4-12-2020

Real-Time PM10 Emission Rates From Paved Roads by Measurement of Concentrations in the Vehicle's Wake Using On- Board Sensors Part 1. SCAMPER Method Characterization

Dennis R. Fitz

University of California, Riverside, dfitz@cert.ucr.edu

Kurt Bumiller

University of California, Riverside

Charles Bufalino

University of California, Riverside

David E. James

University of Nevada, Las Vegas, dave.james@unlv.edu

Follow this and additional works at: https://digitalscholarship.unlv.edu/fac_articles

Part of the [Environmental Engineering Commons](#), and the [Transportation Engineering Commons](#)

Repository Citation

Fitz, D. R., Bumiller, K., Bufalino, C., James, D. E. (2020). Real-Time PM10 Emission Rates From Paved Roads by Measurement of Concentrations in the Vehicle's Wake Using On-Board Sensors Part 1. SCAMPER Method Characterization. *Atmospheric Environment*, 230 1-10.
<http://dx.doi.org/10.1016/j.atmosenv.2020.117483>

This Article is protected by copyright and/or related rights. It has been brought to you by Digital Scholarship@UNLV with permission from the rights-holder(s). You are free to use this Article in any way that is permitted by the copyright and related rights legislation that applies to your use. For other uses you need to obtain permission from the rights-holder(s) directly, unless additional rights are indicated by a Creative Commons license in the record and/or on the work itself.

This Article has been accepted for inclusion in Civil & Environmental Engineering and Construction Faculty Publications by an authorized administrator of Digital Scholarship@UNLV. For more information, please contact digitalscholarship@unlv.edu.

Real-Time PM₁₀ Emission Rates from Paved Roads by Measurement of Concentrations in the Vehicle's Wake Using On-Board Sensors

Part 1. Method Characterization

Dennis R. Fitz*, Kurt Bumiller, and Charles. Bufalino
College of Engineering-Center for Environmental Research and Technology
University of California, Riverside
1084 Columbia Avenue, Riverside CA 92507

dfitz@cert.ucr.edu

and

David E. James Department of Civil and Environmental Engineering
University of Nevada, Las Vegas

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 PM₁₀ of geologic origin. This is an empirical formula based on upwind-downwind measurement of PM₁₀ 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 inconclusive because the PM₁₀ concentration difference between upwind and downwind often is within the measurement uncertainty. In the approach presented here PM₁₀ concentrations were measured directly behind a moving vehicle in order to improve the measurement sensitivity for estimating the emission rates for vehicles on paved roads. Optical sensors were used to measure PM₁₀ concentrations with a time resolution of approximately ten seconds. Sensors were mounted in the front of the vehicle and behind it in the well-mixed wake. A special inlet probe was designed to allow isokinetic sampling under varying speed conditions. As a first approximation the emission rate was calculated by multiplying the PM₁₀ concentration difference between front and rear of a moving vehicle by the frontal area of the vehicle. The results correlated well when compared to rates calculated using the measured silt loading. This technique is also useful for quickly surveying large areas and for investigating hot spots on roadways caused by greater than normal deposition of PM₁₀-forming debris. The method is designated as SCAMPER: System for the Continuous Aerosol Measurement of Particulate Emissions from Roads.

Keywords: Particulate, Paved road, Road dust, Emission factors, Fugitive dust, PM₁₀

Highlights:

- Characterization of PM₁₀ concentrations in a vehicle's wake
- PM₁₀ measurements in a vehicle's wake on paved roadways
- Comparison of emission PM₁₀ rates with AP-42 equation using measured silt loadings
- PM₁₀ Emission rates from a variety of roads in Las Vegas, NV

I. Introduction

Many areas in the United States consistently exceed both the State and Federal PM₁₀ air quality standards. To formulate effective mitigation approaches, the sources of the PM must be accurately known. Receptor modeling has shown that PM₁₀ 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 PM₁₀.^{2,4-12} These studies used upwind-downwind PM measurements to determine the net mass emission due to the roadway.

Studies conducted by Cowherd and Englehart¹⁰, primarily in the Midwest using industrial roads, resulted in an empirical expression relating the PM emission rate with the silt loading of the road and the mass of the vehicles. 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 PM₁₀ originating from roads.⁹

$$E = k(sL)^{0.912} (W)^{1.021} \text{ g/VKT} \quad (1)$$

where:

E = PM emission rate in the units shown

k = A constant dependent on the aerodynamic size range of PM (0.62) for PM₁₀)

sL = Road surface silt loading of material smaller than 75µm in g/m²

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 and vehicle weight.

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 to 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 significant correlation between silt loading and the estimated emission rates. 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 tires. Nicholson and Branson¹³ observed this rapid attainment of equilibrium when particles tagged with a fluorescent dye were deposited on a road and monitored.

Because emissions from a fugitive source cannot, by their nature, be measured directly as in a duct, they must be calculated from the plume characteristics. This is usually achieved by one of the following methods using measurements made upwind and downwind of the source:

- By estimating the flux of material through a horizontal plane downwind of the source, or
- By fitting a dispersion model to measurements of downwind concentrations and meteorological parameters.

In principle, the calculation of horizontal flux can be an accurate method if the sampling density is sufficiently high, but this is very resource-intensive. The approach also requires measurements of low winds close to the ground where the highest concentrations occur. This is also difficult to perform 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 Englehart¹⁰ assumed that the flux at the ground was equal to that at 1 m.

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 rates are needed to estimate the uncertainty of this technique.

The objective of this work was to develop a new method that could measure PM emissions from paved roads in real time by equipping a vehicle with real-time particulate matter analyzers. The approach has been designated as SCAMPER: System for the Continuous Aerosol Measurement of Particulate Emissions from Roads. While the general approach is similar to the TRAKER reported by Etyemezian *et al.*¹⁴, there is a basic difference in sampling position that allows the emission rate to be estimated from the plume dimensions immediately behind the vehicle without performing the upwind-downwind measurements that are necessary to calibrate the TRAKER system that samples in the vehicle's wheel well. A method based on the SCAMPER has been reported by Mathissen *et al.*¹⁵. They characterized the plume using a gaseous tracer and a diffusor to correct for non-isokinetic sampling.

In our approach we made measurements in the wake region directly behind a moving vehicle. An advantage of this approach is that concentrations are expected to be higher directly behind a vehicle compared to a position far enough from the roadway to safely collect samples. Our observations of visible dust plumes from vehicles traveling on unpaved roads showed that a plume does not disperse appreciably for several car lengths. In previous studies, we and others have estimated the lower limit PM₁₀ emission rate of 0.1 g /VKT (vehicle kilometer traveled) on high-speed, high-traffic-count paved roads. Using this emission rate, the plume from the wake directly behind a typical vehicle would have a concentration of approximately 25 µg/m³. Given this low plume concentration,

131 ambient background, and subsequent dispersion, it is understandable why downwind PM
132 measurements are typically only several $\mu\text{g}/\text{m}^3$ higher than upwind.

133 An additional advantage to sampling directly behind the vehicle is that dispersion
134 modeling was not needed since the monitoring was done before any significant dispersion
135 has occurred. We characterized the PM distribution within the wake of the vehicle and
136 used these data to determine the emission rate in g/VKT by multiplying the net PM
137 concentration by the wake area.

138 Our method is also based on a real-time measurement method with an analyzer
139 measuring optical scattering. While these instruments do not directly measure mass
140 concentration and the response is dependent on the particle size distribution, their
141 measurements have been found to be highly correlated with those of filter collection/mass
142 determination.¹⁵ These instruments are generally more sensitive than mass-based methods
143 and allow for collection of a large amount of real-time data and provide immediate
144 feedback to guide experimental procedures. The use of a real-time analyzer greatly
145 simplified the determination of the PM distribution within the wake behind the vehicle.
146 Combining the real-time measurements on a moving vehicle also gave the advantage of
147 being able to rapidly collect data over a wide variety of vehicle operating parameters and
148 road types.

149 The instrument we used was the DustTrak model 8520 Aerosol Monitor
150 manufactured by TSI Incorporated (Shoreview, MN). This instrument is battery-operated
151 and has a resolution of $1 \mu\text{g}/\text{m}^3$ with a time constant of 1 second. It has interchangeable
152 nozzles for either $\text{PM}_{2.5}$ or PM_{10} measurements. DustTraks have been used in a number of
153 published measurements of fugitive PM emissions.^{11,14,16,17}

155 2. Materials and methods

156 The SCAMPER development was divided into three phases. The first involved the design
157 and construction of an isokinetic sampling probe, the second was characterization of the
158 PM distribution in a vehicle's wake, and the third was the measurement of PM emission
159 rates on a variety of roads and driving conditions for comparison with results calculated
160 from AP-42 default emission rates based on measured silt loadings.

2.1 Isokinetic Sampling Inlet

Collecting particulate samples from a moving vehicle required designing an inlet that would provide, as much as possible, isokinetic sampling at all speeds. Figure 1 shows the design of the inlet, and Figure 2 is a detailed drawing of the flow-splitting section. Metal tubing is used to connect the sample inlet to one end of a hollow cylindrical filter and from the other end to the DustTrak (the sampled air is not filtered, but travels from one end of the hollow cylinder to the other). To slow the flow to the sample flow rate of the DustTrak without creating a virtual impactor, excess air is pulled through the outside of the cylindrical filter with a vacuum pump that maintains the bulk air speed at the inlet equal to the speed of the air going past the inlet. The flow rate of the vacuum pump is adjusted by the data logger 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 velocity as the ambient airstream. The outputs of the pressure transducers were recorded on a Campbell Scientific (Logan, UT) CR10X data logger.

Figure 1. Schematic diagram of the isokinetic sampling probe.

Figure 2. Detail of the flow splitting section of the isokinetic sampling probe.

2.2 Characterization of the vehicle wake and sampling point optimization

To determine a representative sampling position in the vehicle wake, the PM concentrations in the vehicle wake were measured at a variety of 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 various positions within the frame. The DustTraks, inlet pumps, and data logger were mounted in plastic boxes strapped to the trailer.

Metal-impregnated ¼ inch OD plastic tubing (Bev-A-Line XX tubing, Thermoplastic Processes, Inc., Warren, NJ) was used to transport air samples 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 associated 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, a 1995 Chevrolet Suburban.

Wake characterization experiments were performed on Seminole Drive in Cabazon, CA, a seldom-traveled 2.4 kilometer long road that ran parallel to Interstate 10. Run speeds were varied from 32 and 96 km/hr in increments of 16 km/hr. The surface of the road was well-weathered asphalt pavement and contained sufficient loading of fine debris to obtain adequate DustTrak responses.

Figure 3 shows five representative sampling positions 4.3 m behind the rear bumper using three DustTraks to determine the wake PM concentration characteristics. The reference sampling position was on the vehicle centerline 0.78 m above the ground and was used for all testing. The inlet for second DustTrak sampling on the centerline was located either 1.98 or 2.59 m above the ground. A third DustTrak sampled 1.22 m left of the centerline at heights of either 1.98 or 2.59 m above the ground. Due to safety concerns we did not extend probes farther than 1.22 m from the centerline.

- ☼ 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)

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

Based on visual screening of plumes on unpaved roads, the reference probe should be located in a high PM_{10} concentration area. The elevation 1.98 m 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 position 1.22 m from the centerline places the probe slightly (0.1 m) 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 visual PM wake of the vehicle.

2.3 Measurements of roadway PM_{10} emission rates

Field measurements were conducted by repeatedly sampling over three test routes in North Las Vegas and Las Vegas, NV using the tow vehicle described in the previous section. Silt sampling was conducted at seven locations during the test runs.

The first one, Loop A, was a short loop (4.5 km long, requiring approximately five minutes to drive) of a roadway heavily impacted with trackout and soil hauling trucks. This loop, shown in Figure 4, was expected to have values well above the detection limit most of the time in order that sampling precision could be determined. The roads were lightly-traveled which made it possible to maintain consistent speeds for each test. Figure 4 also shows the approximate speeds as a function of location for a typical test run. This loop was expected to be a worst-case scenario for precision as the deposits and traffic on the road were highly variable, and significant dust was generated by vehicles in front. Silt sampling was conducted on Gowan Road, a collector road (500-5,000 vehicles/day), as shown in Figure 4.

Figure 4. Loop A 4.5 km Test Route

The second test loop, Loop B, shown in Figure 5, was also chosen to have significant impacts from construction activities. The loop was 11.7 km long and required 20-25 minutes to traverse. The loop contained arterial, collector, and local classes of roadways. Figure 5 also shows locations where silt sampling and traffic counts were conducted. Silt sampling sites included local (<500 vehicles/day), collector (500-5,000 vehicles/day) and arterial (5,000-10,000 vehicles/day) roads.

Figure 5. Loop B Test Route

The third test loop, Loop C, shown in Figure 6, was chosen to be typical of fully-developed areas in which trackout is light or non-noticeable. This loop was 20.2 km long and required 25-30 minutes to complete. The loop contained arterial, collector, and local classes of roadways. Silt sampling and traffic counts were conducted at the locations shown in the Figure. Silt sampling was conducted on two collector roads and one local road.

Figure 6 Loop C Test Route

All sampling was conducted at speeds consistent with the flow of traffic and with safety considerations for towing the sampling trailer. Local agencies were responsible to ensure that no street sweeping was performed during or at least three days before the days that tests were conducted.

The front DustTrak sampling position (reference) was on the centerline 0.43 m in front of the hood of the tow vehicle and 1.07 m from the ground. The rear position was on a trailer 0.76 m from the ground and 4.25 m from the rear of the Chevrolet Suburban test vehicle and located on its centerline. A Campbell CR10X data logger was used to collect all data at intervals of two seconds and operate the isokinetic sampling probes used with both DustTraks. It also collected speed and location data from a Garmin (Olathe KS) GPS Map76™ global positioning system.

320

321 3. Results and discussion

322 3.1 Data quality

323 This section describes the overall quality of data that the DustTraks provide and some of
324 the corrective actions that were undertaken to improve the data quality.

325 *3.1.1 DustTrak Averaging Time.* The DustTrak updates at one hertz. Data from a pair of
326 collocated DustTraks were collected at two-second intervals. The correlation coefficients
327 squared between the two analyzers were always much less than 0.5. When the data are
328 compared as 10-second running averages, the correlation coefficients squared were
329 generally higher than 0.8. Approximately 10-second averaging times were therefore
330 necessary for the DustTraks to provide equivalent data.

331

332 *3.1.2 Uncertainty.* To evaluate measurement uncertainty the coefficient of variation (CV)
333 of the difference between the PM₁₀ measured by the collocated samplers was
334 characterized. We calculated the mean and standard deviation of the percent absolute
335 difference between the three samplers in all combinations 2-1, 3-1, and 3-2. The CV's for
336 the vehicle wake characterization component were 0.8% and 7.1% when characterizing
337 roadway emissions.

338 *3.1.3 Evaluation of PM₁₀ Losses in the Sampling Line.* It was necessary to use sampling
339 lines ranging in length from 1 to 3 m to change positions during the on-board wake
340 characterization. To evaluate losses of particles within the sampling lines, three
341 DustTraks equipped with PM₁₀ inlets were collocated and sampled ambient air for two
342 hours, alternating 10 minutes without tubing and 10 minutes with tubing 1.70 m long.
343 Data were collected as 30-second averages. Based on the means, the tubing caused a loss
344 of PM₁₀ ranging from 21 to 29 percent depending on which DustTrak was evaluated. In
345 addition, the data sets for each DustTrak were compared with and without the tubing
346 using the Wilcoxon¹⁸ non-parametric ranking test. For all three DustTraks the without-
347 tubing and with-tubing data sets were shown to not be equivalent at the $p < 0.05$ level.
348 The loss of PM₁₀ due to the tubing, therefore, is significant, but nearly equal among the

Commented [DJ1]: You could say "coefficients of determination, (r^2)" instead here, if you wish

Commented [DJ2]: Same thing here

Commented [DJ3]: Do you mean "three" here to distinguish the number from 2 samplers in the previous paragraph?

Commented [DJ4]: You have 3 combinations in the previous line but only two results here. Did you mean to add a 3rd number?

Commented [DJ5]: Is this the correct probability level used for rejection of the null hypothesis?

three lines. No corrections were therefore made since the tubing was only used for the wake characterization tests in which the relative responses were compared.

3.2 Wake Characterization

The objective of the wake characterization was to determine the following:

- 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.
- The appropriate sampling position.

The three DustTraks were collocated during the characterization of the vehicle wake. Identical lengths of the anti-static tubing were used from each sampling point to the DustTraks' inlets. 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.

For actual testing, the tubing was manually interfaced to the DustTrak to obtain sequential samples at each test point. Figures 7 and 8 summarize data from several test runs.

Figure 7. PM10 concentrations while towing the test trailer in configuration 1.

Figure 8. PM₁₀ concentrations while towing the test trailer in configuration 2.

These figures show that the PM₁₀ concentrations for this roadway were well within the detection limit of the DustTraks. This was expected since we were sampling directly in the plume of the vehicle and the chosen roadway was little-traveled and weathered. These plots also show a significant increase in concentration as the speed increases.

The PM₁₀ concentration of the plume clearly drops at the centerline sampling point near the top of the tow vehicle and drops further at the sampling point above the top. At the centerline 1.98 m high sampling point the overall average ratio (averaged over all speeds) of PM₁₀ concentration to that of the reference point at 0.78 m was 0.45 whereas at the 2.59 m high center sampling point the ratio was 0.31.

The PM₁₀ concentration drops rapidly when sampling just beyond the horizontal edge of the 2 m wide tow vehicle (1.22 m from the centerline). The overall ratio of PM₁₀ concentrations at the sampling point 1.22 m to the left of the centerline at a height of 1.98m (averaged over all speeds) compared to the reference point were 0.12 and 0.14 for these two test run examples.

These typical data show that, at the 4.25 m sampled distance, the plume is confined primarily within the frontal area of the tow vehicle. We therefore chose, for roadway emission measurements a sampling probe on the centerline of the test vehicle located 4.25m behind the rear bumper and 0.78m above the ground. This was the location of highest PM₁₀ concentrations behind the vehicle, and should approximately represent the mean concentration within the wake since the drop in concentration with height within the frontal area of the vehicle is somewhat compensated by the PM₁₀ concentrations outside the frontal area. While a more robust measurement of PM₁₀ concentrations within the wake might be desirable when performing roadway emission measurements, this was not justified by the inherent concentration variability when generating these dust plumes. This single sampling position supported the primary goal of comparing these PM₁₀ emission rates with those determined by silt sampling.

Commented [DJ6]: Does this better align with Figure 8?

Similar wake characterization results were reported by Mathissen et al.¹⁵ using NO as a tracer gas with 22 sampling positions 1.8 and 4 m behind a tow vehicle. They assumed that the tracer gas release positions would simulate the resuspension process and that the gas would disperse the same way as resuspended particles. Their optimum sampling height was determined to be 0.65m for a slightly lower tow vehicle.

3.3 PM₁₀ emission measurements on various road types and comparison with the AP-42 equation for segment in which silt sampling was conducted.

Tests were conducted over seven days in early November 2005 when no rain occurred; the most recent measurable antecedent rain occurred on October 25. Table 1 summarizes the number of test runs that were conducted on each route along with the average PM₁₀ emission rate and standard deviation of this average. The DustTrak values were zero-corrected, and 5-second medians and running averages were calculated.

Since the DustTrak is an optical method of measuring PM₁₀, the values are dependent on the size distribution and index of refraction of the PM being measured. It was therefore necessary to calibrate the response to PM₁₀ determined by mass measurement in order to be equivalent to the EPA reference method upon which the AP-42 equation is based. One approach was to collect filter samples using an EPA approved PM₁₀ inlet collocated with the rear DustTrak. A Graseby-Andersen model 246B (Smyrna, GA) inlet was used, but modified so that the inlet attaches directly to a 47mm filter holder during test runs. Since this inlet was designed for stationary use, it may not sample isokinetically at the higher highway speeds. To collect sufficient sample for accurate weighing, one sample was collected each day. The 47mm Gelman Teflo filters were equilibrated to 70°F and 45% RH prior to weighing. The average DustTrak PM₁₀ was calculated for the intervals in which the filters were collected. Plotting the seven data points from the filters versus the averaged DustTrak values showed significant scatter and resulted in a slope of 1.25 with an R² of 0.83. This is a lower correction factor than the slope of 2.4 (R² 0.84) reported by Langston et al¹⁹ when Las Vegas road dust was suspended in a chamber and sampled with filters while collecting measurements from a DustTrak. Although both correction

factors have their limitations, the DustTrak data reported here were multiplied by the 1.25 correction factor.

The road PM emission factor was determined by subtracting the concentrations measured by the front DustTrak from that of the one mounted in the trailer. Data were excluded when the test vehicle speed was less than 15 km/hr. since one would not expect a well-developed plume under these low speed conditions. These net PM concentrations were multiplied by 3.3 m², the approximate frontal area of the tow vehicle, to estimate the emission rate in mg/m.

446

		Loop A			Loop B			Loop C	
		Mean EF	Std. Dev.		Mean EF	Std. Dev.		Mean EF	Std. Dev.
Day	# Laps	mg/m	mg/m	# Laps	mg/m	mg/m	# Laps	mg/m	mg/m
1	20	0.52	0.19	2	0.71	NA	1	0.015	NA
2							9	0.029	0.008
3				15	0.68	0.21			
4				16	0.35	0.09			
5				17	0.45	0.13			
6				18	1.07	0.24			
7							6	0.034	0.034

447

448 **Table 1.** Summary of conducted roadway tests and determined PM emission rates.

449

450 Figure 9 shows a time series of the test runs for Loop A conducted on day 1. The PM₁₀
 451 emission rates are highly variable, a result that is consistent with the construction
 452 activities being performed along the route. The emission pattern with location, however,
 453 is accurately replicated from run-to-run. This is also shown by the overall relative
 454 variability from test run to test run being 37%, which given the variation of traffic and
 455 observed dust production showed that the measurements were quite reproducible.

456

457 The PM₁₀ emission rate was also calculated for specific segments where silt sampling
 458 was conducted. For test loop A the entire section of Gowan Road was defined the
 459 segment. The mean PM₁₀ emission rate for all traverses was 0.94 mg/m with a standard
 460 deviation of 0.46 mg/m.

Figure 9. Time series PM₁₀ emission rates for Loop A on Day 1.

Figure 10 shows a time series of the test runs for Loop B conducted on day 6. The PM₁₀ emission rates are highly variable as expected. The patterns of the emission rates are again quite reproducible from run-to-run. While the averaged PM₁₀ emission rates were similar to Loop A, the peak rates were much higher. This may be due to either greater impact from construction activities or greater speeds obtained on this mix of roadway types.

As shown in Table 1, the overall relative variability ranged from 23-30%. Days 4 and 5 were weekend days and emissions were significantly lower by almost a factor of two.

The jump in emission rate from Sunday to Monday was particularly striking. The average speed was consistent at 46 km/h for all four days. This change indicates the impact of construction activities on emissions during the week days.

Commented [DJ7]: My version of the manuscript couldn't show a Figure 9

476 The mean PM₁₀ emission rates were calculated for each of the three segments for which
477 silt sampling was conducted. The following overall average (all days) emission rate
478 results were obtained:

479

480 Emerald Stone and Sapphire Light: average: 2.20 mg/m; standard deviation: 1.18 mg/m

481 Lone Mountain and Losee: average: 0.43 mg/m; standard deviation: 0.68 mg/m

482 Goldfield and Washburn: average: 0.78 mg/m; standard deviation: 1.27 mg/m

483

484

485

486 **Figure 10.** Time series PM₁₀ emission rates for Loop B on Day 6.

487

488

489 Figure 11 shows time series plots of the emission rates for loop C on day 2. The average

490 PM₁₀ emission rate was 0.027 mg/m, about a factor of twenty less than the other two

491 loops that were chosen for high potential PM₁₀ emission rates. Despite these much lower

492 rates, the relative variability was 30%, consistent with values obtained from the other two

493 loops. All the mean loop speeds were consistently 50 km/h.

Commented [DJ8]: My version of Word (Mac 2016)
couldn't show Fig 10

494

495 The mean PM₁₀ emission rates were calculated for each of the three segments for which
496 silt sampling was conducted. The following average (all days) PM₁₀ emission rates were
497 obtained:

498 Crestdale and Hillpointe: mean = 0.04 mg/m; standard deviation = 0.07 mg/m

499 Banbury Cross and Crestdale: mean = 0.02 mg/m; standard deviation = 0.02 mg/m

500 Aspen Glow and Warm Walnut: mean = 0.17 mg/m; standard deviation = 0.10 mg/m

501

502 An emission factor of 0.02 mg/km represents a net concentration difference of only 0.005
503 mg/m³, which is near the detection limit of the DustTrak. The conclusion is that
504 comparison of emission rates with silt sampling on “clean” portions of roads will
505 generally be near the SCAMPER detection limit and therefore will produce data with
506 lower confidence limits.

507

508 **Figure 11.** Time series PM₁₀ emission rates for Loop C on Day 2

509

510 Figure 12 is a plot of the emission factors calculated from AP-42 silt sampling compared
511 with that of the average value obtained from the SCAMPER (using the mass correction

512 factor of 1.25) for the seven segments of roads where the silt sampling was conducted.
513 The R^2 value from the least-squares regression is 0.90, and the slope is 0.60, indicating
514 that the SCAMPER emission rates are somewhat lower than those derived from silt
515 sampling. The least-square regression line, however, is considerably influenced by a
516 single high emission location (Emerald Stone and Sapphire Light). The primary
517 conclusion is that SCAMPER and AP-42 emission rates are generally correlated as well
518 as can be expected given the variability of the emission source.

519
520
521
522

Commented [DJ9]: Do you wish to regenerate the correlation without that high value and report the R^2 and slope for that one?

523
524
525 **Figure 12.** Comparison of PM_{10} emission rates determined by AP-42 silt sampling and
526 the average segment values obtained with the SCAMPER.
527

528 3.4 Discussion

529 Due to the variability of on-road emissions and the difficulty in accurately measuring
530 them there is a wide variation in results within various studies and between them. The
531 AP-42 PM_{10} paved roadway emissions factor equation has been revised several times
532 based on additional studies and review of the data quality of all of the studies performed⁹.

533

AP-42 currently recommends collecting site-specific data for silt loading, since the loading depends on many uncontrolled variables. In absence of site-specific data, AP-42 suggests four default silt loadings depending on the daily traffic count, but warns that these are “order of magnitude” values. The silt loading sites used in this study ranged from arterial to local roads and measured silt loadings ranged from 0.03-4.2 g/m². The corresponding AP-42 default values ranged from 0.06-0.6 g/m². The high value of 4.2 g/m² was located on a road with significant impact from construction activities and would be better compared with the AP-42 default value of 2 mg/m² for roads upon which anti-skid abrasive material had been applied. The measured silt loading values were therefore well within the AP-42 default values.

4. Conclusion

Real-time measurements in front of and behind a vehicle were found to be a useful method to characterize PM₁₀ emission rates from paved roads that correlated well with the AP-42 silt sampling method. Unlike the upwind-downwind approach, a significant concentration differential was measured. The emission rate could also be calculated without applying modeling techniques, and calibration using an upwind-downwind approach that is time-consuming and likely to have high uncertainty. Sampling behind a vehicle in the geometric center of the frontal area, unlike within a vehicle’s wheel well, would also allow easy switching among vehicles without the need for calibration due to changes in wheel well flow characteristics and sampling probe location. This would allow a variety of vehicles to be tested to more accurately determine the PM₁₀ emission rates of roadways. An added advantage of this sampling location is that sample dilution is not required to avoid coincidence effects of the optical measurement technique. The concentrations encountered are expected to be well within the sensor’s capability, even on unpaved roads.

We therefore believe that this approach is more accurate than the upwind-downwind approaches used in the past, is comparable to the AP-42 method, and also allows the comprehensive, high-resolution and cost-effective measurement of entire roadways.

REFERENCES

1. Chow, J.C.; Watson, J.G.; Lowenthal, D.H.; Solomon, P.A.; Magliano, K.; Ziman, S.; Richards, L.W., 1992. PM₁₀ source Apportionment in California's San Joaquin Valley; *Atmos. Environ.*, 26A, 3335-3354.
2. Zimmer, R.A.; Reeser, W.K.; Cummins, P. Evaluation of PM₁₀ Emission Factors for Paved Streets. In: PM₁₀ Standards and Nontraditional Particulate Source Controls Volume I, J.C. Chow and D. M. Ono Eds., Air and Waste Management Association, Pittsburgh, PA, 1992.
3. Gaffney, P.; Bode, R.; Murchison, L. PM₁₀ Emission Inventory Improvement Program for California; California Air Resources Board, 1995.
4. Venkatram, A.; Fitz, D.; Bumiller, K.; Du, S.; Boeck, M.; Ganguly, C., 1999. Using a dispersion model to estimate emission rates of particulate matter from paved roads; *Atmos. Environ.*, 33, 1093-1102.
5. Ashbaugh, L.; Chang, D.; Flocchini, R.G.; Carvacho, O.F.; James, T.A.; Matsumara, R.T. Traffic Generated PM₁₀ "Hot Spots"; Air Quality Group, Crocker Nuclear Laboratory, University of California, Davis, August 1996.
6. Harding Lawson Associates, Final Report for the 1993-1994 ADEQ Paved Road Emissions Research Study; Maricopa Association of Governments, 1996.
7. Kantamaneni, R.; Adams, G.; Barnesberger, L.; Allwine, E.; Westberg, H.; Lamb, B.; Claiborn, C., 1996. The measurement of roadway PM₁₀ emission rates using atmospheric tracer ratio techniques; *Atmos. Environ.*, 24, 4209-4223.
8. Claiborn, C.; Mitra, A.; Adams, G.; Barnesberger, L.; Allwine, G.; Kantamaneni, R.; Lamb, B.; Westberg, H., 1995. Evaluation of PM₁₀ emission rates from paved and unpaved roads using tracer techniques; *Atmos. Environ.*, 29, 1075-1089.
9. U.S. Environmental Protection Agency. AP-42 Compilation of Air Emission Factors, Section 13-2.1 Paved Roads, January 2011.
10. Cowherd, C., Jr.; Englehart, P.J. Paved Road Particulate Emissions; EPA-600/7-84-077. U.S. Environmental Protection Agency, Washington, D.C., 1984.
11. Abu-Allaban, M.; Gilles, J.A.; Gertler, A.W., 2003. Application of a multi-lag regression approach to determine on-road PM₁₀ and PM_{2.5} emission rates; *Atmos. Environ.*, 37, 5157-5164.
12. Abu-Allaban, M.; Gilles, J.A.; Gertler, A.W., 2003. Tailpipe, resuspended road dust, and brake-wear emission factors from on-road vehicles; *Atmos. Environ.*, 37, 5283-5293.
13. Nicholson, K.W.; Branson, J.R., 1990. Factors affecting resuspension by road traffic; *Sci. Tot. Environ.*, 93, 349-358.
14. Etyemezian, V.; Kuhns, H.; Gillies, J.; Green, M.; Pitchford, M.; Watson, J., 2003. Vehicle-based road dust emission measurement: I – methods and calibration; *Atmos. Environ.*, 37, 4559-4571.
15. Mathissen, M.; Scheer, V.; Kirchner, U.; Vogt, R.; Benter, T., 2012. Non-exhaust PM emission measurements of a light duty vehicle with a mobile trailer; *Atmos Environ.*, 59, 232-242.

16. White, W.H.; Macias, E.S.; Nininger, R.C.; Schorran, D., 1994. Size-resolved measurements of light scattering by ambient particles in the Southwestern U.S.A; Atmos. Environ., 28, 909-921.
17. Etyemezian, V.; Ahonen, S.; Nikolic, D.; Gilles, J.; Kuhns, H.; Gillette, D.; Veranth, J., 2004. Deposition and removal of fugitive dust in the Arid southwestern United States; J. Air Waste Manage. Assoc., 54: 1099-1111.
18. Mendenhall, W. Introduction to Probability and Statistics. Duxbury Press, Belmont, CA, 1971.
19. Langston, R., Russell S. Merle, R.S., Jr., Hart, D. Etyemezian, V., Kuhns, H., Gillies, J., Fitz, D., Bumiller, K., and James, D. (2007) The Preferred Alternative Method for Measuring Paved-Road Dust Emissions for Emission Inventories – “Mobile Technologies versus the Traditional AP-42 Methodology. Presented at the 16th Annual International Emissions Inventory Conference, Raleigh, NC May 15-17.

ABOUT THE AUTHORS

Dennis R. Fitz is an Associate Researcher, Kurt Bumiller is a Senior Development Engineer, and Charles Bufalino is a Staff Research Associate. These authors are with the Center for Environmental Research and Technology at the University of California, Riverside.

David E. James is an Associate Professor of Civil and Environmental Engineering at the University of Nevada, Las Vegas.

ACKNOWLEDGMENTS

We wish to thank the Clark County Department of Environmental Quality for supporting this research and staff members Russ Merle and Rodney Langston for their help in the field measurements.

Commented [DJ10]: Hi Dennis, did you mean “retired”?