A case study of the impact of Winter road sand/salt and street sweeping on road dust re-entrainment

Alan Gertler\textsuperscript{a,*}, Hampden Kuhns\textsuperscript{a}, Mahmoud Abu-Allaban\textsuperscript{b}, Christopher Damm\textsuperscript{c}, John Gillies\textsuperscript{a}, Vicken Etyemezian\textsuperscript{a}, Russ Clayton\textsuperscript{d}, David Proffitt\textsuperscript{d}

\textsuperscript{a}Desert Research Institute, 2215 Raggio Parkway, Reno, NV 89512, USA
\textsuperscript{b}Faculty of Natural Resources and Environment, The Hashemite University, Zarqa, Jordan
\textsuperscript{c}Milwaukee School of Engineering, Milwaukee, WI 53202, USA
\textsuperscript{d}ARCADIS Geraghty & Miller, 4915 Prospectus Drive, Durham, NC 27713, USA

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Abstract

Resuspended road dust is an important contributor to ambient particulate matter (PM). In areas with significant snow events, the use of wintertime roadway abrasives for traction control can result in increased PM emissions. An alternative control measure is to use chemical deicers prior to a snow event to minimize ice formation and reduce the need for abrasives. Street sweeping is also commonly used to reduce the impact of re-entrained abrasive material after the snow pack on the road has cleared. In this study, we performed roadside measurements of PM flux and instrumented vehicle PM measurements to evaluate the effectiveness of street sweeping to reduce dust re-entrainment and assess the impact of abrasives and deicers on ambient PM near an alpine lake (Lake Tahoe). The results indicate use of liquid deicers contributes less to road dust emissions than abrasives. Street sweeping was found to increase the PM\textsubscript{10} re-entrainment rate of the remaining road dust. Emission factors for roads in the study area (a snowy mountain climate) tend to decrease significantly from late Spring to early Summer by as much as a factor of 4.

Keywords: Resuspended road dust; On-road emissions; PM\textsubscript{10}; PM\textsubscript{2.5}; Emission factors

1. Introduction

Many areas suffer from elevated levels of PM\textsubscript{10} and PM\textsubscript{2.5} (Molina et al., 2004). In order to meet air quality standards, there is a need to develop a better understanding of the sources of elevated particulate matter (PM). The transportation segment, in particular, has been identified as a major contributor to observed PM levels (Gertler et al., 2000). In addition to direct tailpipe emissions of particulates, mobile sources are also responsible for fugitive dusts such as those re-entrained from roadways. In the US, the inventory approach for estimating emissions is based on the US Environmental Protection Agency’s AP-42 guidelines that contain questionable emission factors for road dust re-entrainment based on a limited, narrow empirical database (Venkatram, 2000, Nicholson, 2001, Venkatram, 2001). Thus, the quality of the emission estimates and overall contributions from this source category are highly uncertain.
This uncertainty is further compounded in wintertime when the use of abrasives for traction following snow events is common. In addition, studded snow tires can increase roadway wear, leading to elevated PM levels (e.g., Kupiainen et al., 2005). In a previous study performed in the US to assess the impact of mobile sources on ambient PM$_{10}$, the impact of Winter abrasives (sand and salt) was evaluated for three cites: Albany, NY, Denver, CO, and Reno, NV (Wittorff et al., 1996). Ambient PM$_{10}$ samples were collected following Winter periods when the roads were sanded and salted and had just begun to dry out. These conditions tended to produce the maximum emissions from resuspended material. The Reno site had an average PM$_{10}$ level of 78 $\mu\text{g m}^{-3}$ with 57% of this amount attributed to resuspended sanding material. Over the sampling period, six samples exceeded 100 $\mu\text{g m}^{-3}$, with a maximum value of 144 $\mu\text{g m}^{-3}$. During 19 of the 20 measurement periods, road sand was the major contributor to ambient PM$_{10}$ in Reno. The average PM$_{10}$ in Albany during this study was 37 $\mu\text{g m}^{-3}$, with 44% of the total attributable to road sand and 17% attributed to road salt. The average PM$_{10}$ in Denver was 25 $\mu\text{g m}^{-3}$, with 59% due to road sand. Based on these results, applied road sand and salt were found to be a major contributor to ambient PM$_{10}$ loadings following wintertime storms.

To reduce the impact of traction control abrasives on PM emissions, many areas use liquid deicers, and/or street sweeping to reduce re-entrainment. In order to develop a better understanding of the effectiveness of these control measures, the objectives of this study included:

- Evaluate the contribution of wintertime road sand and liquid deicers to ambient PM$_{10}$ and PM$_{2.5}$ following a storm event.
- Assess the effectiveness of street sweeping for controlling road dust re-entrainment.
- Investigate the temporal pattern of road dust emissions from Spring to early Summer in a snowy mountain climate.

2. Methodology

In order to achieve these objectives, we applied two techniques: (1) cross roadway flux measurements to determine emissions from individual vehicles (Moosmüller et al., 1998; Abu-Allaban et al., 2003) at a single location and (2) an instrumented vehicle used to assess road dust resuspension potential (Etyemezian et al., 2003a,b) over a greater spatial area and time period. By driving the instrumented vehicle past the flux tower, we were able to link the two types of measurements and calibrate the instrumented vehicle in order to determine absolute emission factors based on the resuspension potential measurements. Thus the combination of the two methods enabled us to evaluate spatial and temporal changes in road dust re-entrainment emission rates.

Measurements were performed in the Lake Tahoe basin, located on the border between Nevada and California. The Lake Tahoe Basin is a bowl-shaped watershed dominated by Lake Tahoe, a high elevation (1898 m) lake that occupies 37% of total area in the Basin. Lake Tahoe is known for its exceptional clarity, which makes Lake Tahoe arguably one of the most scenic lakes in the world. Part of the motivation for performing this work in the Lake Tahoe basin is the fact that the lake has been experiencing a decline in water clarity of 0.25 m yr$^{-1}$, which has been partially attributed to the deposition of resuspended road dust. A second motivating factor was the lake experiences frequent snow events and roadway abrasives and deicers are commonly used, making it an ideal location to perform a study of this nature.

2.1. Flux towers and ambient monitors

The flux tower measurements were based on an upwind/downwind technique that has been used by other investigators (Moosmüller et al., 1998; Abu-Allaban et al., 2003). In this case, one tower was setup ~1.5 m away from the lane edge on a section of Highway 28 (Fig. 1). Historical meteorological data indicated that winds would be predominantly from the West to southwest with speeds increasing in the afternoon.

The downwind tower was instrumented with eight TSI DustTraks (four with PM$_{10}$ inlets and four with PM$_{2.5}$ inlets) mounted at 0.5, 1, 2 and 3 m above the level of the travel lane. The DustTraks use 90° light scattering to measured particle concentrations at intervals of 1 s. The continuous DustTrak measurements were compared with integrated concentration results obtained from filter-based samplers. The DustTrak results were lower than those obtained using the filter-based technique and ranged between 59% and 89% (Kuhns et al., 2004). One combined
wind vane/anemometer was mounted ~2 m from the tower to record wind speed and direction at 2 m above the level of the lane. Assuming a log profile for the wind speed, use of the 2 m height overestimates the average wind speed between 0.5 and 3 m by about 7%. The meteorological data were stored on a data logger (Campbell Scientific, Model # 10X) at 1 s intervals. A Metro Count road tube counter was installed across the road to record gross vehicle type (based on axle spacing), speed and direction. Background PM concentrations between PM peaks were less than 7 $\mu$g m$^{-3}$ on all days.

The flux of PM perpendicular to the roadway was calculated using the equation

$$
\text{Flux} (\text{mg m}^{-1}) = \sum_{i=1}^{n} \sum_{j=1}^{4} u_i (\text{m s}^{-1}) \cos(\theta_i) C_{ij} (\text{mg m}^{-3}) \times \Delta z_j (\text{m}) \Delta t_i (\text{s}),
$$

where Flux is the total flux of PM in mg m$^{-1}$ perpendicular to the road, $\theta$ is the angle between the wind direction and a line perpendicular to the road, $i$ corresponds to a data point for a given time, $n$ is the number of data points in the sampling period, $j$ is one of the four positions of the monitors on the tower, $u$ is the measured wind speed in m s$^{-1}$, $C_{ij}$ is the $i$th PM concentration measurement in mg m$^{-3}$ as measured at the $j$th monitor over the period $\Delta t$, $\Delta z$ in m is the vertical interval represented by the $j$th monitor and $\Delta t$ in s is the time between data points. The upper panel of Fig. 2 shows the average PM concentration over the sampling interval from 12:06 to 16:36 on 31 March 2003. The marker points represent the location of each DustTrak sampler measuring $C_j$. The error bars represent the vertical layer represented by each DustTrak (\$\Delta z\$). In this example, the concentration of PM$_{10}$ at the 3 m DustTrak is less than half the concentration measured at the 0.5 and 1 m levels.

Previous deployments of the flux tower on low volume (<50 vehicle h$^{-1}$) unpaved roads permitted the calculation of emission factors for individual vehicle passes (Etyemezian et al., 2003a). The intensity of traffic at this site (maximum 170 vehicle h$^{-1}$) during the study precluded the isolation of individual vehicle plumes based on the PM data set. An example time series of the 1 s PM flux rate perpendicular to the highway is shown in the lower panel of Fig. 2. Flux measurements were considered valid only when winds were blowing within 45° of perpendicular to the road. In Fig. 2, a wind direction of 240° corresponds to perpendicular to...
the road. The time series shows large variability in the flux rates associated with sporadic traffic flows and the occasional passage of heavy duty tractor trailers that suspended large amounts of road dust in their wake. Aggregate emission factors for all vehicles were calculated by dividing the total flux of PM perpendicular to the road (Eq. (1)) by the number of vehicles passing the tower:

\[
EF (g/km^{-1}) = \frac{\text{Flux (mg/m}^{-1})}{\text{TrafficVolume (vehicles)}} \times \left(\frac{1 g}{1000 mg}\right) \times (1000 m/1 km).
\]
In addition to the real-time instrumentation deployed on the flux tower, filter-based PM$_{10}$ and PM$_{2.5}$ samplers were also operated. The samples collected with these medium-volume samplers were used to determine the contribution of the observed emissions attributable to tailpipe emissions, road sanding and brine application using the methodology and source profiles described by Abu-Allaban et al. (2003). These were subtracted from the calculated aggregate emissions to yield resuspended road dust emission factors.

2.2. TRAKER vehicle

The testing re-entrained aerosol kinetic emissions from roads (TRAKER) system is an instrumented vehicle designed to measure road dust resuspension (Kuhns et al., 2001; Etyemezian et al., 2003a, b; Kuhns et al., 2005). The TRAKER is composed of a van that has been equipped with three exterior steel pipes acting as inlets for the onboard instruments. Two of the pipes are located behind the left and right front tires and are used to measure the emissions created as the tires entrain road dust. The third pipe runs along the centerline of the van underneath the body and extends through the front bumper. This pipe is the inlet for background air. Dust and exhaust emissions from other vehicles on the road can cause fluctuations in the particle concentration above the road surface. The background measurement is used to correct the measurements made behind the tires for those fluctuations. The difference in the signals between the influence monitors and the background monitor is related to the amount of road dust generated:

$$T = T_T - T_B,$$

where $T$ is the raw TRAKER signal, $T_T$ is the particle concentration measured behind the tire and $T_B$ is the background concentration.

The three exterior pipes enter the cargo compartment of the van through the underbody. Each pipe then goes into a plenum/manifold; the plenum can be used to distribute the sample air to up to five instruments. For the present study, two TSI DustTraks with 10 µm inlets were operated in parallel at each of the three inlet lines. A central computer collected all the data generated by the onboard instruments.

Prior to these measurements, the TRAKER vehicle had not been compared with directly measured paved road PM emissions. On 31 March 2003 the roadside flux tower measurements were used to calibrate the TRAKER measurements. The relationship between the TRAKER signal and the fleet average road dust PM$_{10}$ emission factor was found to be Kuhns et al., (2004)

$$EF_{(g \ km^{-1})} = 0.33(T_{(mg \ m^{-3})})^{0.33}.$$  (4)

Thus by calibrating the TRAKER signal against the flux tower, we could determine emission factors for road dust resuspension using the instrumented vehicle and link the TRAKER and flux tower results.

3. Results

Observed changes in the road dust re-entrainment for three cases are discussed in sections that follow. The three cases are: (1) following the use of a roadway deicer (NaCl brine) and abrasive using the flux tower measurements, (2) after street sweeping using the flux tower approach, and (3) the temporal and spatial changes in emissions using the instrumented vehicle measurements. For the flux tower components of the study, the average vehicle count ranged from 130 to 170 vehicle h$^{-1}$, with an average speed of between 70 and 90 km h$^{-1}$, and heavy-duty fraction of 1–4%. Traffic was free-flowing and braking was minimal during the measurement periods.

3.1. Impact of deicer and abrasive application

Paved road emission factors were calculated using the algorithm described above (see Eq. 1 and 2). The remaining calculations for each of the sampling periods are summarized in Table 1. Road tube counters were used to classify vehicles as either heavy duty or light duty based on axle spacing. The table also shows the ambient concentrations of PM$_{10}$ and PM$_{2.5}$ measured by filter samplers and DustTraks. The road tube counters were not operational when the baseline data were taken. The table shows that DustTrak PM$_{10}$ and PM$_{2.5}$ emission factors ranged from 229 and 76 mg km$^{-1}$, respectively, during baseline monitoring to 660 and 133 mg km$^{-1}$, respectively, after the second day following the storm.

A plume apportioning method to distinguish background PM from the source-generated PM (Kuhns et al., 2004) was applied to the roadside PM
concentrations. Background concentrations at the Sand Harbor site were generally very low: ~3 µg m⁻³ PM₁₀ and ~7 µg m⁻³ PM₂.⁵. Between 38% and 82% of the PM₂.⁵ and between 53% and 89% of PM₁₀ measured by the DustTraks at the 50 or 100 cm levels were associated with emissions from the roads. These results are consistent with the study mentioned earlier (Wittorff et al., 1996) in which ~60% of PM₁₀ in Albany, Denver and Reno was attributed to resuspended road dust (in Winter).

Following the application of brine (NaCl) solution and the drying out of the road surface, there was ~30% increase in the paved road emission factor for both PM₁₀ and PM₂.⁵. A prolonged 5-day snowstorm occurred between sampling periods 2 and 3. During this time an abrasive was applied to the road surface to improve traction. The abrasive consisted of a mix of hard sand and cinders, along with a small amount of salt, designed to minimize material breakdown to form PM₁₀ and PM₂.⁵ sized particles that could be re-entrained.

The emission factors measured directly after the storm and after the road had dried off (sampling period 3) showed a doubling of the PM₁₀ emission factors from 310 to 612 mg km⁻¹. PM₁₀ emissions remained elevated (660 mg km⁻¹) on the next day (sampling period 4).

### 3.2. Impact of street sweeping

In order to minimize the air quality impacts of Winter roadway abrasives, local protocols call for using street sweepers to remove residual abrasive within 4 days following the drying out of the road surface. This provided an opportunity to assess the short-term impact of street sweeping on resuspension from a heavily loaded surface. For the Nevada side of the lake where the flux tower was located, brush and water wash street sweepers were used. PM₁₀ emissions increased slightly to 735 mg km⁻¹ after the road was swept by a street sweeper (sampling period 5). For PM₂.⁵, there was a less dramatic increase after the storm event (from 99 to 112 mg km⁻¹) and a more dramatic increase after sweeping (from 133 to 211 mg km⁻¹). These results are consistent with a recent study by Kuhns et al. (2003), where the authors did not find a detectable reduction in road dust emission potential immediately after street sweeping. It is unclear if routine street sweeping reduces emissions of PM from a paved road over longer periods of time.

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Table 1: Samples collected and emission factors measured downwind of paved road on Highway 28 near Sand Harbor State Park, Lake Tahoe, NV

<table>
<thead>
<tr>
<th>Sampling period</th>
<th>Condition</th>
<th>Vehicles</th>
<th>% Heavy-duty</th>
<th>Downwind filter concentration (µg m⁻³)</th>
<th>Downwind Ave. DustTrak concentration (µg m⁻³)</th>
<th>Fraction of DustTrak signal associated w/ highway</th>
<th>Emission factor (mg km⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Baseline</td>
<td>174</td>
<td>3</td>
<td>599</td>
<td>969</td>
<td>2</td>
<td>5976</td>
</tr>
<tr>
<td>2</td>
<td>After salting</td>
<td>599</td>
<td>1</td>
<td>2.7</td>
<td>2.3</td>
<td>1</td>
<td>30</td>
</tr>
<tr>
<td>3</td>
<td>After 1st dry</td>
<td>969</td>
<td>2</td>
<td>2.3</td>
<td>5.1</td>
<td>2</td>
<td>612</td>
</tr>
<tr>
<td>4</td>
<td>Second dry/storm</td>
<td>853</td>
<td>2</td>
<td>27.7</td>
<td>5.1</td>
<td>2</td>
<td>660</td>
</tr>
<tr>
<td>5</td>
<td>After sweeping</td>
<td>963</td>
<td>4</td>
<td>63</td>
<td>14</td>
<td>4</td>
<td>735</td>
</tr>
</tbody>
</table>

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3.3. Temporal and spatial variation of emissions

For this study, 13 passes over a mountain pass (Nevada State Route 431, elev. 2700 m) under a variety of snow, sand and brine conditions were completed. In addition, nine circuits were completed around Lake Tahoe (~100 km per circuit). The lake circuits incorporated several sections of residential roads. Segments were chosen to represent various jurisdictions (e.g., Nevada and California), modal conditions and maintenance practices in the Lake Tahoe basin.

Data from the TRAKER portion of the study have been analyzed for both spatial and temporal trends. The spatial analysis includes a comparison among groups of roads as well as between Nevada and California roads. All data were imported into a GIS-format database. When attributing road dust emission potentials or emission factors to an individual link, all 1 s valid TRAKER data obtained on the same day for that link were averaged. If there were fewer than ten valid TRAKER measurements for a link, the link was not considered to have a valid measurement.

Road segments (links) were grouped, based on location, for the purpose of spatial analyses. The mutually exclusive groups correspond to roads in Incline Village, NV (NV-Urban), South Lake Tahoe, CA (CA-Urban), highways adjacent to the lake on the Nevada side (NV-Highway, Lake Level) and the California side (CA-Highway), residential roads that are not directly on the Tahoe loop in Nevada (NV-Residential) and in California (CA-Residential), Mt. Rose Pass to the Northeast of the Lake (NV-Highway, Mountain Pass) and Rt. 267 to the Northwest of the Lake (CA-Highway, Mountain Pass).

Fig. 3 shows a time series of all TRAKER data obtained during the Lake Tahoe study on measurement days when at least 500 valid data points were obtained. Emission factors over the study period ranged from 80 mg km\(^{-1}\) (18 June 2003) to 560 mg km\(^{-1}\) (8 April 2003) with an overall average of 230 mg km\(^{-1}\). The dashed line in the figure represents a best linear fit to the data (\(R^2 = 0.58\)). There is no reason to believe that the change in emission factors over time should follow a linear trend. However, the line in the figure serves to illustrate that a temporal pattern associated with these data exists. Overall, emission factors tend to decrease significantly from late Spring to early Summer by as much as a factor of four based on the linear regression. This observation applies not only to the study area as a whole, but also to individual groups of roads as is illustrated in Figs. 4 and 5.

This type of seasonal difference in emission factors between Winter/early Spring and Summer is not unique to the Lake Tahoe area. Kuhns et al. (2003) noted that in Treasure Valley, ID, emission factors for PM\(_{10}\)-paved road dust decreased between February and July 2001. Fig. 4 compares the average emission factors during early to mid-Spring (31 March to 16 May 2003) and late Spring through
mid-Summer (17 May to 17 July 2003). The May 16 date was chosen as a cutoff because data from the National Resources Conservations Service (NRCS) snow pack telemetry (SNOTEL) monitors indicate that snowfall in the area had curbed significantly after that date (see Fig. 5). The measurements indicate that emission factors decrease for several groups of roads between late Spring and early Summer by more than a factor of two.

There are several reasons why such seasonal differences may appear. First, during Winter, road sanding and de-icing materials may contribute to the loading of dust that is available for suspension by tires. Fig. 5 shows data from a SNOTEL monitor located in the Lake Tahoe Basin and emission factors from groups of roads in the Lake Tahoe Basin. The figure shows that the end of the major snowfall season (nominally 16 May 2003) coincides with a decrease in emission factors.
with an overall reduction in emission factors. While a correlation exists between PM$_{10}$ road dust emission factors and periods when snowfall occurs frequently (i.e., late Spring), it is difficult to discern from these data if there is a direct correlation between the magnitude of individual snow events and the emission factors for PM$_{10}$ road dust measured shortly after those events. Sufficient data do not exist to determine if (a) the most recent snow event has the greatest influence on the emission factors, or (b) increased emission factors are the cumulative effect of long-term snowfall. A second reason for the differences between late Spring and Summer is that precipitation during Winter may result in the movement of sediment from hillsides, unpaved road shoulders and from sources of mud (such as construction sites) onto the road surface, thereby causing an increase in the potential for PM$_{10}$ road dust emissions. Third, during the Winter, vertical mixing close to the ground occurs to a lesser degree than during warmer periods. This results in a decrease in the dispersion and movement of air pollutants emitted close to the ground. Thus, in Winter, it is possible that a larger fraction of the PM$_{10}$ dust that is emitted from vehicle travel on roads deposits back onto the road. In contrast, when atmospheric mixing occurs to a greater degree, pollutants that are emitted close to the ground may be dispersed vertically and carried further downwind of the source prior to depositing. It is unclear which of these factors most influences the apparent seasonal dependence of road dust emissions. Future work should aim to provide additional insight into the relative contribution of these three factors.

4. Summary

During Spring and early Summer 2003, a series of PM$_{10}$ and PM$_{2.5}$ roadside flux measurements were performed at a location in the Lake Tahoe basin. Experimental periods included: pre-storm baseline, application of liquid deicer, one day after a storm during which a road abrasive was applied, two days after a storm and following street sweeping. The results indicate the use of an abrasive material significantly increased the rate of road dust re-entrainment. The use of a liquid deicer (NaCl) resulted in a smaller increase in the emission rate. Street sweeping to remove deposited material from the road surface resulted in an increase in the observed emission rate. Temporal data, obtained using an instrumented vehicle on Lake Tahoe area roads, showed a significant decrease (by a factor of two–four) in road dust emission factors from late Spring to early Summer. This temporal pattern is likely due to the termination of the application of road-sanding and de-icing materials as the snow season comes to an end. While these results are specific to the Lake Tahoe basin, we would anticipate similar qualitatively similar findings in other areas were roadway abrasives and deicers are used during Winter storm events.

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