A Semi-portable Enclosure to Measure Emission Rates from Small Fugitive Sources: Leaf Blower Dust Example

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ABSTRACT

Small fugitive particulate matter (PM) sources may make significant impacts to urban air pollution. The PM emission rates for most of these have not been measured due to the difficulty involved. The EPA addresses some fugitive emission rates in their AP-42 document and these generally were developed from upwind-downwind sampling. A semi-portable enclosure was designed and evaluated for determining PM emission rates from landscape cleaning operations such as leaf blowing, raking, and sweeping. The method consisted of an enclosure to confine PM emissions while performing these activities on actual in-use surfaces while making measurements using real-time PM analyzers. In this enclosure the cleaning equipment could be operated in the same way as normally done. To evaluate the enclosure's mixing and exchange rates, propene gas was released and used as a tracer. PM emission rates were then calculated from measured concentrations, the dimensions of the enclosure, and area over which the cleaning operation was performed. To evaluate this method, cleaning operations were performed after the pavement floor was vacuumed and then spiked with surrogate debris. To make actual emission rate measurements, the enclosure was placed over landscape with debris from normal grounds maintenance activities. Emission rates were determined as a function of surface type and cleaning tool used. The method would be appropriate for many types of small fugitive dust sources.

Keywords: Particulate matter, Test enclosure, Fugitive dust, PM_{10}

1 INTRODUCTION

In epidemiological studies particulate matter (PM) has been shown to cause a wide variety of adverse health effects and to contribute to premature deaths (Pope *et al.*, 1995; Layden *et al.* 2006). In order to mitigate the effects of this pollutant, the sources of the PM should be known as well as possible so that effective control methods may be applied. Receptor modeling by Chow *et al.* (1992) has shown that PM_{10} of geologic origin is often a significant contributor to the PM concentrations in urban areas that are not in PM attainment.

Landscape cleaning operations often appear to be obvious sources of particulate emissions and quite likely include geologic dust. To compile emission inventories, many agencies use the AP-42 document compiled by the U.S. Environmental Protection Agency (U.S. EPA) to estimate PM emission rates from various processes (U.S. EPA, 1995). While this document has a section on fugitive dust emission rate measurements and default values, it focused on industrial emissions, not small consumer sources. For example, in order to estimate PM emission rates from leaf blowers, the California Air Resources Board (2000) used estimates based on engineering principles and assumptions in addition to silt loadings measured in the gutters of paved roadways. These estimations of emission rates have not been validated and therefore have unknown accuracy.

The objective of this study was to develop a method to measure PM emission rates from small fugitive dust generating activities. These emission rates combined with activity data could then
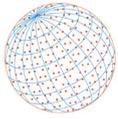
be used to determine the significance of such activities and the need for mitigation. Landscape cleaning operations such as leaf blowers and their alternatives were chosen to measure their PM$_{10}$ emission rates using actual surfaces on which they are most commonly performed. While fugitive emission sources are generally quantified using upwind-downwind measurements coupled with either flux measurements or modeling (dispersion and/or receptor), this was impractical for landscape cleaning operations since a single cleaning operation would significantly and irreversibly alter the characteristics of the surface to be cleaned.

2 METHODS

The approach used in measuring the PM emissions from landscape cleaning operations was based on having the tool clean a defined area in an enclosed space to confine the emissions and monitoring the PM concentrations in real-time until they stabilized. This would indicate that the PM inside the enclosure was well mixed, and concentrations at this point could be used to calculate the emission rate before more particles settled out. To determine the extent of mixing and the air exchange rate, propene tracer gas was injected into the enclosure just prior to the cleaning operation and the concentrations were then measured in real-time during the cleaning operation and mixing periods. The details of this approach can be found in the final report (Fitz, 2006). Some of the results were also presented at a U.S. PEA conference (Fitz *et al.*, 2006) in a presentation that focused on a 20 m long chamber, which was used initially to evaluate the approach, but was too unwieldly to move from location to location.

In order to evaluate the enclosure method on different surfaces, a surrogate mixture of soil and vegetative debris was developed that would be representative of those surfaces in actual use. With this consistent medium, various cleaning devices could be compared and the measurement variability determined. Emission measurements were then made *in situ* on actual landscape next to maintenance operations.

The enclosure had to be large enough to operate the devices and long enough to generate consistent emissions while being of a manageable size and weight so that two people could move it to various locations. A ten meter long enclosure was constructed that was two meters high and wide. The frame for the enclosure was constructed of 1-inch (2.5 cm) PVC pipe using aluminum slip-on framing fittings. The frame was then covered with polyethylene sheeting which was held to the ground with sand bags. Fig. 1 shows the enclosure and frame supporting the polyethylene sheeting.

Thermo Systems Inc. (Shoreview, MN) Model 8520 DustTrak™ aerosol monitors placed directly in the enclosure were used to measure total suspended particulate matter (TSP), PM$_{10}$, and PM$_{2.5}$ in real-time. The size cuts were made by impactors that came with the instruments. PM concentrations were determined by measuring the intensity of light from a diode laser that is scattered by 90°. The DustTraks were calibrated at the factory using Arizona road dust (NIST SRM 8632). Since the response of these analyzers depends on the size-distribution and optical properties of the particulate matter, collocated filter samplers were used during each test to

Fig. 1. Photographs of the enclosure and frame.
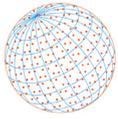
verify the mass-accuracy for the aerosol being measured. For all measurements the instruments' time constant was set at one second, which is the fastest available rate. The response to zero air was checked daily by placing a filter on the inlet. Data were collected using instruments' internal storage.

Filter samples were collected on 47 mm Pall (Port Washington, NY) Teflo™ filters with a 2.0 μm pore size. Graseby-Andersen model 246B Federal Reference Method inlets placed directly in the enclosure were used to collect filter samples with a PM_{10} size-cut. These inlets were modified to allow a Savillex (Eden Prairie, MN) PFA Teflon open-face filter assembly to be directly attached to the inlet. These filter samplers operated at 16.7 L min^{-1}. A Sensidyne (St. Petersburg, FL) model 240 cyclone also placed directly in the enclosure and sampling at approximately 110 L min^{-1} was used to provide the $PM_{2.5}$ cut-point. The outlet of the cyclone was directly attached to a closed-face Savillex filter assembly. Prior to weighing, filters were equilibrated at 23°C and 40% relative humidity for at least 24 hours prior. Filter were weighed before and after sampling to the nearest μg with a Cahn Model 34 microbalance.

For each test propene gas was introduced into the enclosure to determine the mixing and exchange rates. A polyethylene bag was filled with 3 L of pure propene gas and then released in several seconds over the entire length of the enclosure. The propene concentrations were measured using a RAE Systems (San Jose, CA) ppbRAE photoionization hydrocarbon analyzer using a 10.3 electron-volt detector. The lower detection limit for propene is approximately 50 ppb. The calculated propene concentration was approximately 75 ppm after mixing, well above this detection limit. The concentration data as a function of time was obtained with 5-sec resolution from the instrument's internal data logger. The instrument was placed 2 m in at a height of 2 m.

The hand-held electric leaf blower was a Black & Decker (Towson, MD) Model BV 4000. A rake and push broom were procured from a home supply store for examining alternate methods to leaf blowers.

Collected debris was sorted into six size fractions using a W.S. Tyler Co. (Mentor, OH) Ro-Tap model RX-29 sieve shaker. The five sieves used were No. 3/8 (9500 μm), No. 4 (4750 μm), No 18 (1000 μm), No. 40 (425 μm) and No. 200 (75 μm). Soil used to make the surrogate debris was made by sieving local soil and using the soil that passed through the No. 40 sieve.

3 RESULTS

Emission factors for fugitive sources are generally determined by performing upwind/downwind measurement studies and using either flux measurements or modeling to calculate emission rates. The flux measurement method was used to determine many of the fugitive dust emission factors presented in the U.S. EPA AP-42 document (Cowherd *et al.*, 1974). These included paved and unpaved roads, construction activities, and surface mining sources. This approach is useful for continuous or semi-continuous fugitive emissions that allows enough time to collect measureable and representative data. For activities such as using a leaf blower to clean a surface, there is insufficient time to collect a representative sample and the activity cannot be repeated since the surface is now "clean". Another approach has been to perform the activity in a tunnel in which the emissions can be briefly confined so that more robust measurements can be made. This was reported by Fitz and Bumiller (2000) to determine the PM emission rates from street sweepers. While this approach was practical and useful for comparing various models of sweepers, there was significant uncertainty in the determination of representative concentrations and air velocity. The approach presented here was to confine the activity to an enclosure of known size and wait until the PM concentrations were generally uniform.

3.1 Method Validation

In order to evaluate the enclosure method to determine leaf blower emissions, it was necessary replicate tests as precisely as possible. A surrogate representative sample of debris was needed. Following landscaping staff at University of California campuses, 32 locations were selected from areas that were about to be cleaned by leaf blowers or sweeping. One-meter square areas were vacuumed and the collected material was separated into six size ranges using sieves.
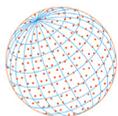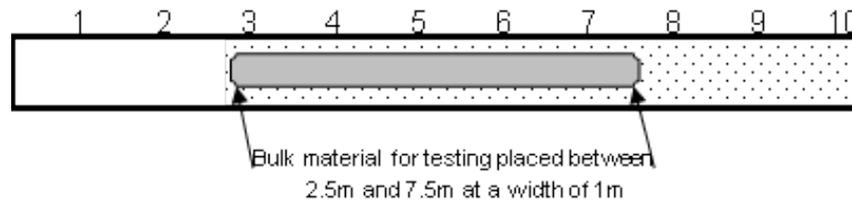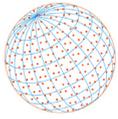
Table 1 shows the mass found in each of the six size fractions along with the composite mass for each of the samples collected. The total mass ranged from 2 to 377 grams. Some vegetative material was found in the finer sieve fractions and some soil in the larger sieve fractions. Based on these results, a surrogate material was chosen that consisted of 120 g of soil (mass after passing through a No. 40 sieve), 60 g of leaves and 60 g of grass clippings. This surrogate mixture was spread evenly over a 5 m^2 area in the 10 m long enclosure as shown in Fig. 2.

Since eight DustTraks would be used to characterize the PM concentrations in the enclosure, it was necessary to compare them collocated using surrogate material dispersed by a leaf blower in the enclosure. Correction factors were then calculated to normalize the responses of the DustTraks to a single reference instrument. Separate tests were performed using the TSP, PM_{10} and $PM_{2.5}$ inlets. Fig. 3 shows a concentration time series for three separate test runs for the eight collocated DustTraks using the TSP inlet. The DustTraks were mounted at a height of 2 m and 6 m from the start point of the leaf blower at one end of the 10 m enclosure. Approximately five minutes were required for the concentrations to stabilize. The mean TSP concentrations for the eight samplers were calculated between minutes five and ten, ten and fifteen and fifteen and twenty. For each time interval, the values of each of seven DustTraks were divided by the value of the one selected as a reference. The inverse of this ratio was then used to normalize the values of each DustTrak to the reference. Table 2 shows these normalization factors obtained for the three particle cut-points for all of the DustTraks.

The homogeneity of the PM concentrations in the enclosure as a function of time was evaluated in order to measure the amount of PM generated by the cleaning device used. The goal was to

Table 1. Fractionation of debris collected immediately before actual leaf blowing/operations.

Sample Description	Total Mass (grams)	> 3/8 fraction (grams)	< 3/8, > #4 fraction (grams)	< #4, > #18 fraction (grams)	< #18, > #40 fraction (grams)	< #40, > #200 fraction (grams)	< #200 fraction (grams)
Asphalt Driveway - General cleaning	377.3	2.8	24.8	136.5	74.2	104.8	34.3
Concrete Walkway - Lawn trimmings	49.5	0.9	0.5	8.3	32.3	4.8	2.7
Textured Concrete Walkway - Lawn trimmings	10.4	0.4	1.1	5.0	2.5	1.1	0.2
Concrete Walkway - General cleaning	36.1	9.9	8.8	14.6	1.9	0.8	0.1
Brinks - General cleaning	55.2	11.5	19.6	12.3	5.4	4.9	1.5
Concrete Walkway - General cleaning	24.2	11.9	7.3	1.5	1.1	1.7	0.8
Concrete Walkway - General cleaning	10.5	0.5	0.5	4.4	1.3	2.5	1.4
Concrete Steps - General cleaning	16.2	6.9	2.9	3.6	1.5	1.2	0.2
Concrete Walkway - General cleaning	4.6	1.0	0.7	0.5	0.4	1.4	0.6
Concrete Walkway - Lawn trimmings	14.6	2.8	1.2	6.4	3.2	1.1	0.0
Concrete Walkway - Lawn trimmings	36.6	12.1	5.9	13.0	4.0	1.5	0.1
Asphalt Parking Lot - Lawn trimmings	26.2	1.9	3.6	17.3	3.1	0.2	0.0
Concrete Walkway - Lawn trimmings	2.3	0.0	0.2	0.9	0.8	0.3	0.1
Concrete Walkway - General cleaning	22.6	4.3	6.2	8.8	2.3	0.7	0.2
Asphalt Parking Lot - General cleaning	75.2	9.4	11.2	13.5	12.1	20.8	8.2
Lawn - Leaves and debris	109.7	0.0	4.3	34.9	37.1	26.1	7.2
Gutter - Debris	30.9	0.3	2.1	9.6	7.9	7.5	3.5
Concrete Walkway - Lawn trimmings	2.8	0.0	0.0	0.4	0.5	1.8	0.2
Gutter - Debris	96.8	0.1	3.2	19.8	29.6	42.1	2.0
Lawn - Leaves and debris	5.0	0.0	0.0	1.1	2.3	1.6	0.1
Asphalt Driveway - General cleaning	12.2	0.0	0.7	3.4	2.7	4.3	1.1
Packed Dirt and Gravel Parking - General cleaning	50.0	21.6	6.5	8.0	5.4	6.3	2.3
Lawn - Leaves and debris	35.0	9.1	4.1	7.7	4.2	9.1	0.8
Average	48	5	5	14	10	11	3
Minimum	2	0	0	0	0	0	0
Maximum	377	22	25	136	74	105	34
Median	26	2	3	8	3	2	1
Standard Deviation	77	6	6	28	17	23	7

**Fig. 2.** Top view of test enclosure showing test material distribution.
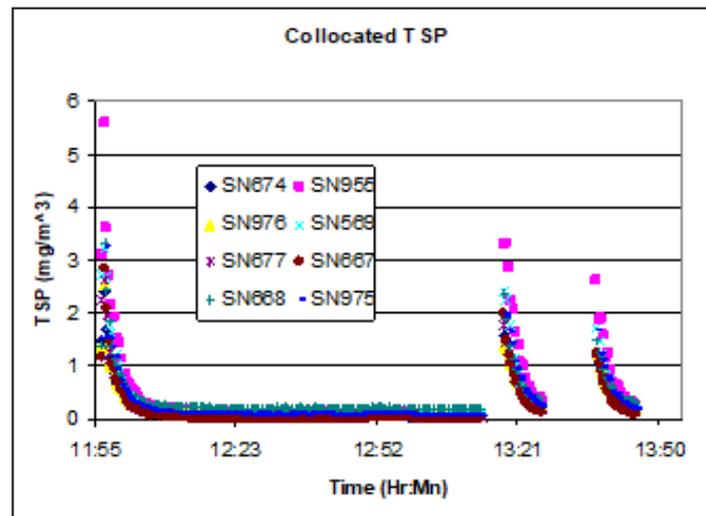
Fig. 3. TSP time-series plots for eight DustTraks collocated in 10 m enclosure.

Table 2. Collocated DustTrak™ mean response ratios.

Size	677	955	674	667	975	976	668	569
2.5	1	1.00	0.84	1.24	0.94	1.00	1.00	1.01
10	1	0.68	0.71	0.83	0.62	0.75	0.81	0.96
TSP	1	0.76	0.93	1.03	0.85	1.14	0.87	0.71

determine at what time the PM became adequately mixed to minimize the losses incurred due to particle settling. Pairs of DustTraks were placed at horizontal distances of 2, 4, 6 and 8 m from the enclosure entrance (where cleaning operations were started). One from each pair was placed at a height of 1 m and the second was placed at a height of 2 m. With this configuration both the vertical and horizontal concentration gradients could be determined at the same time. Testing of the 10 m enclosure was performed on a concrete surface using surrogate material deposited as shown in Fig. 2 and the electric leaf blower. Separate tests were performed with the DustTraks equipped with TSP, PM_{10} and $PM_{2.5}$ inlets. The results from these tests were used to determine the placement of PM samplers in performing subsequent emission rate measurements using "as found" debris.

Fig. 4 shows a time series for a homogeneity test with the DustTraks using the PM_{10} inlets. The concentrations were initially high and variable during the leaf blowing operation, especially for the locations further from the start of blowing location. During the leaf blowing operation, the high and variable concentrations are likely due to short-lived plumes of dust around each instrument. After completion of leaf blowing, the PM concentrations at all locations began to reach a more common value. Between three to six minutes after the end of the leaf blowing, the PM concentrations became nearly similar. After six minutes, concentrations continued to decrease at a near constant rate for the following twenty minutes. The tracer gas concentrations generally showed this mixing trend, although the gas is expected to mix more rapidly due to the dispersal method and not settling out, as the particles would. After being well-mixed, the tracer gas concentrations decreased
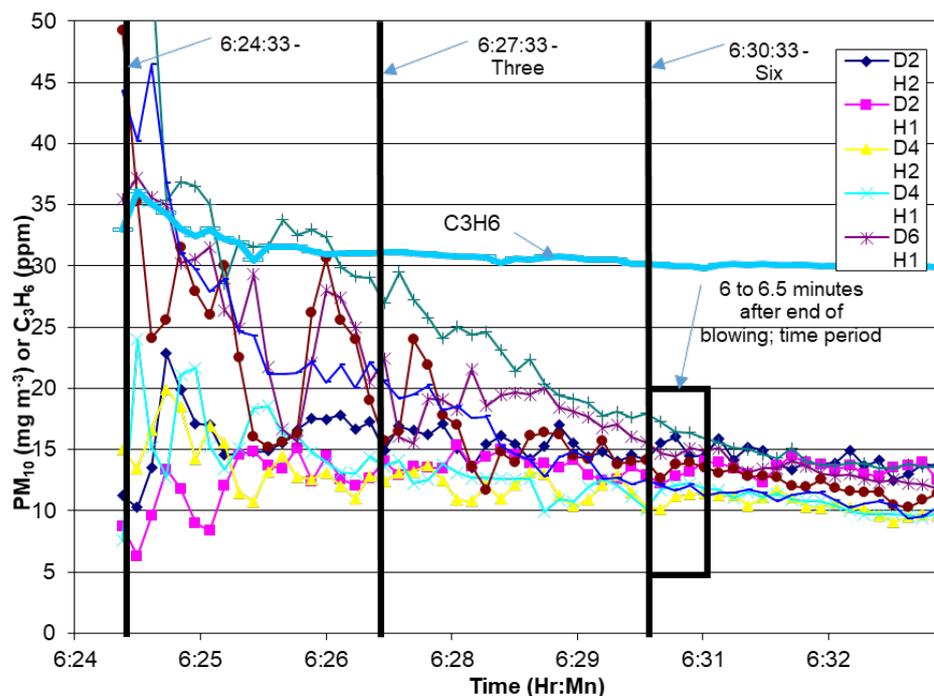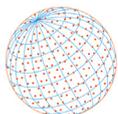
Fig. 4. Time series of DustTrak™ PM_{10} responses for horizontal and vertical distribution characterization. In the key the D value is the distance from the end of the enclosure where blowing is initiated in meters, and the H value is the height, in meters.

at a rate of approximately one percent per minute, indicating that the enclosure was slowly exchanging ambient air.

Table 3 presents the results from subsequent test runs for TSP, PM_{10} and $PM_{2.5}$ averaging the concentration data between 6.0 and 6.5 minutes after completing the blowing operation. Although there was some variability between which DustTrak™ was highest or lowest for a specific run, indicating incomplete and inconsistent mixing, these data indicate that there was sufficient mixing and repeatability within our experimental error.

Since the vertical concentration differences were approximately the same as the measurement uncertainty, for subsequent tests the DustTraks were all placed at a height of 2 m. Using the data in Table 3 shows that if only two locations were used, (2 m high, 2 m in and 2 m high and 6 m in) the results would differ from the overall mean by 18% for $PM_{2.5}$, 0% for PM_{10}, and 12% for TSP.

Sets of three DustTraks (one each of TSP, PM_{10} and $PM_{2.5}$) were then placed at two and six meters in from the start of cleaning operations. Two additional DustTraks (PM_{10} and $PM_{2.5}$) were placed at six meters in to collect collocated samples for calculating measurement precision. At a height of 2 m, the relative standard deviation for all horizontal distances was 15% for $PM_{2.5}$, 21%

Table 3. Concentration data (mg m^{-3}) from tests to determine concentration gradients in the enclosure.

Run	Size	Distance 2 M		Distance 4 M		Distance 6 M		Distance 8 M	
		Height 1 M	Height 2 M	Height 1 M	Height 2 M	Height 1 M	Height 2 M	Height 1 M	Height 2 M
0906_1	PM_{10}	12.7	15.3	11.3	10.5	14.9	13.5	17.2	12.1
0906_2	PM_{10}	12.0	11.2	11.7	11.7	14.1	13.4	15.6	12.3
0906_3	PM_{10}	6.1	7.1	7.7	7.6	9.3	9.7	12.8	11.5
0906_4	$PM_{2.5}$	2.3	1.7	2.4	1.3	1.8	1.3	1.8	1.4
0906_5	$PM_{2.5}$	1.7	1.9	2.8	1.4	2.1	1.6	2.5	1.2
0906_6	$PM_{2.5}$	2.0	1.9	2.6	1.5	3.0	1.8	2.9	1.5
0906_7	TSP	5.9	6.7	5.5	7.5	9.5	8.2	11.4	6.0
0906_8	TSP	5.9	5.7	5.8	4.5	7.6	4.7	12.5	4.0
0906_9	TSP	9.3	6.6	10.8	4.4	14.4	4.8	11.7	4.2
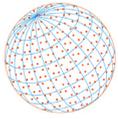
for PM$_{10}$ and 25% for TSP. This is the order of values expected assuming that smaller particles are more homogeneous due to their lower settling velocity.

3.2 Measurement Tests Performed on "as Found" Areas

The enclosure method was used to measure leaf blower PM emissions from "as found" areas that were tested as landscape maintenance staff were ready to use a leaf blower. These areas included a lawn-concrete interface, a curb gutter, a lawn, and packed soil. Fig. 5 shows the cleaning patterns used for these areas. For the lawn-concrete interface, the enclosure was placed 0.3 m onto a recently mowed lawn with remaining 1.7 m covering the adjoining concrete surface. Starting 1 m in, the leaf blower was directed to blow material directly back on to the lawn to the end of the enclosure. For the gutter, the enclosure was set up in a similar manner with 0.3 m covering the sidewalk. In this case material was either raked or blown from 1 m in straight to the end of the enclosure. To test only lawn orpacked soil surfaces, the enclosure was placed to fully cover these surfaces and raked or blown from 1 m in straight to the end of the enclosure.

The emission rate were calculated using the following equation:

$$ER = [((C2_{ave,t=6} + C6_{ave,t=6})/2) \times V_{enclosure}]/A_{debris} \tag{1}$$

The ER (mg m^{-2}) is the emission rate, C2 and C6 are the concentrations (mg m^{-3}) determined at 2 and 6 m in, respectively, from the end where cleaning operations were started averaged over the interval between 6 and 6.5 minutes after cleanup started, V is the volume of the enclosure (m^3) and A is the area (m^{-2}) over which the debris was spread.

Table 4 lists the emission rates measured varying the surface type and the cleaning tool. The emission rates varied over a much broader range than the measurement variability of 15–25% observed during the characterization of the enclosure using surrogate debris. Power blowing of grass on concrete was the most comparable to power blowing of the surrogate soil on concrete performed during the enclosure evaluation and the emission rates were similar but varied widely. Using the results from collocated DustTrak™ analyzers, the precision for the PM$_{2.5}$ measurement was calculated to be 19% and that for PM$_{10}$ 27%. These values are similar to the variability between test runs using surrogate debris.
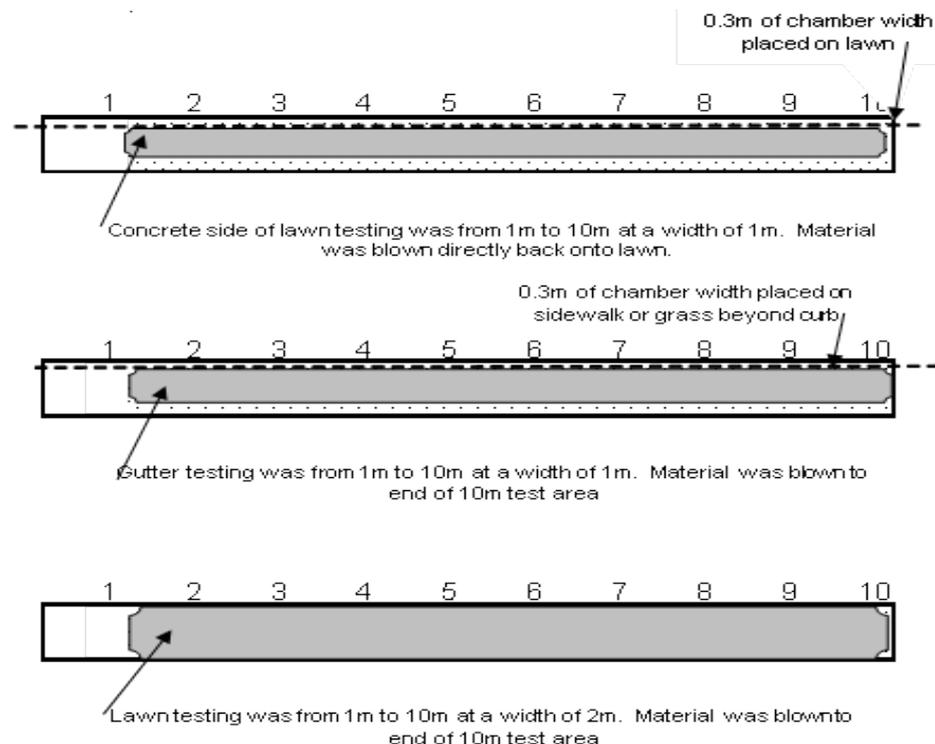
Fig. 5. Placement of the enclosure on various surfaces readies for leaf blowing operations.
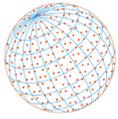
Table 4. Emission rates for leaf blowing "as found" surfaces.

| Surface Cleaned/Location	Cleaning Tool	Area Cleaned (m^{-2})	Cleaning Time (mm:ss)	PM$_{2.5}$ (mg m^{-2})	PM$_{10}$ (mg m^{-2})	TSP (mg m^{-2})
Lawn - Riverside	Elec. Leaf Blower	18	01:46	0.2	0.5	0.5
Lawn - Riverside	Elec. Leaf Blower	18	01:15	0.2	0.3	0.5
Lawn - Kearney	Elec. Leaf Blower	18	00:44	0.1	0.2	0.3
Lawn - Kearney	Elec. Leaf Blower	18	00:20	1.5	5	9
Asphalt Driveway - Riverside	Elec. Leaf Blower	10	01:43	3.8	14	15
Asphalt Driveway - Riverside	Elec. Leaf Blower	10	02:11	3.1	10	10
Asphalt Driveway - Kearney	Elec. Leaf Blower	18	00:49	2.6	11	20
Asphalt Driveway - Kearney	Elec. Leaf Blower	18	00:20	34	67	93
Gutter - Riverside	Elec. Leaf Blower	5.4	00:57	1.5	5	7
Gutter - Kearney	Elec. Leaf Blower	9	00:58	17	50	106
Gutter - Kearney	Elec. Leaf Blower	9	00:50	7	21	25
Grass on Concrete Walkway - Kearney	Elec. Leaf Blower	9	00:46	2.3	9	16
Grass on Concrete Walkway - Kearney	Elec. Leaf Blower	9	00:58	0.3	1.3	2.4
Packed Dirt Parking Lot - Kearney	Elec. Leaf Blower	18	00:35	71	118	163
Gutter - Kearney	Rake	9	01:28	0.4	2.2	3.2
Gutter - Kearney	Rake	9	01:31	0.7	3.1	3.9
Lawn - Kearney	Rake	18	02:20	0.3	0.8	1.3
Grass on Concrete Sidewalk - Kearney	Push Broom	9	01:36	2	8	10
Asphalt Driveway - Kearney	Push Broom	18	03:11	9	35	39

Several conclusions can be drawn from Table 4:

- Using a leaf blower on a gutter or grass on a concrete walkway resulted in emission rates similar to using this device on the surrogate debris.
- Using a leaf blower on packed soil resulted in the highest emission rates
- Using a leaf blower on grass did not create significant emissions
- Sweeping with a broom on concrete created significant PM emissions rates whereas sweeping asphalt resulted in much lower emission rates.
- Raking leaves did not generate significant amounts of PM.

The DustTrak™ monitor measurement is based on light scattering of particles. The amount of scattering is dependent on the particle size-distribution and the optical properties of the particulate matter. The instruments are calibrated at the factory using a suspension of Arizona road dust. Air quality regulations are often based on mass measurement methods. The relationship between the DustTrak™ measurement and the mass-based one was derived by comparing the DustTrak™ average signal integrated over the filter sampling time with the filter-based mass concentration measurement. Since PM$_{10}$ is generally the primary regulatory concern for fugitive dust, this comparison was made for the 'as found' measurements as shown by the plot in Fig. 6. The R^2 value is 0.99 and the slope is 1.46, indicating that the DustTrak™ value is significantly lower than that of the filter-based measurement for these types of particles.

4 DISCUSSION AND CONCLUSIONS

Measurements of fugitive dust emission rates are usually determined from upwind-downwind sampling by either calculating the PM flux through a horizontal plane using a high density of vertical and horizontal measurements or using a dispersion model with inputs from a smaller number of measurements. The flux method was used in determining the fugitive PM emission factors in AP-42 (U.S. EPA, 1995). This is described in detail by Cowherd *et al.* (1974). While this method is useful for continuous or semi-continuous sources, it would not be practical for cleaning operations such as leaf blowing. It would be very difficult to characterize a plume from these one-shot emission sources, since once it was conducted, the surface being cleaned would be irreversibly changed. It would be difficult to characterize the emission rate from a single plume of dust that disperses in a few seconds using a practical number of measurement instruments.
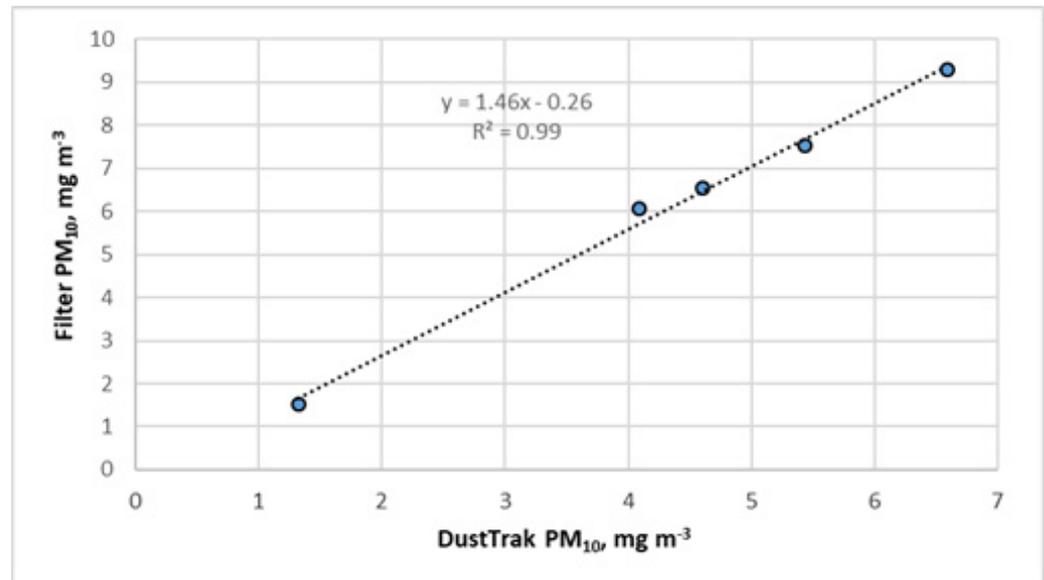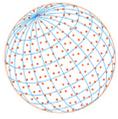
**Fig. 6.** Comparison of PM$_{10}$ of collocated DustTrak™ and filter sampler.

More recent approaches to measuring fugitive dust emissions use optical remote sensing to characterize the plume concentrations. Hashmonay *et al.* (2009) used laser transmissometry combined with a vertical array of dust monitors used to calibrate the laser data. Another approach Holman *et al.* (2001) used lidar combine with a vertical array of PM sensors. These optical techniques are more useful for large emission areas due to the long path length on which they are operated on. Enclosures have been shown to be more useful for small one-time emission sources. Roney and White (2006) used a wind tunnel to measure emissions from soil surfaces. Scaling up a wind tunnel to the size required to operate a leaf blower would not be practical due to the cost and difficulty in moving from site-to-site. Lundgren (1986) used an enclosure to characterize the emissions from handling granular mining products. This was the approach that was used in the results presented here.

A semi-portable enclosure was used to measure fugitive PM emission rates from small fugitive sources such as paved surface cleaning operations. The approach was to use an enclosure (2 m wide, 2 m high and 10 m long) in which cleaning devices such as leaf blowers, rakes, and brooms could be operated as normally conducted while confining PM emissions for concentration measurement by real-time sensors. The enclosure used a lightweight PVC pipe frame covered with polyethylene sheeting that could be moved from one location to another. To calculate the amount of PM generated in mg m^{-2}, the concentration in mg m^{-3} (once it stabilized) is multiplied by the volume of the enclosure (m^3) and then divided by the area treated in m^2.

Surrogate debris was used on an asphalt surface within the enclosure to characterize the PM variability within the enclosure, estimate the time needed for the PM concentrations to stabilize, and determine the optimum placement of the analyzer inlets. Two sample points at 2 m above the surface and 2 and 6 m along the length of the enclosure were found to be appropriate. The PM concentrations were found to be adequately stabilized after 6 minutes and concentrations were averaged between 6 and 6.5 minutes to determine the PM emission rate. Propene gas was injected prior to cleaning operations to measure the exchange rate, which was approximately 1% per minute, indicating little leakage from the enclosure. The reproducibility was found to be from 15 to 25% depending on the PM size range.

The PM$_{10}$ concentrations were stable in the test enclosure for at least 20 minutes showing that the PM$_{10}$ does not settle out rapidly and would stay in suspension long enough to contribute to the general PM$_{10}$ loading in the area. Since many urban areas are not in compliance with PM$_{10}$ standards based on adverse health effects, any additional increase in concentration would be detrimental to the health of the area's population.

The test system was then transported to areas that were about to have grounds maintenance personnel use leaf blowers for cleaning the surface. We report here for the first time the PM emission
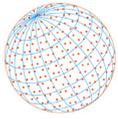
rates from surface cleaning operations using leaf blowers, rakes, and brooms on "as found" surfaces. The highest emission rate was from packed soil, while the lowest was on grass. While raking leaves resulted in low PM emission rates, using a broom resulted in higher emission rates, especially on a concrete surface. In the latter case the PM emission rates were comparable to those from a leaf blower and similar to those observed when using surrogate debris.

While using a leaf blower on asphalt driveways and street gutters is a much different process than street sweeping blowsand on asphalt as reported by Fitz and Bumiller (2000), it is interesting to compare their emission rates. The former resulted in a mean of 25 mg m^{-2} while the latter had a mean of 40 mg m^{-3}. While there was significant variability in both and too few samples to statistically treat, the emissions of these processes were qualitatively similar.

This general enclosure approach would be useful in determining emission rates for many small operations that produce fugitive emissions, whether it is a particulate or gaseous pollutant. For operations that don't include pollutants from disturbing the soil, it would be prudent to evaluate the chamber for the time needed to achieve homogeneity.

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DISCLAIMER

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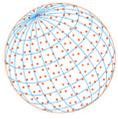
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