Measurement of PM_{10} Emission Factors from Paved Roads Using On-Board Particle Sensors

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ABSTRACT

Based on emission factors derived from the AP-42 algorithm, particulate matter from paved roads has been estimated to be a major source of PM_{10} of geologic origin. This is an empirical formula based on upwind-downwind measurement of PM_{10} concentrations and is dependent solely on the silt loading of the pavement and the weight of vehicles. A number of upwind-downwind studies conducted in urban areas to validate this algorithm have been generally unsuccessful because the PM_{10} concentration difference between upwind and downwind often is within the measurement uncertainty. In the approach presented here PM_{10} concentrations were measured directly on moving vehicles in order to improve the measurement sensitivity for estimating the emission factors for vehicle on paved roads. Optical sensors were used to measure PM_{10} concentrations with a time resolution of approximately ten seconds. Sensors were mounted in the front and behind the vehicle in the well-mixed wake. A special inlet probe was designed to allow isokinetic sampling under all speed conditions. As a first approximation the emission factor was based on the concentration difference between upwind and downwind and the frontal area of the test vehicle. The emissions factors for a wide variety of roads in southern California ranged from 64 to 124 mg/km. These are consistent with but generally lower than measurements using upwind-downwind techniques. This technique is useful for quickly surveying large areas and for investigating hot spots on roadways caused by greater than normal deposition of PM_{10} forming debris.

INTRODUCTION

Many areas in the United States consistently exceed both the State and Federal PM_{10} air quality standards, and they are expected to exceed the new $PM_{2.5}$ standards. To formulate effective mitigation approaches, the sources of the PM must be accurately known. Receptor modeling has shown that PM_{10} of geologic origin is often a significant contributor to the concentrations in areas that are in non-attainment[1]. A significant portion of this geologic material has been estimated to originate from paved roads[2,3]. A number of studies have been conducted to determine the contribution of paved roads to measured concentrations of PM_{10} [2,4,5,6,7,8,9,10]. These studies used upwind-downwind sampling by filtration to determine the net mass emission due to the roadway.

The studies conducted by Cowherd and co-workers primarily in the Midwest using industrial roads resulted in an empirical expression relating the PM emission rate with the silt loading of the road. This expression was incorporated into the EPA document AP-42 for predicting emission rates and has been widely used all over the country to estimate the fraction of PM_{10} originating from roads:

Equation (1) $E = k(sL/2)^{0.65} (W/3)^{1.5}$ g/VKT

where:

E = PM emission factor in the units shown
k = A constant dependent on the aerodynamic size range of PM (1.8 for $PM_{2.5}$; 4.6 for PM_{10})
sL = Road surface silt loading of material smaller than 75μm in g/m^2
W = mean vehicle weight in tons
VKT = vehicle kilometer traveled

Equation (1) is an empirical equation derived by measuring the total flux across roadways using a PM_{10} monitoring array and based solely on surface silt loading. The AP-42 states that the sL reaches an equilibrium value without the addition of fresh material. If equilibrium is attained, then the emission rate should go to zero, although this is not what the equation predicts. Therefore, it is difficult to understand how this equation could be universally applicable unless the material is continuously replaced, a phenomenon which for most public roads is not likely.

If the silt loading were decreased by sweeping, PM_{10} emissions would be expected to decrease proportionately. The EPA has estimated that a thorough sweeping program could reduce the emissions from paved roads by approximately one-third[11]. In a study conducted in Reno, NV, however, no relationship was observed between sweeping streets and ambient PM_{10} concentrations[12]. This lack of relationship could be caused by the emissions created during the sweeping process canceling out the expected benefits. We have recently quantified the emission rates of regenerative sweepers similar to those used in the Reno study and found them to be insignificant compared with the silt removed[13]. Another explanation is that the silt loading is rapidly replaced after sweeping to an equilibrium level dependent on factors such as vehicle speed and traffic density. A third explanation is that the Reno study was not sufficiently sensitive to detect a change.

We previously conducted a study to measure and model the PM_{10} emissions from paved roads in southern California[4]. Emissions were quantified by making filter-based PM_{10} measurement upwind and downwind of several types of paved roads. In most instances, the differences in concentrations were very close or at the measured precision of the measurement method. This resulted in a large amount of error when calculating the emission factors from a modeling approach. Silt measurements were made concurrently for a number of the tests. There was no correlation between silt loading and the estimated emission factors. Silt loadings were generally lower than those suggested as defaults in AP-42. This is not unexpected since many of the roads in southern California do not have

a significant source of crustal material to create emissions. The silt loadings are likely to rapidly equilibrate at a low level due to the effective "vacuuming" from the vehicle's wake or motion of the tire. Nicholson and Branson observed this rapid attainment of equilibrium when particles tagged with a fluorescent dye were deposited on a road and monitored[14].

As an extension of this program, we performed measurements before and after sweeping the streets[13]. Even on a street that is not routinely swept, there was no significant change in either the PM_{10} emission factor or in the silt loading of the active traffic lane.

Because emissions from a fugitive source cannot be measured directly, they must be inferred. This is usually achieved by one of the following methods:

- By estimating the flux of material through a horizontal plane downwind of the source[10], or

- By fitting a dispersion model to measurements of concentrations and winds[15,16] made at locations downwind of the source; the emission rate is essentially the parameter that results from this analysis.

In principle, the calculation of horizontal flux can be an accurate method if the sampling density is sufficiently high. In practice, this density is difficult to achieve. The approach also requires measurements of low winds close to the ground where the highest concentrations occur. This is also difficult to do experimentally. To calculate emission factors it is often necessary to make assumptions about the behavior of the concentrations and wind velocities near the ground. For example, Cowherd and Englehart[10] assumed that the flux at the ground was equal to that at 1m. The validity of this assumption has not been justified. The robustness of the flux measurement depends on good coverage of several downwind locations using profilers. Most studies to date have used only one profiler.

The second method of inferring emissions involves fitting a dispersion model to a small set of concentration measurements. The accuracy of the method depends upon information on wind speed, release height, and vertical plume spread, and a physically realistic dispersion model applicable to surface releases. Some of these parameters must be estimated for emissions due to vehicles. To avoid the problem of the wind speed being zero at the surface, a release height can be chosen at which the velocity is specified. Independent measurements of emission factors are needed to estimate the uncertainty of this technique.

There were two major differences in our approach compared with previous studies of PM emissions from paved roads. First, we used real-time measurement methods based on optical scattering. While these instruments do not directly measure mass concentration and the response is dependent on the particle-size distribution, their measurements have been found to be highly correlated with those of filter collection/mass determination[17]. These instruments are generally more sensitive than mass-based methods and allow for immediate feedback to guide experimental procedures. The instrument we used was the

DustTrak model 8520 Aerosol Monitor manufactured by TSI Incorporated (Shoreview, MN). This instrument is battery operated and has a resolution of 1 $\mu g/m^3$ with a time constant of 1 second. It has interchangeable nozzles for either $PM_{2.5}$ or PM_{10} measurements.

The second major difference is that we made measurements directly behind moving vehicles and characterized the emissions under a wide variety of driving conditions. One advantage of this approach is that concentrations are expected to be much higher when nearer to the source since the PM would disperse in the process of reaching a position far enough from the roadway to safely collect a sample. Our observations of vehicles traveling on unpaved roads showed that the plume does not appreciably disperse for several car lengths. In previous studies, others and we have estimated the lower limit emission factor of 0.1g VKT (vehicle kilometer traveled) on high-speed, high-traffic-count paved roads. Using this emission factor, the plume from the wake would have a concentration of 25 $\mu g/m^3$. Given this low plume concentration, ambient background, and subsequent dispersion, it is understandable why downwind PM measurements are typically only several $\mu g/m^3$ higher than upwind.

The second advantage to real time sampling is that dispersion modeling was not needed since the monitoring was done before any significant dispersion occurred. We characterized the PM distribution within the wake of the vehicle and used these data to determine the emission rate in g/VKT by dividing the PM concentration by the wake area. Combining the real-time measurements on a moving vehicle also gave the advantage of being able to rapidly collect data over a wide variety of vehicle operating parameters and road types.

BODY

Experimental Methods

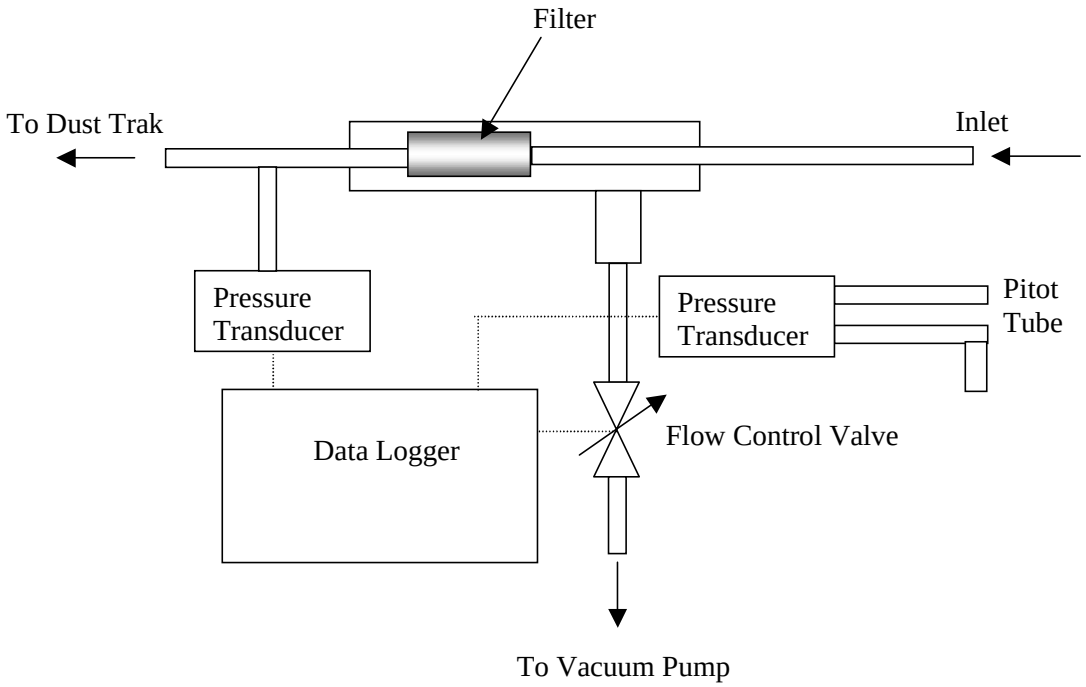
This task was broken into three phases. The first involved the design and construction of an isokinetic sampling probe, the second was characterization of the PM distribution in a vehicle's wake, and the third was the measurement of PM emission rates on a variety of roads and driving conditions.

Isokinetic Sampling Inlet

Collecting particulate samples from a vehicle moving at speeds of 0 mph to 60 mph required designing an inlet that would provide, as much as possible, isokinetic sampling at all speeds. Figure 1 shows the design of the inlet. A vacuum pump is used to maintain the bulk air speed at the inlet equal to the speed of the air going past the inlet. To slow the flow to the sample flow rate of the DustTrak without creating a virtual impactor, excess air is pulled through a hollow, cylindrical filter. The flow to the vacuum pump is adjusted at speed to produce a reading of zero pressure on the gauge. When the pressure equals zero, there is no pressure drop from the probe inlet to the tubing that leads to the DustTrak. This condition creates a no-pressure-drop inlet; therefore, the sampled

airstream has the same energy as the ambient airstream. The output of the pressure transducers were recorded on a Campbell CR10X data logger.

Figure 1. Schematic diagram of the isokinetic sampling probe.



Characterization of the Vehicle Wake and Sampling Point Optimization

To determine where in the vehicle wake to collect samples, the PM concentrations in the vehicle wake must be characterized. To do this, it was necessary to measure PM concentrations at many locations behind a moving vehicle under controlled conditions. A rectangular frame (2m wide and 2m high) was constructed on a small trailer to hold sampling inlets at any position within the frame. A small trailer was used to hold sampling inlets at various locations behind the vehicle. The DustTraks, inlet pumps, and data logger were mounted in plastic boxes strapped to the rear of the trailer.

Metal impregnated ¼ inch OD plastic tubing (Bev-A-Line XX tubing, Thermoplastic Processes, Inc., Warren, NJ) was used to transport sample from the isokinetic sampling probe to the DustTrak. The metal impregnation of the tubing reduced static charges on the tubing for greater particle penetration. In addition, the shortest possible lengths of tubing were used. The trailer, frame, and associate components were designed to minimize the aerodynamic influence of the trailer with respect to the vehicle's wake. The trailer was equipped with a 6 m adjustable tongue to vary the distance of the sampling array from 1.4 to 5.9 m behind the tow vehicle. Figure 2 shows a photograph of the trailer; the tow vehicle is out of the picture to the left. The DustTraks were attached to the bed of the trailer (left foreground). Three of the plastic boxes contain DC vacuum pumps

for the inlet; the fourth one housed the data logger. All the isokinetic sampling probes are located on the left side of the sampling bar.

Figure 2. Trailer used to mount the isokinetic sampling probes and the DustTrak PM sensors.

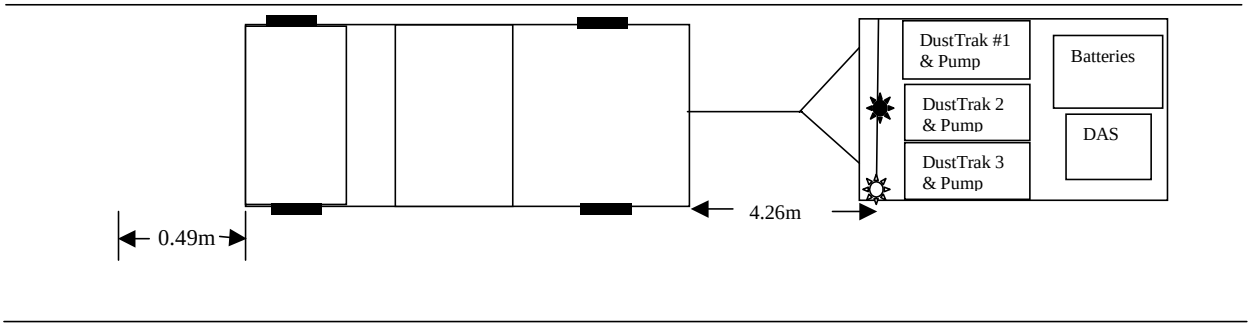
Wake characterization experiments were performed on Seminole Drive in Cabazon, CA, an infrequently traveled 1.5-mile-long road that runs parallel to Interstate 10. We were able to sample at several speeds between 20 and 60 mph. In addition, the surface was quite weathered and contained sufficient loading of fine debris to obtain adequate DustTrak responses.

As shown in Figure 3, PM_{10} measurements were made at several positions 4.3m behind the rear bumper using three DustTraks to determine the wake PM concentration characteristics. The reference sampling position was on the vehicle centerline 0.78m above the ground. The other two positions were located either 1.98 or 2.59m from the ground, with one on the centerline and the other 1.22m from the centerline. Due to safety concerns we did not extend probes farther than 1.23m from the centerline.

Based on the initial screening on unpaved roads, the reference probe should be located in a high PM_{10} concentration area. The elevation 1.98m above the ground is near the vehicle's roofline height. As observed on the unpaved roads, this is the approximate height of the visual plume. The 1.22m from the centerline places the probe slightly (0.1m) past the maximum width of the vehicle and therefore near the dust plume's visually observed lateral boundary. These test positions should therefore outline the PM

wake of the vehicle. Table 1 summarizes the probe positions and speeds used for the various tests that were conducted. All data was collected as 2-second averages.

Figure 3. Schematic diagram of the sampling configuration for vehicle wake PM characterization.



DustTrak #1 Inlet (1.98 or 2.59 m above ground)

DustTrak #2 & #3 Inlets (0.78 & 1.98 or 2.59 m above ground)

Table 1. Vehicle wake characterization sample test matrix inlet locations.

Speed (mph)	**Sample location on trailer* (meters)**			
		X	**Y**	**Z**
20, 30, 40, 50	DT#1	4.25	1.98	1.23
	DT#2	4.25	0.78	0.0
	DT#3	4.25	1.98	0.0
20, 25, 30	DT#1	4.25	1.98	1.23
	DT #2	4.25	0.78	0.0
	DT#3	4.25	1.98	0.0
30, 40, 50, 60	DT#1	4.25	1.98	1.23
	DT#2	4.25	0.78	0.0
	DT#1	4.25	1.98	0.0
20, 30, 40, 50, 60	DT#1	4.25	1.98	1.23
	DT#2	4.25	0.78	0.0
	DT#3	4.25	1.98	0.0
20, 30, 40, 50, 60	DT#1	4.25	2.59	1.23
	DT#2	4.25	0.78	0.0
	DT#3	4.25	2.59	0.0

*X distance from rear of vehicle, Y distance from ground, Z distance from centerline

Field Measurements

Field measurements were conducted using a Chevrolet Suburban towing the test trailer. Two sampling probes were place on the trailer:

- Position 1: 0.76m from the ground and 4.25m from the rear of the vehicle.
- Position 2: 2.59m from the ground and 4.25 m from the rear of the vehicle.

Both sampling ports were on the centerline of the vehicle. The front sampling port (reference) was centered 0.43m in front of the hood of the vehicle and 1.07m from the ground. A Campbell CR10X data logger was used to collect all data at intervals of two seconds.

After several rounds of initial testing on various roads and speeds in the Riverside, CA, area, we settled on a test routes that contained segments of arterial, collector, and local roads. Separate tests were conducted on Interstate 215 to gather data on high-speed, limited-access freeways. Notes were taken coinciding with the data logger time to describe the segment tested, the speed, and any unusual circumstances. On each road and for each PM nozzle (10μm and 2.5μm) the tests were repeated twice.

At the end of each day the three DustTraks were collocated using one inlet or the other. To help minimize measurement noise and other factors the collocated data from the entire sample period were compiled and used to normalized DustTrak measurements. The method for performing the normalization is fully described under the Results and Discussion sections.

Results and Discussion

Data Quality

Before describing the experimental results we include this section to describe the overall quality of data that the DustTraks provide and some of the corrective action that we undertook to improve the data quality.

- **DustTrak Averaging Time**

The DustTrak updates at one hertz. We originally compared collocated sampling using data collected and stored at two-second intervals. Figure 4 is typical example of the two-second data collected from two collocated DustTraks. The correlation coefficients were always much less than 0.5. With these data we could not ascertain whether the DustTraks were even measuring the same phenomenon, let alone whether they were equivalent. When the data are compared as 60-second running averages as shown in Figure 5, the correlation coefficients are quite acceptable. It was discovered that approximately 30-second averaging times were necessary for the DustTraks to provide equivalent data. Therefore, it was necessary to maintain a speed for approximately 30 seconds to be able to calculate a net difference between the DustTrak's responses.

Figure 4. Comparison of two collocated DustTraks using 2-second PM_{10} data.

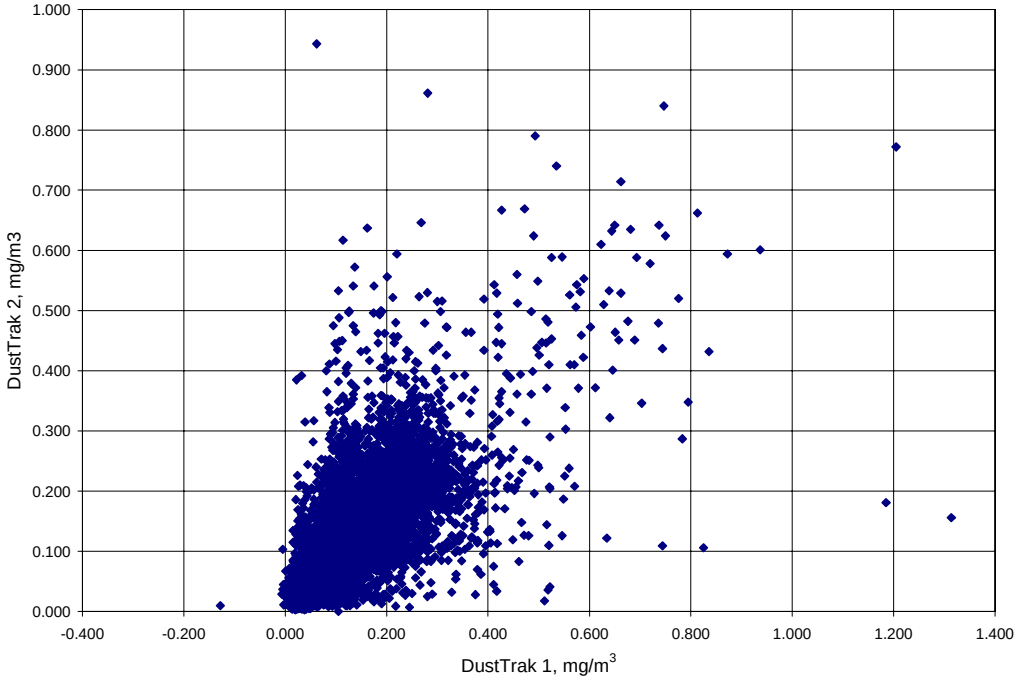
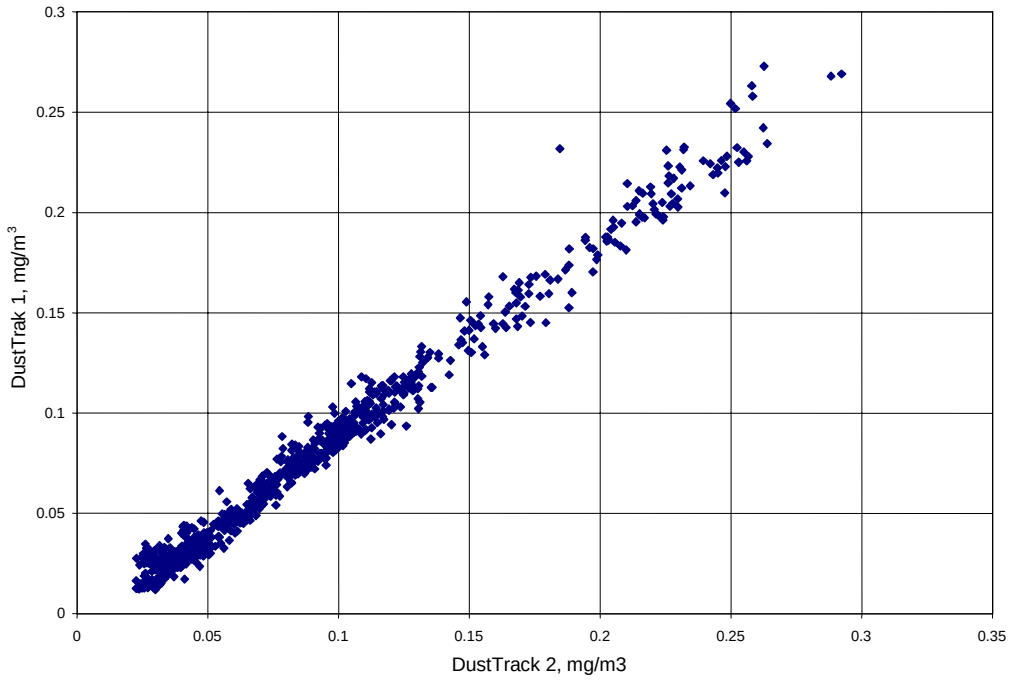


Figure 5. Comparison of two collocated DustTraks using 60-second PM_{10} data.



- **Uncertainty**

To calculate measurement uncertainty we calculated the relative standard deviation (RSD) of the difference between the PM_{10} measured by the collocated samplers. We calculated the mean and standard deviation of the percent absolute difference between the three samplers in all combinations 2-1, 3-1, and 3-2. The standard deviation was divided by the mean, and the average RSD was calculated. To reduce measurement noise, two-second data was used to calculate six second running means was followed by one-minute averages. The calculated RSD for the vehicle wake characterization component was 0.79%. For the other on-board vehicle PM measurements the calculated RSD was 7.1%...

- **Evaluation of PM Losses in the Sampling Line**

It was necessary to use sampling lines ranging in length from 1 to 3m to change positions during the on-board wake characterization. To evaluate losses of particles within the sampling lines, three DustTraks equipped with PM_{10} inlets were collocated and sampled ambient air for two hours, alternating 10 minutes without tubing and 10 minutes with tubing (1.70m). Data were collected as 30-second averages. The data are summarized in Table 2. Based on the means, the tubing caused a loss of PM ranging from 21 to 29 percent depending on which DustTrak was evaluated.

Table 2. Mean DustTrak PM_{10} response on ambient air with and without Bev-A-Line tubing.

Mean DustTrak 1 (mg/m^3) w/o tubing	Mean DustTrak 1 (mg/m^3) w/ tubing	Difference DT 1 (w/o - w)	% Difference DT 1 (w/o - w)	SD of Difference
0.269	0.213	0.056	20.8%	0.117
Mean DustTrak 2 (mg/m^3) w/o tubing	Mean DustTrak 2 (mg/m^3) w/ tubing	Difference DT 2 (w/o - w)	% Difference DT 2 (w/o - w)	SD of Difference
0.359	0.272	0.087	24.2%	0.132
Mean DustTrak 3 (mg/m^3) w/o tubing	Mean DustTrak 3 (mg/m^3) w/ tubing	Difference DT 3 (w/o - w)	% Difference DT 3 (w/o - w)	SD of Difference
0.253	0.179	0.074	29.3%	0.110

In addition, the data sets for each DustTrak were compared with and without the tubing using the Wilcoxan[18] non-parametric ranking test. For all three DustTraks the data sets were shown to not be equivalent. The loss of PM due to the tubing, therefore, is significant, but could be corrected.

Results

Wake Characterization

With the sampling matrix presented in Table 1 we determined:

- The precision of the measurement (with all three DustTraks sampling from the same point).
- The homogeneity of the PM within the vehicle's wake with respect to the vehicle's speed.
- The vertical and horizontal extent of the plume as a function of vehicle speed and cross wind.
- The optimum sampling position.

The three DustTraks were collocated on several days during the characterization of the vehicle wake. Identical lengths of the anti-static tubing was used from each sampling point to the DustTraks' inlet. Collocated data from DustTrak #2 and #3 were plotted against DustTrak #1, and least-squares regression analyses were performed. To improve the comparability, data from DustTraks #2 and #3 were normalized to #1 using the following regression equations:

DustTrak #2 normalized = 0.1.13(DustTrak #2 raw) –0.005 (R^2 = 0.926)

DustTrak #3 normalized = 0.93 (DustTrak #3 raw) + 0.13 (R^2 = 0.920)

For actual testing the length of the tubing varied with the location of the inlet on the matrix (1.70, 2.29, or 2.74m). The tubing was manually interfaced to the DustTrak to obtain sequential samples at each test point. Losses of PM were not corrected for.

The data are summarized in Figures 6 and 7. The PM_{10} concentration of the plume clearly drops as the sampling point is raised from the top of the tow vehicle to 2 feet above it. The concentration also drops rapidly when sampling just beyond the edge of the 2m wide tow vehicle (1.2m from the centerline). These data confirm that the plume is confined primarily within the frontal area of the tow vehicle and that a sampling position in the geometric center of the tow vehicle frontal area is an appropriate sampling point. Therefore, we chose two sampling locations in the rear of the vehicle: 1) a reference probe centered 0.78m from the ground and 2) a probe centered 2.59m from the ground.

Figure 6. PM_{10} concentration while towing the test trailer at various speeds with two of the isokinetic sampling probes located 78 inches (1.98 m) above the ground.

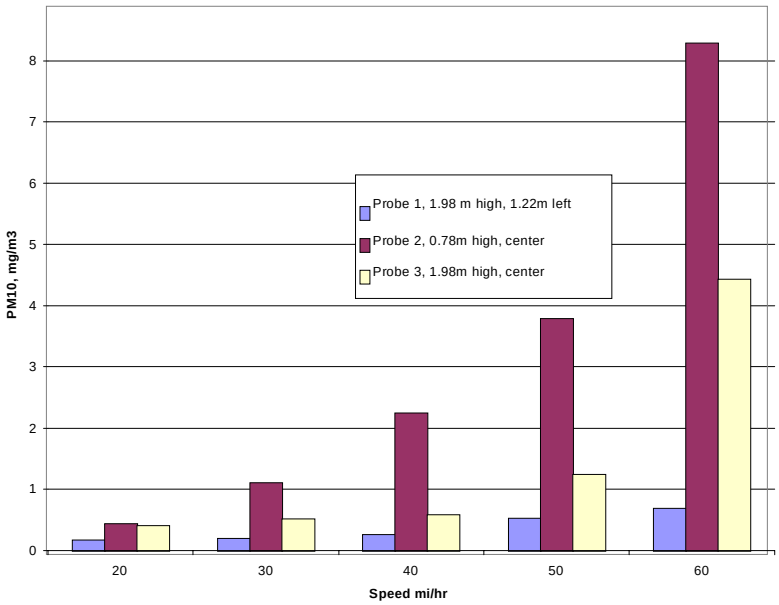
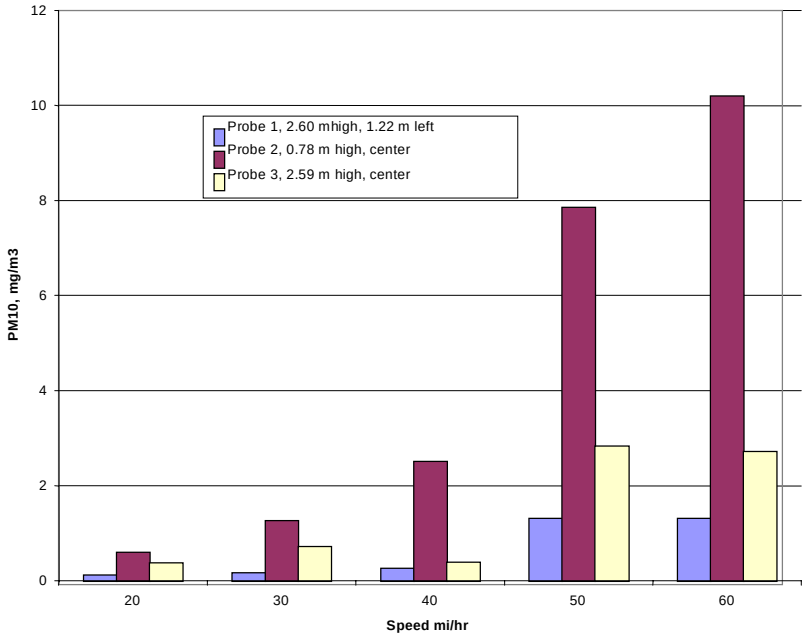


Figure 7. PM_{10} concentrations while towing the test trailer at various speeds with two of the isokinetic sampling probes located 102 inches (2.59 m) above the ground.



On-Vehicle Real-Time Emission Measurements

Segments of roadways were driven at several different speeds as safety and road conditions permitted. For surface streets, segment was defined as the total time and distance traveled between stops (stop sign, stoplight, or sharp turn). For freeways the segments were the distance from entering to exiting the freeway. The roadways driven were defined as the following:

- Freeways: Four or more lanes with a car count of at least 150,000 cars per day.
- Arterial: Three, four, or more lanes with a car count of 10,000 - 150,000 cars per day.
- Collector: Two lanes with a car count of 500 - 10,000 cars per day.
- Local: Two lanes with a car count of 500 or fewer cars per day.

Table 3 shows the number of segments of various types for which we report data. Thirty-second averages were taken within each segment. DustTrak #1 is the reference, mounted in front of the vehicle, DustTrak #2 is on the trailer at the centerline 0.78m from the ground, and DustTrak #3 is in a similar location except 2.59m above the ground.

The collocated data from the entire sample period were compiled and the data were normalized to DustTrak #1 to improve the comparability. This was done by regressing DustTraks #2 and #3 against DustTrak #1 using the following equations:

$$\text{DustTrak \#2 normalized} = 0.86(\text{DustTrak \#2 raw}) - 0.001 \quad (R^2 = 0.967)$$

$$\text{DustTrak \#3 normalized} = 0.94\ (\text{DustTrak \#3 raw}) + 0.013 \quad (R^2 = 0.986)$$

Table 3. Number of segments for which PM emission measurements were made.

Roadway Type	PM sampled	20 MPH	25 MPH	30 MPH	35 MPH	40 MPH	45 MPH	50 MPH	55 MPH	57 MPH	60 MPH
Freeway Roadway	10					1		1	12	2	
Freeway Roadway	2.5				1				7	1	
Arterial Roadway	10					4	44	3	4		
Arterial Roadway	2.5					2	27		5		
Collector Roadway	10						22				
Collector Roadway	2.5						17				
Local Roadway	10				15						
Local Roadway	2.5				15						
Decomposed Roadway	10	4		6		6		6			6
Decomposed Roadway	2.5										
Unpaved Roadway	10	3	1	4		2		2			
Unpaved Roadway	2.5										

Table 4 has segment averages and emission factors for each roadway type driven for PM_{10} and $PM_{2.5}$. The table also presents the difference of DustTrak 2 from DustTrak 1, and DustTrak 3 from DustTrak1. An emission factor was calculated for each segment and is also shown in this table.

To calculate emission factors we assumed that a wake of the dimensions of the vehicle's frontal area was swept out and that the PM emissions remained within this volume until the point of measurement. We further assumed that the mean of the two PM measurements behind the vehicle (one located at 0.76m and the other at 2.59m above the road surface was representative of the concentration within the plume created by the wake). The emission rate was calculated by multiplying the frontal area by the concentration. This results in units of mg/m, which was then converted to mg/km by multiplying by 1000.

Table 4. Average PM emission rates for various road types.

DustTrak #1[a] (mg/m^3)	DustTrak #2[b] (mg/m^3)	DustTrak #3[b] (mg/m^3)	Difference DustTrak #2-#1 (mg/m^3)	Difference DustTrak #3-#1 (mg/m^3)	Road type	DustTrak #2-#1 emission factor[d] (mg/km)	DustTrak #3-#1 emission factor[d] (mg/km)	PM
0.019	0.040	0.040	0.021	0.021	Local	68.7 +/- 4.9	68.0 +/- 4.8	10
0.044	0.057	0.053	0.013	0.009	Collector	43.2 +/- 3.1	30.7 +/- 2.2	10
0.059	0.088	0.073	0.030	0.015	Arterial	98.4 +/- 7.0	48.6 +/- 3.5	10
0.056	0.089	0.061	0.033	0.005	Freeway	79.3 +/- 5.6	14.9 +/- 1.1	10
0.012	0.031	0.032	0.019	0.020	Local	61.1 +/- 4.3	64.9 +/- 4.6	2.5
0.074	0.084	0.079	0.010	0.005	Collector	31.7 +/- 2.3	15.4 +/- 1.1	2.5
0.048	0.058	0.058	0.013	0.011	Arterial	41.5+/- 3.0	35.7 +/- 2.5	2.5
0.026	0.035	0.038	0.009	0.013	Freeway	29.4 +/- 2.1	41.3 +/- 2.9	2.5

a) DustTrak #1 in front 1.07 m from ground, 0.42 m from vehicle, centered
b) DustTrak #2 on trailer 0.76 m from ground, 4.26 m from vehicle, centered
c) DustTrak #3 on trailer 1.35 m from ground, 4.26 m from vehicle, centered
d) Emission factor = (PM concentration difference, mg/m^3)*(frontal area, $3.30m^2$)*$1000mkm^{-1}$

The data in Table 4 show that there are measurable PM_{10} emissions in the wake of the test vehicle. Local roadways, traversed at moderate speeds of 35 mph, had a mean difference of 0.21 mg/m^3 between the front and the two levels sampled in the rear of the test vehicle. Both rear elevations showed similar concentration differences. We calculated an average emission factor of 68 mg/km for these roads. Based on our collocated DustTrak and filter-based measurements, multiplying this value by 1.74 would convert the DustTrak response to mass filter measurements and yield 118 mg/km.

Collector roads were traveled at an average speed of 45 mph. The average concentration difference was 0.013 mg/m^3 at the 0.76 m elevation and 0.009 mg/m^3 at the 1.35 m sampling position. At the greater speed the plume may have higher concentration near the ground. The average PM_{10} emission rate was 64 mg/km on a corrected mass basis (i.e., DustTrak values multiplied by 1.74 to obtain corrected mass). Both the arterial and freeway was traveled at an average speed of 50 to 55 mph. The average concentration difference for the arterial roadway at the 0.76 m position was much higher, 0.030 mg/m^3, than the higher position or the local or collector roads. The average emission rate on the corrected mass basis was 129 mg/km. The emission rates from the freeway driving was between arterial and collector roads but the much lower emission from the measurements at 1.35 m indicate that even more of the plume remains lower to the ground. The average corrected mass basis emission rate was 82 mg/kg.

Table 5 compares the emission factors from this study with others expressing the results in grams per vehicle kilometer travels (VKT) and pounds per vehicle mile traveled (VMT). The data indicate that the low end of the emission factors are similar to the other studies, but we did not find the spread observed by these other studies. The values are also lower than the default values calculated by AP-42 using silt loadings. Unlike the other studies that measured emission rates from concentration differences, it is possible that the roads we evaluated had low amounts of PM-generating material being deposited on them. The other studies also did not report negative concentration differences (i.e., downwind concentrations lower than upwind) since such results cannot be modeled. Their values would then be skewed high, particularly when making measurements near the detection limit.

Table 5. Comparison of PM_{10} emission factors reported by other sources.

Study	Road Type	Emission Factor (g/VKT)	Emission Factor (lbs/VMT)
This Study	Freeway-local	0.06 – 0.13	0.00022-0.00047
Venkatram and Fitz, 1998 [4]	Freeway-local	0.1-0.3	0.00036-0.0011
Cahill et al., 1995 [19]	Intersection	<0.3	<0.001
Claiborn et al., 1995 [8]	Freeway-local	0.5 to 34	0.0018-0.12
Harding Lawson, 1996 [6]	Freeway-local	0.03 to 180	0.00011-0.65
AP-42 Default[a]	Arterial-local	0.08-0.53	0.00030-0.0019
ARB Default	Arterial-local	0.10-0.61	0.00036-0.0022

a: From silt loadings measured in southern California, assuming 2 ton vehicles

The concentration differences we observed were generally higher, as expected than what we saw for upwind-downwind measurements. Since the concentration differences are

higher and a much greater amount of roadway was sampled compared with upwind-downwind or silt measurements conducted at a single or several sites, we feel that this technique is able to measure PM emissions from vehicles with greater accuracy than previous determinations. This conclusion recognizes that we are assuming that the frontal area of the vehicle represents the volume of the plume behind the vehicle and that the concentrations within the plume are uniform. While these are significant assumptions, the assumptions and uncertainties of the other methods are likely to be as great or greater.

The data for $PM_{2.5}$ are in general agreement with those from PM_{10} sampling but lower in magnitude. Only a limited number of filter samples were collected for $PM_{2.5}$, and, therefore, we do not have a regression equation to adjust the values to a mass basis. All types of roadways developed higher concentrations at the 0.76 m sampling position than at the 1.35 m sampling position for PM_{10}. Collector and arterial roadways produced higher concentrations at the 0.76 m sampling position and local and freeway roadways.

CONCLUSIONS

Real-time measurements in front of and behind a vehicle were found to be a useful method to characterize PM_{10} emission rates from paved roads. The emission rates ranged from 64 to 124 mg/km and are in general agreement, but on the low side, of those previously reported. By contrast, the current ARB emission factors range from 130 to 830 mg/kg using the AP-42 methodology with California-specific silt loading values. The emission rates did not vary a great amount from local roads with a few hundred cars per day to freeways with over 40,000 cars per day per lane of traffic. Unlike the upwind-downwind approach, a significant concentration differential was measured. The emission rate could also be calculated without applying modeling techniques that are also likely to have high uncertainty. Therefore, we feel that this approach is more accurate than the upwind-downwind approaches used in the past and also allows the testing of much longer sections of roadway with relatively little effort.

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KEY WORDS

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