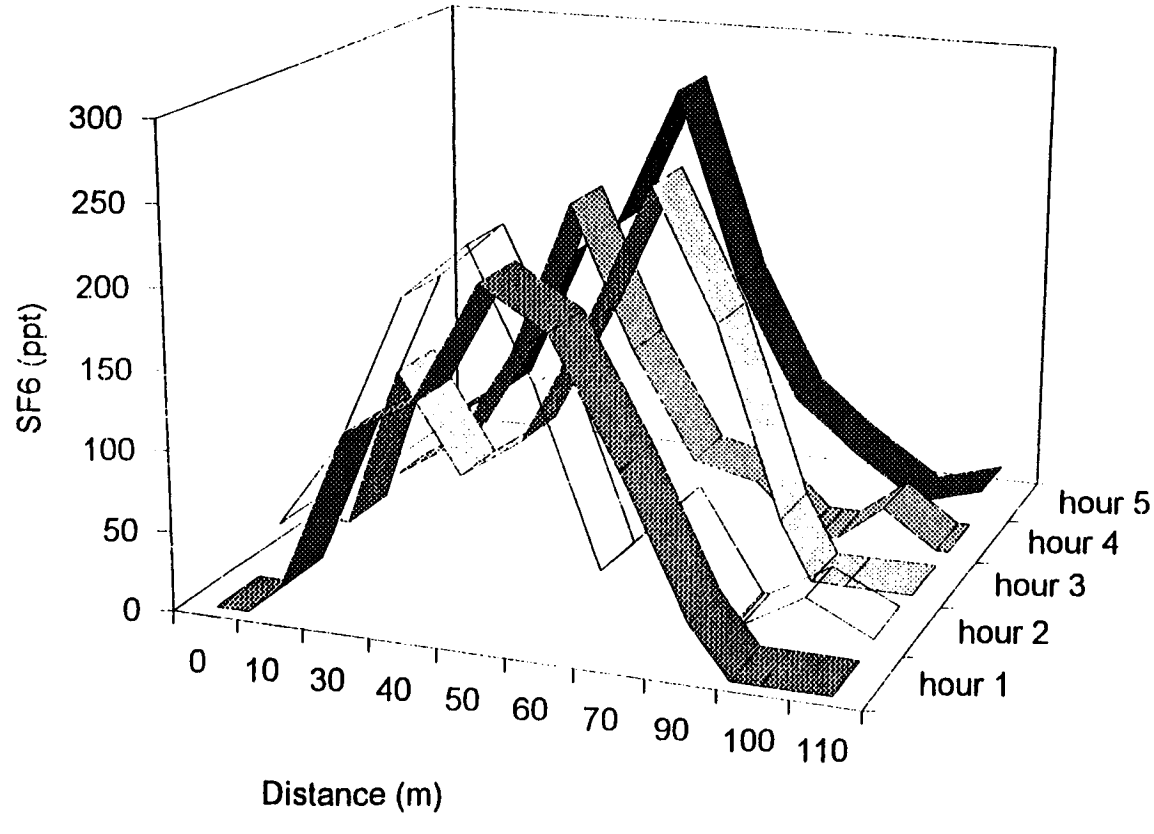

Figure 3. Downwind tracer plume profiles: 2/21/95.

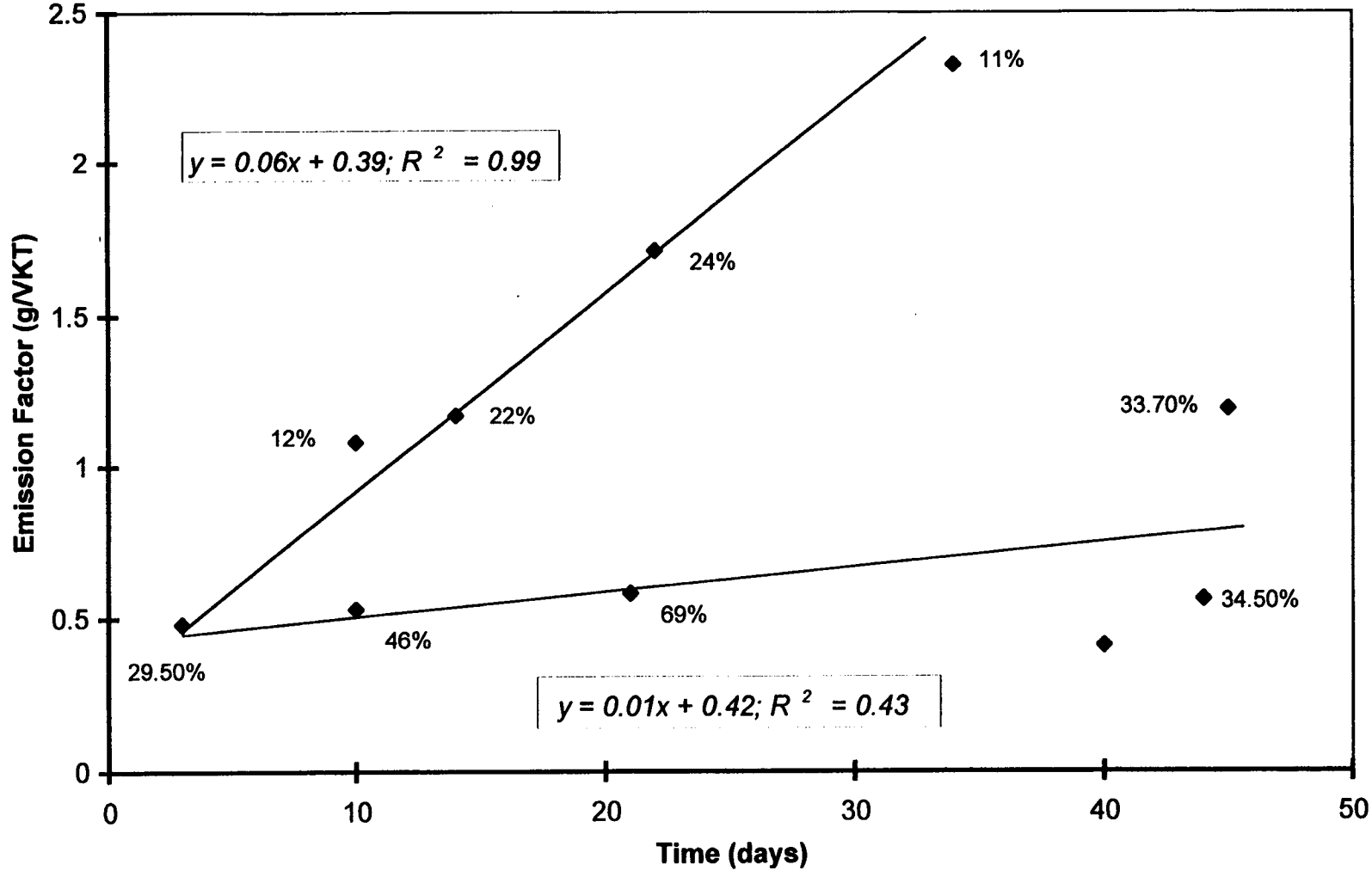

Figure 4. Observed relationship between emission factor and the duration of time each experiment was performed after a street sweep; 11/18/93 -5/21/95 unsanded expts.

equation $y = 0.5497 + 0.8898x$ ($R^2 = 0.759$; based on 16 data points), where y is the saturation sampler concentration and x is the TEOM concentration, upwind saturation sampler concentrations were converted to a corresponding TEOM concentration and then used to calculate C_{PM10}. A similar operation was performed on the 11/18/93 to 5/12/94.

Street sweeping exercises were performed on 11/7/93, 1/7/94, 1/10, 3/7, 3/19, 4/8, 4/20, 6/7/94 and 4/11/95. Figure 4 illustrates the relationship observed between emission factor and the duration of time the respective experiment was performed after a street sweep ($R^2 = 0.024$). It is evident that the overall correlation is composed of two distinct trends, governed by relative humidity. Although further analysis of the data indicates no correlation between wind parameters and emission factors, it was observed that under unsanded conditions there is an inverse correlation ($R^2 = 0.413$) between atmospheric relative humidity and the emission factor (Figure 5). Each humidity reading presented is the average value observed over the course of the experiment. A multiple regression analysis of the independent relative humidity and street sweep time variables against emission factors indicate an overall increase in the R^2 correlation coefficient to 0.512. With respect to the sanded experimental condition, no humidity, wind speed or wind direction correlations were observed.

Silt Loading Measurements

Table 3 presents silt loading and emission factor data, calculated using the AP-42 empirical equation. Three samples were collected from each lane of the four-lane Country Homes Blvd.

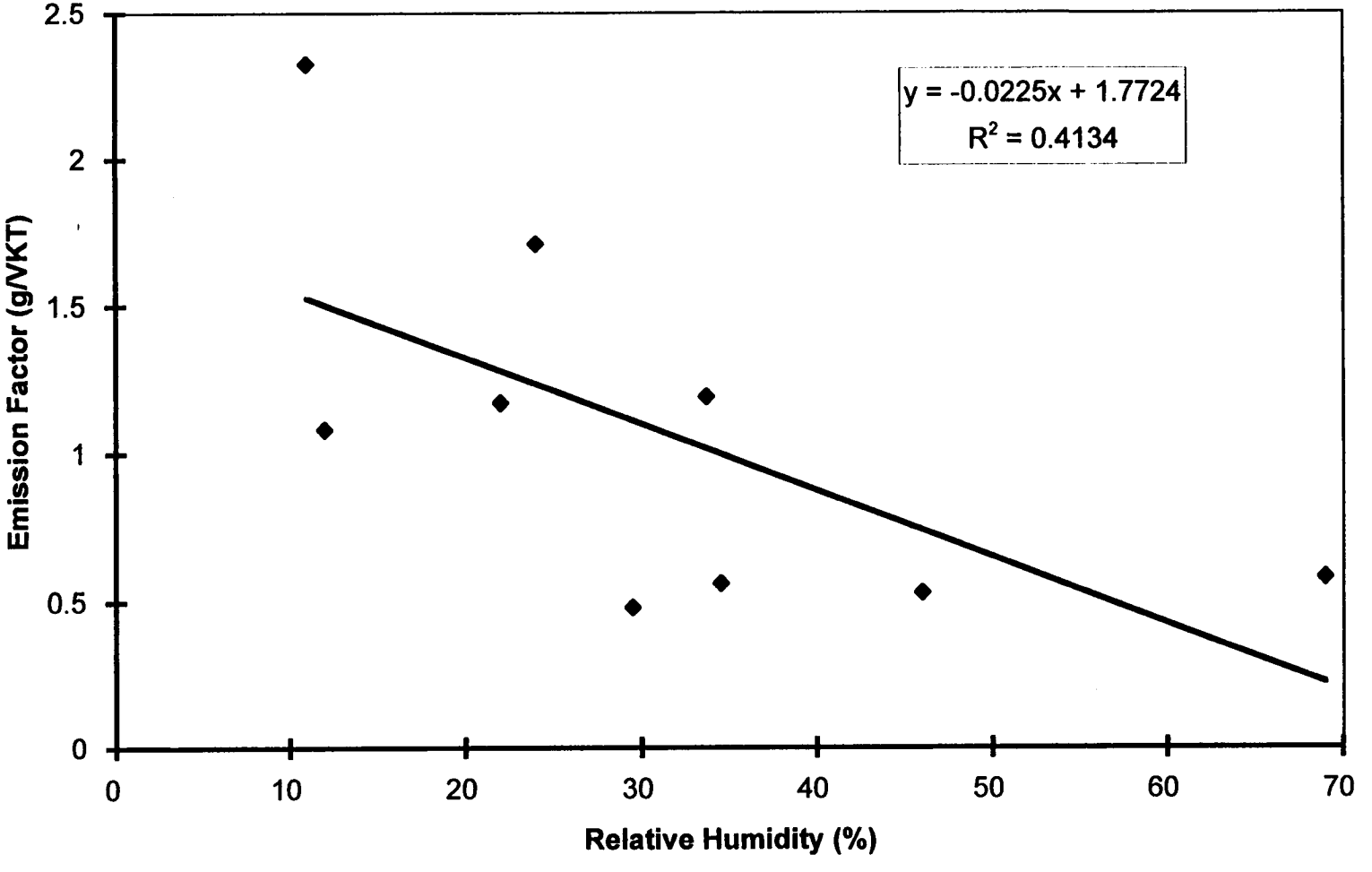

Figure 5. Observed correlations between relative humidity and emission factor; 11/18/93 - 7/11/94 unsanded expts.

TABLE 3: SILT LOADING MEASUREMENTS

Date[1]	Silt Load[2] (g/m^2)	Calculated Emission Factor (g/VKT)	Observed Emission Factor (g/VKT)
5/10/94	3.18 ± 0.08	10.02	1.71[3]
6/01/94	3.56 ± 0.38	10.96	0.76[4]
6/08/94	3.26 ± 0.78	10.22	0.47[5]
6/15/94	3.36 ± 1.04	10.47	—
6/23/94	3.23 ± 1.17	10.14	1.17[6]
7/14/94	2.71 ± 1.46	8.81	—
		10.1 ± 0.65	

[1] Country Homes Blvd. was swept on 6/7/94
[2]The AP-42 reported silt loads range between 0.16 - 0.82gm^{-2} (US EPA, 1985)
[3]The experiment was performed on 5/12/94
[4]The experiment was performed on 6/3/94
[5]The experiment was performed on 6/10/94
[6]The experiment was performed on 6/21/94

Sample collection was coordinated with tracer experimental runs, to identify surface conditions as close as possible to those experienced during point source tracer release tests. Street sweeping exercises, using a regenerative-air vacuum sweeper, were performed on 6/7/94. Measurements on the road indicate a consistent silt loading, with an observed overall standard deviation of less than 10%. Corresponding emission factors are exceptionally high compared to the AP-42 default value of 1.8g/VKT. No correlation was found between experimentally calculated factors and silt loading observations.

Mobile Point Source Release Experiments

Five experiments (2/15, 2/18, 4/5, 4/8, and 4/15/94) were conducted on unpaved roads near Washington State University. Due to problems resulting from an inadequate SF_6 release rate (25cc/min), resultant upwind and downwind tracer concentrations were comparable for only the 2/15 and 2/18/94 experiments. As a result the release rate was increased to 235cc/min for all subsequent experiments. Table 4 summarizes the emission factors and meteorological data observed during these experiments. Also included are empirically deduced values using the Fitzpatrick formula (Fitzpatrick, 1987), as well as values presented by Claiborn et al. (1995) for comparison. Results indicate that the unpaved road sites were emitting at different PM_{10} rates during each of the three experimentation periods. The primary observed meteorological difference between these three periods is the precipitation history recorded in the area. During the three days prior to the 4/8/94 experiment, Pullman recorded 8mm of precipitation. No

TABLE 4: UNPAVED ROAD TRACER RELEASE EXPERIMENTS

Date	Site[1]	Humidity (%)	Temp. (^oC)	Precip. History[2] (mm/72hrs)	Soil Moisture (%)	WS[3] (m/s)	Wind Angle w/ road	Calc'd Emission Factor (g/VKT)	Published Emission Factor[4] (g/VKT)	Comparison To Published Factor[5]
4/5/94	CC Left	35	13	0.85	0.85	4.7	45^o	586	227	258.1%
4/8/94	CC Main	35	24	8.0	0.73	4.3	90^o	70	227	-69.2%
4/15/94	CC Main	20	21	0.17	___	3.0	90^o	228	227	0.4%
9/21/92	55th Street	___	___	___	___	2.6	109^o	336	283	18.7%
9/22/92	55th Street	___	___	___	___	3.	41^o	140	283	-50.5%

[1] Mobile tracer release experiments were conducted on CC Left and Main, near Pullman, WA. 55th Street, Spokane, WA experiments are referenced from Claiborn et al., (1995). These were conducted using a line source release system.
[2] Precipitation History recorded at the Laboratory for Atmospheric Research, approximately three miles from the experimental sites.
[3] Wind Speed in meters per second
[4] Published factors were determined from U.S. EPA, AP-42 documentation.
[5] Percentage difference is calculated from ((calculated factor-published factor)/published factor)*100.

further contributing correlation can be deduced between the weather parameters and emission factors recorded on 4/5 and 4/15/94.

Model Analysis

Using data obtained from all the unsanded Country Homes Blvd. experiments (except for the 6/3/94, 5/21/95 and 11/18/93 experiments which were omitted in this analysis due to incomplete wind and/or temperature gradient data), SIMFLUX Gaussian plume simulations were performed. Calibration of the model was conducted using experiment specific tracer concentration and wind data. Only those hours which were observed to have good plume profiles, and which were subsequently used to calculate the reported emission factors, were modeled. Stability criteria were determined through temperature gradient determination, and confirmed through wind direction standard deviation observations. Atmospheric classification tables were obtained from Zannetti (1990). Having deduced the necessary volume source dimensions describing the mechanical mixing process, measured downwind PM_{10} concentrations were used to estimate the fugitive mass emission rate released from the road. Table 5 provides information on all the relevant parameters used and corresponding model calibration results. Fractional bias (FB), fractional standard deviation (FS) and normalized mean square error (NMSE) statistical tests were used to assess the performance of the modeled point source release predictions. Differences between predicted and experimentally observed crosswind integral concentrations (CWI) were also appraised. Table 6 summarizes the observed differences in experimental and model deduced emission factors.

TABLE 5: MODEL CALIBRATION PARAMETERS AND RESULTS

Date[1]	Y, H[2] (m,m)	WS[3] (ms^{-1})	Wind angle w/ road	Temp. Grad.[4] ($^oCm^{-1}$)	Stabil. Class	Mean Co[5] (μgm^{-3})	Mean Cp[6] (μgm^{-3})	Max. Co (μgm^{-3})	Max. Cp (μgm^{-3})	FB[7]	FS[8]	M[9]
2/24/94	30,0	3.8	61 ± 31	-0.02	A	0.495	0.25	0.923	0.4	0.658	0.692	0.821
	15,0	4.0	73 ± 17	-0.0155	C	0.683	0.5	1.437	1.0	0.31	0.347	0.28
	15,0	3.6	70 ± 17	-0.0155	C	0.865	0.54	1.79	1.0	0.462	0.35	0.394
	20,0	2.8	71 ± 21	-0.018	B	1.33	0.73	2.53	1.3	0.583	0.63	0.547
3/29/94	30,0	3.4	18 ± 34	-0.021	A	0.304	0.182	0.791	0.3	0.502	0.633	0.568
	40,0	3.6	46 ± 18	-0.0174	B	0.448	0.408	0.844	0.8	0.093	-0.14	0.169
	20,0	3.2	34 ± 29	-0.0193	A	0.293	0.191	0.496	0.4	0.423	0.011	0.453
4/18/94	20,0	4.2	44 ± 18	-0.018	B	0.714	0.44	1.217	1.0	0.475	0.176	0.306
	30,0	4.9	49 ± 14	-0.016	C	0.791	0.48	1.306	1.1	0.489	0.083	0.436
	50,0	5.6	54 ± 8	-0.008	D	0.992	0.58	1.738	1.7	0.523	0.003	0.371
	30,0	5.7	53 ± 8	-0.007	D	1.072	0.73	2.33	1.6	0.379	0.216	0.269
	40,0	4.4	46 ± 4	-0.003	E	1.485	0.91	3.17	1.8	0.481	0.366	0.378
5/12/94	30,2	7.1	52 ± 8	-0.01	D	0.337	0.333	0.91	1.0	0.012	-0.27	0.491
	20,3	6.1	48 ± 18	-0.0182	B	0.333	0.267	0.75	0.7	0.223	-0.05	0.215
	38,0	6.6	47 ± 20	-0.018	B	0.35	0.275	0.751	0.6	0.24	-0.005	0.159
	33,5	6.2	38 ± 9	-0.012	D	0.295	0.3	0.7	0.6	-0.01	0.048	0.098
	32,0	6.3	39 ± 14	-0.016	C	0.339	0.333	0.767	0.7	0.02	0.028	0.107

[1] For each date, modeled results correspond to hourly experimental periods.
[2] These parameters correspond to initial horizontal and vertical plume dimensions.
[3] Hourly wind speed in meters per second.
[4] Temperature gradient.
[5] Mean observed SF_6 concentration for each experimental hourly period.
[6] Mean predicted SF_6 concentration for each experimental hourly period using the SIMFLUX model.
[7] Fractional Bias = 2((mean Co - mean Cp)/(mean Co + mean Cp))
[8] Fractional Standard Deviation = 2((mean σ_o - mean σ_p)/(mean σ_o + mean σ_p))
[9] M = (mean Co - mean Cp)/(mean Co. mean Cp)

TABLE 5 (continued): MODEL CALIBRATION PARAMETERS AND RESULTS

Date	Y, H (m,m)	WS (ms^{-1})	Wind angle w/ road	Temp. Grad. ($^oCm^{-1}$)	Stabil. Class	Mean Co (μgm^{-3})	Mean Cp (μgm^{-3})	Max. Co (μgm^{-3})	Max. Cp (μgm^{-3})	FB[1]	FS[2]	M[3]
6/10/94	50,0	1.36	61 ± 54	-0.023	A	0.804	0.82	1.23	1.4	-0.02	-0.026	0.072
	120,0	1.78	75 ± 76	-0.03	A	0.472	0.46	0.72	0.8	0.025	0.186	0.635
6/21/94	6,10	1.66	40 ± 72	-0.021	A	0.329	0.392	1.645	1.0	-0.17	0.312	0.688
	35,1.5	1.49	10 ± 73	-0.02	A	0.542	0.567	1.48	1.3	-0.05	0.169	0.151
	90,30	1.06	33 ± 89	-0.021	A	0.176	0.208	0.51	0.4	-0.17	0.223	0.537
	25,34	0.8	49 ± 84	-0.024	A	0.121	0.142	0.387	0.3	-0.15	0.27	0.794
7/11/94	50,2	5.4	38 ± 27	-0.025	A	0.2	0.21	0.75	0.4	-0.04	0.586	0.733
	70,0	5.4	33 ± 32	-0.024	A	0.302	0.236	0.243	0.669	0.243	0.668	1.1
	50,0	4.8	25 ± 25	-0.026	A	0.25	0.218	1.27	0.6	0.136	0.765	1.467
	50,10	3.8	2 ± 34	-0.025	A	0.119	0.127	0.96	0.2	-0.07	1.129	4.013

[1] Fractional Bias = 2((mean Co - mean Cp)/(mean Co + mean Cp))
[2] Fractional Standard Deviation = 2((mean σ_o - mean σ_p)/(mean σ_o + mean σ_p))
[3] M = (mean Co - mean Cp)/(mean Co. mean Cp)

TABLE 6: COMPARISON OF OBSERVED AND PREDICTED EMISSION FACTORS

Date[1]	Observed Emission Rate[2] ($\mu gm^{-1}s^{-1}$)	Predicted Emission Rate[3] ($\mu gm^{-1}s^{-1}$)	Observed Emission Factor ($gV^{-1}K^{-1}T^{-1}$)	Predicted Emission Factor[4] ($gV^{-1}K^{-1}T^{-1}$)	% Difference[5]
2/24/94	209.7	330	0.85	1.38	62.3%
	443.6	530	1.32	1.82	38.1%
	287.9	375	0.82	1.11	35.4%
	378.8	657	0.95	1.65	73.7%
3/29/94	375.2	526	1.34	1.88	40%
	332.9	400	1.01	1.21	19.8%
	397.3	500	1.04	1.31	30%
4/18/94	278.4	360	0.95	1.23	29.5%
	241.0	360	0.75	1.12	49.3%
	174.5	250	0.49	0.7	42.8%
	139.9	180	0.33	0.43	30.3%
	97.7	135	0.27	0.37	37%
5/12/94	552.2	600	2.11	2.29	8.5%
	326.8	400	1.28	1.42	10.9%
	379.9	450	1.35	1.6	18.5%
	722.9	682	1.96	1.84	-6.1%
	703.9	722	1.87	1.92	2.7%
6/10/94	156.0	170	0.38	0.42	10.5%
	337.9	423	0.83	1.04	25.3%
6/21/94	462.2	402	1.55	1.35	12.9%
	343.0	543	1.06	1.68	58.5%
	645.5	760	1.68	1.98	17.9%
	179.0	157	0.41	0.36	-12.2%
7/11/94	930.1	1336	3.02	4.34	43.7%
	568.4	836	1.97	2.9	47.2%
	403.6	695	1.23	2.12	72.3%
	993.6	2350	3.08	7.29	136.7%

[1] For each date, modeled results correspond to hourly experimental periods.
[2] PM_{10} Emission rates observed during each hourly period.
[3] PM_{10} Emission rates predicted using a line source model of the 4-lane road environment.
[4] Predicted emission factors are based on hourly observed traffic volumes on the road.
[5] % Difference = ((Predicted - Observed)/Predicted)*100

DISCUSSION

Stationary Point Source Release Experiments

Results indicate an overall 'baseline' emission factor of 1.04 ± 0.6g/VKT (Table 1). This is approximately 42% lower than the AP-42 reference value. The observed variance is associated with an inversely proportional relationship between relative humidity and emission factor (R^2= 0.413).

Humidity effects are strongly associated with the accumulation range of the fine particulate mode (Seinfeld, 1986; McMurry and Stolzenburg, 1989). Paved road dust chemical species include not only the expected coarse geological components (Al, Si, Fe, Ca) but also significant organic carbon, elemental carbon and to a minor extent SO_4^{-2}, NO_3^- and NH_4^+ contributions (Chow et al., 1990; Chow et al., 1992; Chow 1995). While SO_4^{-2}, NO_3^- and NH_4^+ are the primary water soluble and hygroscopic materials (Hanel and Lehmann, 1981, Charlson et al., 1984), Countess et al. (1981) also observed organic and elemental carbon mean size growth, but only at humidities greater than 70%.

As the relative humidity increases, water vapor condenses on the particles. Whether the suppressant nature of this occurrence is increased particle mass or hydrogen bonding induced (particles attract and bond due to water molecular forces), it is important to note that the effect relies heavily on the presence of a strong SO_4^{-2}, NO_3^- and NH_4^+ or other hygroscopic

component in the road dust. To our knowledge, similar correlations between PM_{10} emission rates and relative humidity have not been reported in literature. Further research is warranted to describe the ongoing microscopic processes.

The importance of traction material resuspension was observed, with a corresponding emission factor of 1.45 ± 0.37g/VKT (Table 2). Unlike the unsanded condition, no correlation was observed with relative humidity. Under this sanding material environment, PM_{10} emissions would be expected to exhibit a larger coarse, geological component, which may explain the non-existent humidity correlation. The overall emission factor calculated here resulted from traction material that is coarser than current Washington State traction implementation practices (i.e., 3/8" and less, as compared to 1/4" and less). The City of Spokane Department of Transportation utilizes the larger sized aggregates with the intent that they will produce lower emissions than the finer traction material (1/4"). Hence, it is possible to interpret this general situation as representing a best case scenario for the State as a whole. With respect to the experimental observations, it is important to note that current WSDOE emission inventory estimates, for similar roads under unsanded and sanded conditions, are approximately 0.3g/VKT and 1.2g/VKT respectively (WSDOE, 1993). Considering this class of road accounts for approximately 90% of the annual paved road PM_{10} emissions, a re-alignment of attainment goals may be necessary.

Silt Loading Measurements

Measured mean silt loads for the road were consistent throughout the summer with less than a 10% variation (Table 3). Corresponding calculated emission factors were erroneously high, with values consistently in the range of 10g/VKT. These results concur with Zimmer et al. (1992), who also observed that the empirical equation tends to overpredict paved road emissions. It is important to note that the AP-42 equation was developed from a limited data base that did not incorporate the impact of street conditions, public works practices and seasonal silt load variations. Furthermore, for this class of road, it is bounded within a range of silt loads of 0.16-0.82g/m^2 (U.S. EPA, 1985).

As Table 3 illustrates, measurements taken the day after a street sweep provided negligible evidence of any notable silt load reduction. This observation is consistent with the results of other street sweeping studies, which also noted insignificant particulate reductions (U.S. EPA, 1981; Chow et al., 1990). Additionally, where particulate mass concentration reductions have been noted after street sweeping studies, site meteorological variability has been identified as probably having a notable contributing effect (Cuscino et al., 1983). The poor correlation (R^2 = 0.43) between emission factor and the time each experiment was performed after a street sweep for relative humidity 30% or higher further strengthens this observation.

Figure 4 indicates that the time component, as compared to the relative humidity variable, plays a limited role in the observed relationship. Multiple regression analysis of the relative humidity

and time variables results in an increased R^2 ($R^2 = 0.413 \rightarrow 0.512$) and adjusted R^2 correlation ($R^2_{adj} = 0.33 \rightarrow 0.349$). The increased correlation indicates that inclusion of the time variable is somewhat statistically significant. Since it has been observed that the emission factor/time of street sweep relationship has a strong humidity dependence, then we can infer this increased multiple regression correlation as further strengthening the argument for there being a dominant relationship between emission factor and relative humidity. The split in data trends at a relative humidity of approximately 30% may indicate that above this value the previously described mass/hydrogen bonding suppression effect may have a more significant impact on resulting emissions (Figure 4).

As shown in Table 3, no correlation between observed emission factors and those calculated from silt loadings, can be defined. This may be due to a poor correlation between the road silt content (<75μm) and PM_{10}. This argument is further augmented by the observed relationship between relative humidity and emission factor, which infers that the typical measured silt sizes are not representative of the actual ongoing physical/emission processes.

Street Sweeping Inadequacies

Chow et al. (1990) report negligible resuspended particulate reductions to stem from regenerative-air vacuum sweeper inadequacies. The system uses redirected exhaust air to dislodge surface compacted dirt particles. The air is forced vertically on to the pavement and all subsequent entrained particles are sucked up at the pickup head and transported to a collection

hopper. Chow et al. (1990) identify the importance of the seal between the pickup head and the pavement surface. Since road surfaces are generally uneven, the seal is hard to maintain. Hence, it was noticed that a portion of the blast air tended to escape into the atmosphere for re-deposition on the road surface. Since the Spokane County Road Maintenance Department uses a similar machine, site observations could be attributable to these sweeping deficiencies.

Mobile Point Source Release Experiments

Emission factors of 586, 70 and 228g/VKT were observed during this experimentation period. The low factor can be explained by the considerable precipitation experienced during the three days prior to the experiment. Determining a correlation between meteorological parameters and the emission factors observed on the 4/5 and 4/15/94 experiments, however, is not so apparent. Although the experiments were conducted at different sites, and hence may describe different site conditions, they were nevertheless only a quarter of a mile apart and located on connecting roads. The only observation of note concerns the differing wind speeds. During the 4/5 experiment, wind speeds were approximately 60% higher than the 4/15 wind conditions. It is conceivable that this higher wind speed aided in the erosion and lifting of unpaved road top soil, with that produced by vehicular disturbance. Within the constraints imposed by the variable site conditions, the experimental factors compared reasonably well with the empirically calculated emission factors.

SIMFLUX Model Evaluation

Fractional differences for the "bias" and for deviations between observed and predicted "scatter" are presented in Figure 6. A positive difference with respect to the zero intersection, indicates model underprediction while a negative difference indicates overprediction. Cox and Tikvart (1985) note that fractional differences between ± 0.67 are equivalent to an accuracy within a 'factor of two'. Accordingly, even though the model tends to underpredict both the average and scatter of the observed data, the results do predominantly fall within the set constraints (i.e., the horizontal and vertical lines bounding this range). NMSE analyses provide further quantification of model simulation accuracy. Hanna et al. (1985) state that NMSE valuations near 0.4 indicate good prediction capabilities, while values near 4 reveal very poor model performance. Results from all the dates, neglecting those that fall outside the ± 0.67 criteria, indicate an overall NMSE average of 0.48 ± 0.28. Although this, as well as the FB/FS analyses, provide validation of the model to predict within a factor of two, a better representation of model performance can be provided by looking at the difference between predicted and observed crosswind integral concentrations. An overview of the results indicate an average difference of -21% ± 27%. The underprediction may be explained in terms of the inability of the model to accurately simulate vehicular wake generated turbulence as a function of wind direction and wind speed relative to the road.

It is interesting to note that typical volume source dimensions were greater than those reported in Claiborn et al. (1995). This may result from greater vehicular induced turbulence, due to higher

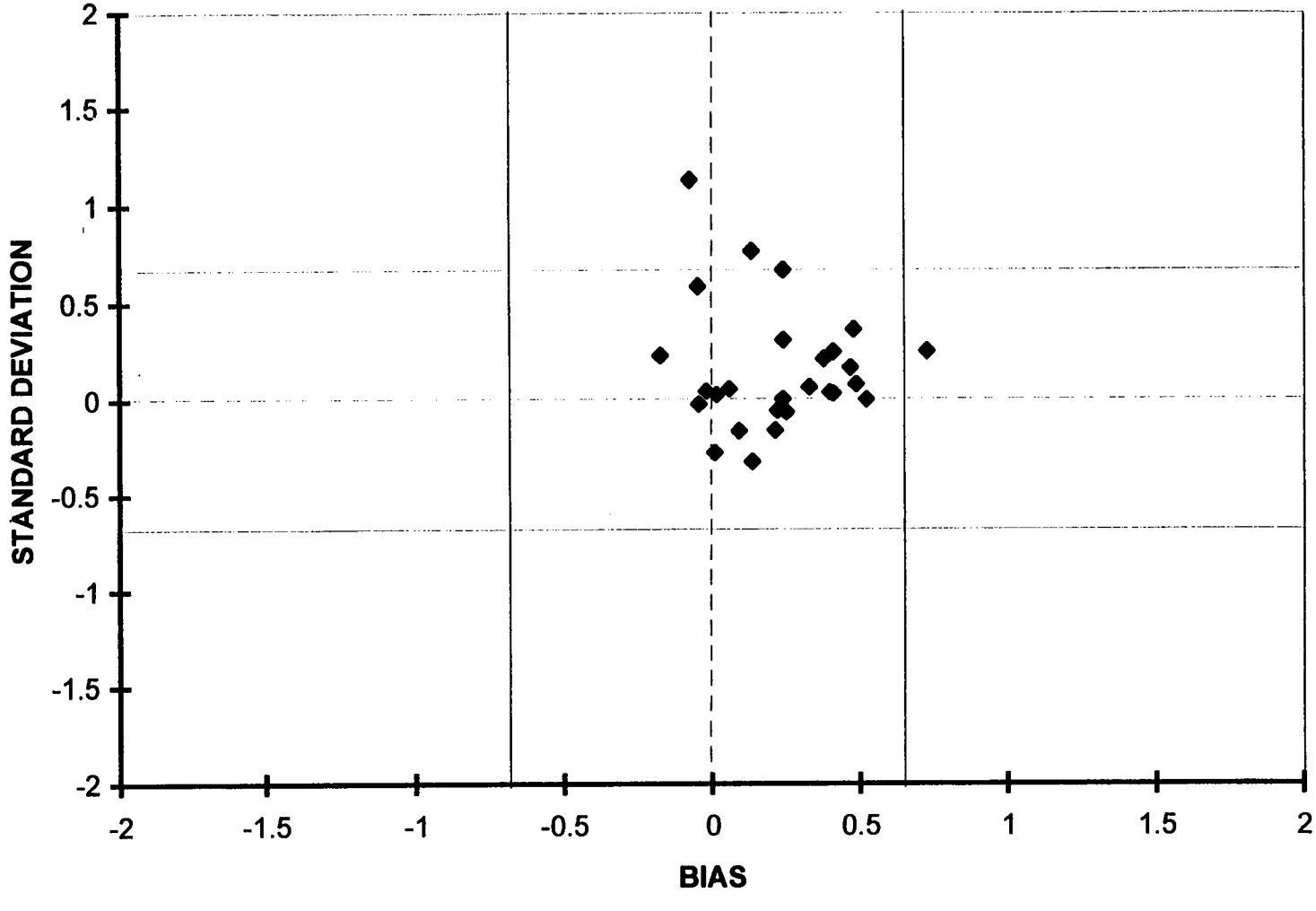

Figure 6. Fractional differences between observed and predicted concentrations.

traffic volumes, on these modeled streets. When wind speeds were low (6/10/94 and 6/21/94 experiments), vertical dispersion was seen to play a greater role in tracer diffusion across the road (i.e. large "H" volume source dimensions). Conversely, under high wind speed conditions lateral dispersion was seen to dominate tracer diffusion. Application of the calibrated dispersion coefficients to a line source release produced reasonably good results. The tendency of the model to underpredict near field dispersion further compounds the errors observed in back-calculated emissions estimates (Table 6). Here, although results are overpredicted, they are nonetheless inside the general factor of two margin and hence useful, according to regulatory requirements, in providing a "best-estimate" (Cox and Tikvart, 1985). Under a line source approach only vertical dispersion parameters are utilized, since it is assumed that the lateral dispersion from one segment of the line is compensated by dispersion in the opposite direction from adjacent segments. Hence, it can be hypothesized that near field dispersion from a line source, under generally high wind speeds ($> 3 ms^{-1}$), undergo limited vertical diffusion. Under low wind speed conditions, vertical dispersion, resulting from high vehicular turbulence effects from the heavy traffic volume, may dominate roadway particulate releases.

CONCLUSIONS AND RECOMMENDATIONS

The results from this study are consistent with those of the previous WSDOT-sponsored project in demonstrating the effectiveness of tracer techniques in estimating roadway particulate emissions. The increasing data base of emission factors, derived from various roadway classifications in the first report and in this report, provide important answers to the many

uncertainties afflicting emission inventory construction. The results of this study indicate that paved road emissions from 4-lane commercial/residential streets, with traffic volumes exceeding 10,000 vehicles/day, are generally overestimated by the U.S. EPA approved empirical algorithm (U.S. EPA, 1985). Under unsanded and sanded conditions, the experimentally observed emissions were 42% and 20% lower than the reference value, where the reference value specifically characterizes the unsanded case. Current methods to predict the sanded condition are highly suspect, in fact current U.S. EPA guidance stipulates applying a factor of four to the empirical value. The observations here represent, to our knowledge, the first set of consistent data detailing a sanded environment. Results indicate that sanding does not increase emissions by a factor of four, but by approximately 40%, under these site specific conditions. It should be noted that the emission factor for paved roads with traction sand applied will be strongly dependent on the nature of the sand used, and since these experiments were conducted with only one kind of material, these results are not expected to be applicable to all traction sands. The impact of these general observations on emission inventory evaluation and subsequent State Implementation Plan (SIP) design is considerable. Tentative predictions based on the Spokane SIP results indicate a tripling of the annual emissions from paved roads (WSDOE, 1993, Appendix H).

For comparison, site specific information, namely the silt loading on the paved road, was collected using a method described in a personal communication with Greg E. Muleski (Midwest Research Institute, 4/26/94), and in the MRI Open Source PM_{10} Method Evaluation Report (U.S. EPA, 1991). Lateral strips of known area were vacuumed at specified distances using a

portable vacuum cleaner. The attached brush on the collection inlet was used to abrade surface compacted dust and to remove dust from the crevices of the road surface. Care was taken to ensure that samples were only obtained from the general area over which the wheels and carriages traveled. It is important to note that no information is given as to the force to which the surface should be abraded with the brush. Hence, the likelihood of differing results depending on the person obtaining the data is considerable. For example, results presented in Table 3 are quite high compared to WSDOE results for the same road (Country Homes Blvd.). In appendix K of the Spokane SIP (WSDOE, 1993), silt loads were typically 99% lower than our observations. The example here clearly illustrates one of the uncertainties associated with this empirical algorithm. Another area of concern stems from the general data base from which the algorithm was formed, which does not truly represent all roadway conditions. Considering the uncertainties associated with the silt loading-based equation, as observed in this study and in that conducted by Zimmer et al. (1992), we recommend that the silt loading-based algorithm is NOT used to predict PM_{10} emissions from roads. Since it was observed that the silt content size fraction ($\leq 75\mu m$) does not characterize PM_{10} emissions, the use of tracer techniques provide more accurate emission factors.

Relative humidity correlations were observed with unsanded emission factors. The reason for this effect is still unclear. It is hypothesized that as a result of water vapor condensation on the aerosol particulates, the correlation is either mass or hydrogen bonding influenced. If so, the overall effect seems to be dependent on a strong accumulation mode component (SO_4^{-2}, NO_3^- and NH_4^+). It was not possible to detect, using our limited analysis techniques, whether this

was the case or not. However, it is hoped that with the continued air sampling occurring at the site 1 SCAPCA Station, sufficient data, in the form of XRF, chemical mass balance analyses etc., will soon be present to assess the viability of this hypothesis. Streetsweeping tends to slightly reduce PM_{10} emission rates when the relative humidity is above 30%.

Experiments conducted in a controlled atmospheric chamber to track relative humidity and filter mass correlations proved unsuccessful. It was felt that the gravimetric assessment of mass differentials in the filter, through microbalance observations, was inadequate. The effects of particle size could be seriously altered, both by the presence of a surface and by the particles contacting other particles. Also, the surface particle contact could afford a site for capillary condensation that does not exist in the airborne state. To our knowledge, no data currently exist concerning this humidity/emission factor phenomenon.

Sanded experiments on Division Street indicated no correlation with relative humidity. It is assumed that the particulate emissions from the roadway and traction sand sources have an increased coarse, geological component. Since humidity effects are strongly associated with the fine fraction, the general coarse component increase is hypothesized to negate the humidity effects. During the 3/16/95 experiment, PM_{10} and $PM_{2.5}$ low-volume samplers were positioned at upwind and downwind road sites. If the ratio of PM_{10} to $PM_{2.5}$ increases after sanding then it would be possible to interpret this condition as quantifying our hypothesis. Since the low-volume samplers could not handle the increased particulate conditions (i.e., the impaction head was too small to cut off all particulates greater than $PM_{2.5}$), results were dismissed. Better

observations and results would be expected with the use of co-located PM_{10} and $PM_{2.5}$ TEOM samplers at upwind and downwind sites. Although limited resources prevented this experimental approach, it is hoped that future work will corroborate this hypothesis.

Mobile point source experiments were conducted on unpaved roads near Pullman. General observations tended to indicate that the use of stationary releases proved invalid as the traffic volume decreased. In effect it was considered that under these conditions the road did not behave as a continuous line source. Hence, at traffic volumes less than or equal to 60 vehicle/hour, mobile tracer releases may provide a better description of the roadway condition. Primary goals were set up so that the experiments would encompass unpaved road conditions ranging from frozen, to thawed, to dry. However, due to mild winter conditions, such objectives were unobtainable, however significant suppression of PM_{10} emissions following a precipitation event was observed. It is important that further work be conducted during more harsh conditions to help quantify the ongoing processes, and build on this limited data base.

Modeling of the unsanded experiments on Country Homes Blvd. using the SIMFLUX Dispersion model produced results in agreement with current model expectations (i.e., within a "factor of two"). Tracer and site observed meteorological data were used to calibrate the model, while upwind/downwind PM_{10} data was used to back-calculate paved road emission factors. Lateral and vertical dispersion were found to dominate roadway diffusion characteristics during periods of high and low wind speed, respectively. Although results were satisfactory, their application to an emission inventory, sensitive to the emission factor applied, could prove

problematic. The use of tracer techniques provide the most accurate emission factors for inventory and subsequent SIP development.

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REFERENCES

American Meteorological Society (1978), Accuracy of dispersion models: A position paper of the AMS Committee on atmospheric turbulence and diffusion, *Bulletin of the American Meteorological Society*, **59**: 1025-26.

Black F.M, Braddock J.N., Bradow R.L., Hare C.T., Ingalls M.N., Kawechki J.M., (1983), Study of particulate emissions from motor vehicles, U.S. Environmental Protection Agency, Research Triangle Park, NC.

Cadle S.H., Chock D.P., Monson P.R., Heuss J.M., (1977), General Motors sulfate dispersion experiment: experimental procedures and results. *J. Air Pollut. Control Assoc.* **27**, 33-38.

Charlson R.J., Covert D.S., Larson T.V., (1984), Observation of the effect of humidity on light scattering by aerosols, *Hygroscopic Aerosls* (edited by Ruhnke L.H. and Deepak A.), 35-44, A. Deepak Publishing.

Chock D.P., (1978), A simple line-source model for dispersion near roadways, *Atmospheric Environment*, **12**:823-829.

Chow J.C., Watson J.G., Egami R.T., Frazier C.A., Lu Z., Goodrich A., Bird A., (1990), Evaluation of regenerative-air vacuum street sweeping on geological contributions to PM_{10}, *J. Air Waste Manage. Assoc.* 40:1134-1142.

Chow J.C., Liu C.S., Cassmassi J., Watson J.G., Lu Z., Pritchett L.C., (1992), A neighborhood-scale study of to PM_{10} source contributions in Rubidoux, California, *Atmospheric Environment*, **26A**:693-706.

Chow J.C., Watson J.G., Lowenthal D.H., Frazier C.A., Hinsvark B.A., Pritchett L.C., Neuroth G.R., (1992), Wintertime PM_{10} and $PM_{2.5}$ chemical compositions and source contributions in Tuscon, Arizona, in *Transactions, PM_{10} Standards and Nontraditional Particulate Controls* (edited by Chow J.C., and Ono D.M.), 311-323. Air Waste Management Assoc., Pittsburgh, P.A.

Chow J.C., Watson J.G., Lowenthal D.H., Soloman P.A., Magliano K., Ziman S., Richards L.W., (1992), PM_{10} source apportionment in California's San Joaquin Valley, *Atmospheric Environment*, **26A**:3335-3354.

Chow J.C., Watson J.G., Ono D.M., Mathai C.V., (1993), PM_{10} standards and nontraditional particulate source controls : a summary of the AWMA/EPA international specialty conference, *J. Air Waste Manage. Assoc.* **43**:74-84.

Chow J.C., (1995), Measurement methods to determine compliance with ambient air quality standards for suspended particles., *J. Air Waste Assoc.* 45:320-382.

Claiborn C., Mitra A., Adams G., Bamesberger L., Allwine E., Kantamaneni R.K., Lamb B., Westberg H. (1995), Evaluation of PM_{10} emission rates from paved and unpaved roads using tracer techniques, *Atmospheric Environment,* **29**:1077-1089.

Countess R.J., Cadle S.H., Groblicki P.J., Wolff G., (1981), Chemical analysis of size-segregated samples of Denver's particulate, *J. Air Pollut. Control Assoc.*, 31:247-252.

Cowherd C., (1988), Gap filling PM_{10} emission factors for selected open area dust sources, *EPA/450/4-88/003.*

Cox W.M., Tikvart J.A., (1985), Assessing the performance level of air quality models, presented at the 15th International Technical Meeting on Air Pollution Modeling and its Applications, NATO/CCMS, Vol. 3.

Cuscino T., Muleski G.E., Cowherd C., (1983), Determination of the decay in control efficiency of chemical dust suppressants, Proceedings - Symposium on Iron and Steel Pollution Abatement Technology for 1982, U.S. EPA, Research Triangle Park, NC.

Fitzpatrick M., (1987), Emission control technologies and emission factors for unpaved road fugitive emissions, User's guide, *EPA/625/5-87/022.*

Hanel G. and Lehmann M., (1981), Equilibrium size of aerosol particles and relative humidity: New experimental data from various aerosol types and their treatment for cloud physics application, *Beitr. Phys. Atmosph.* 54:57-71.

Howard T., Lamb B.K., Bamesburger W.L., Zimmerman P.R., (1992), Measurement of hydrocarbon emissions fluxes from refinery waste-water impoundment's using atmospheric tracer techniques, *J. Air Waste Manage. Assoc.* **42**:1336-1344.

Kim B.M., Lewis R., Hogo H., Chow J.C., (1992), Source apportionment by chemical mass balance: A comparison between measured source profiles and SAFER model estimated source profiles, in *Transactions, PM_{10} Standards and Nontraditional Particulate Controls* (edited by Chow J.C., and Ono D.M.), 311-323. Air Waste Management Assoc., Pittsburgh, P.A.

Kinsey, J.S., (1993), Characterization of PM_{10} emissions from antiskid material applied to ice and snow covered roadways, *EPA/600/R-93/019.*

Kitas V., Lioy P.J., (1992), Near field dispersion of mechanically resuspended dust from an unpaved road, in *Transactions, PM_{10} Standards and Nontraditional Particulate Controls* (edited by Chow J.C., and Ono D.M.), 382-398. Air Waste Management Assoc., Pittsburgh, P.A.

McMurray P.H. and Stolzenburg M.R., (1989), On the sensitivity of particle size to relative humidity for Los Angeles aerosols, *Atmospheric Environment* 23:497-507.

Muleski G.E., Stevens K., PM_{10} emissions from public unpaved roads in rural Arizona, in *Transactions, PM_{10} Standards and Nontraditional Particulate Controls* (edited by Chow J.C., and Ono D.M.), 324-334. Air Waste Management Assoc., Pittsburgh, P.A.

Okamoto S., Kobayashi K., Ono N., Kitabayashi K., Katatani N., (1990), Comparative study on estimation methods for NOx emissions from a roadway. *Atmospheric Environment* **24A**:1535-1544.

Parisi J., Boyum J., Griffin G., Yoshida C., (1993), Cottage grove PM_{10} saturation study., report by the Lane Regional Air Pollution Authority, Springfield, OR. March 1993.

Peterson W.B., (1980), User's guide for HIWAY-2. A highway air pollution model, *EPA-600/8-80-018.*

Ringler E.S., Shrieves V.X., Berg N.J., (1993), The summer 1992 PM_{10} saturation monitoring study in the Ashland, KY area.

Seinfeld J.H., (1986), *Atmospheric Chemistry and Physics of Air Pollution.* John Willey, New York.

U.S. Department of Transport and Washington State Transportation Commission (1992), Analysis of particulate matter dispersion near urban roadways, Report #GC8719, March 1992.

U.S. Environmental Protection Agency (1981), Study of street cleaning impact on particulate levels in Kansas City, Kansas, prepared by PEDCo Environmental, Inc. under contract No. 68-02-2535.

U.S. Environmental Protection Agency (1985), Compilation of air pollutant emission factors, Volume I : stationary point and area sources, AP-42., U.S. EPA, Office of Air Quality Planning and Standards, Research Triangle Park, NC.

U.S. Environmental Protection Agency (1987), Protocol for reconciling differences among receptor and dispersion models, U.S. EPA, Air Management Technology Branch, Monitoring and Data Analysis Division, Research Triangle Park, NC.

Wang I.T., Countess R.J., Umenhofer T.A., Farber R.J., Mirabella V., (1992), Adaptation of the ISCST model for local fugitive PM_{10} modeling with application to the Riverside/Rubidoux area, in *Transactions, PM_{10} Standards and Nontraditional Particulate Controls* (edited by Chow J.C., and Ono D.M.), 357-381. Air Waste Management Assoc., Pittsburgh, P.A.

Washington State Department of Ecology (1993), Air Quality Program: Spokane PM_{10} attainment plan, WSDOE, Olympia, WA.

Washington State Department of Ecology (1993), Air Quality Program: Spokane PM_{10} attainment plan, Appendix H and K, WSDOE, Olympia, WA.

Washington State Department of Ecology (1994), Spokane PM_{10} attainment plan: Public review draft, WSDOE, Olympia, WA.

Watson, J.G., (1984), Overview of receptor model principles. *J. Air Pollut. Control Assoc.* **34**: 619-623.

Watson J.G., Chow J.C., Mathai C.V., (1989), Receptor models in air resources management : a summary of the APCA international specialty conference., *J. Air Waste Assoc.* **39**:419-426.

Zannetti P., (1990), *Air Pollution Modeling: Theories, Computational Methods and Available Software*, Van Nostrand Reinhold, New York.

Zimmer R.A., Reeser W.K., Cummins P., (1992), Evaluation of PM_{10} emission factors for paved streets, in *Transactions, PM_{10} Standards and Nontraditional Particulate Controls* (edited by Chow J.C., and Ono D.M.), 311-323. Air Waste Management Assoc., Pittsburgh, P.A.

Zimmerman J.R., Thompson R.S., (1975), User's guide for HIWAY, a highway air pollution model, EPA-650/4-74-008.

APPENDIX: Derivation Of The Crosswind Integrated Equation

Continuous, Infinite Line Source Release:

$$C_{(x,\,0,\,0;\,0)} = \frac{2q}{(2\pi)^{1/2}\, u\sigma_z}$$

q is the source strength per unit distance ($\mu g m^{-1} s^{-1}$), u is the mean wind speed (ms^{-1}) and σ_z is the vertical dispersion coefficient. The release and receptors are at groundlevel, and it is assumed that the wind direction is normal to the line source. σ_y does not appear in the equation since it is assumed that the lateral dispersion from one segment of the line is compensated by dispersion in the opposite direction from adjacent segments. Also, y does not appear since the concentration at a given x is the same for any value of y.

Continuous, Point Source Release:

$$C_{(x,\,y,\,z;\,H)} = \frac{Q}{2\pi\, u\sigma_z} \exp\left[-\frac{y^2}{2\sigma_y{}^2}\right]\left\{\exp\left[\frac{(H-z)^2}{2\sigma_z{}^2}\right] + \exp\left[-\frac{(H+z)^2}{2\sigma_y{}^2}\right]\right\}$$

Q is the release rate in mass per unit time ($\mu g s^{-1}$), and σ_y is the horizontal dispersion coefficient.

Assuming the release and receptors are at groundlevel, H and z = 0:

$$C_{(x,\,y,\,0;\,0)} = \frac{2Q}{2\pi\, u\sigma_z} \exp\left[-\frac{y^2}{2\sigma_y{}^2}\right]$$

The groundlevel crosswind integrated concentration is determined by integrating the above equation with respect to y from $-\infty$ to $+\infty$:

$$\int_{-\infty}^{+\infty} C_{(x,y,0,0)} = \int_{-\infty}^{+\infty} \frac{2Q}{2\pi\ u\sigma_z} \exp\left[-\frac{y^2}{2\sigma_y{}^2}\right]$$

Crosswind Integrated, Continuous Point Source Release:

$$C_{(CWI)} = \frac{2Q}{(2\pi)^{1/2}\ u\sigma_z}$$

Q is the source strength per unit distance ($\mu gm^{-1}s^{-1}$).

Since the crosswind integrated and line source concentration equations are the same, the use of the crosswind system to model the line source release is a viable approach.