Atmospheric Pollution Research 5 (2014) 344-351

Atmospheric Pollution Research

www.atmospolres.com

Diesel particulate matter emission factors and air quality implications from in–service rail in Washington State, USA

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ABSTRACT

We sought to evaluate the air quality implications of rail traffic at two sites in Washington State. Our goals were to quantify the exposure to diesel particulate matter (DPM) and airborne coal dust from current trains for residents living near the rail lines and to measure the DPM and black carbon emission factors (EFs). We chose two sites in Washington State, one at a residence along the rail lines in the city of Seattle and one near the town of Lyle in the Columbia River Gorge (CRG). At each site, we made measurements of size–segregated particulate matter (PM_1, $PM_{2.5}$ and PM_{10}), CO_2 and meteorology, and used a motion–activated camera to capture video of each train for identification. We measured an average DPM EF of 0.94 g/kg diesel fuel, with an uncertainty of 20%, based on PM_1 and CO_2 measurements from more than 450 diesel trains. We found no significant difference in the average DPM EFs measured at the two sites. Open coal trains have a significantly higher concentration of particles greater than 1 µm diameter, likely coal dust. Measurements of black carbon (BC) at the CRG site show a strong correlation with PM_1 and give an average BC/DPM ratio of 52% from diesel rail emissions. Our measurements of $PM_{2.5}$ show that living close to the rail lines significantly increases $PM_{2.5}$ exposure. For the one month of measurements at the Seattle site, the average $PM_{2.5}$ concentration was 6.8 µg/m^3 higher near the rail lines compared to the average from several background locations. Because the excess $PM_{2.5}$ exposure for residents living near the rail lines is likely to be linearly related to the diesel rail traffic density, a 50% increase in rail traffic may put these residents over the new U.S. National Ambient Air Quality Standards, an annual average of 12 µg/m^3.

Keywords: *Diesel particulate matter, diesel emissions, train emissions, coal dust, DPM*

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Article History:
Received: 03 November 2013
Revised: 27 January 2014
Accepted: 28 January 2014

doi: 10.5094/APR.2014.040

1. Introduction

Rail is an efficient way to move people and freight. However, diesel–powered trains may have a significant impact on air quality. In Washington State, nearly all rail locomotives are powered with diesel fuel and many rail lines are located in busy urban corridors, including Seattle, Tacoma and Spokane, and also pass through the Columbia River Gorge National Scenic Area. At present, there is limited information to evaluate the air quality impacts associated with rail transport for residents living close to the train lines. Recently, there have been proposals to increase rail shipments through Washington and Oregon for transporting coal to west coast ports for export to Asian markets. One proposed facility, the Gateway Pacific Terminal near Bellingham, Washington, could export up to 54 million metric tons of coal per year (WA DOE, 2013). Similar facilities have also been proposed at two other sites in Washington and Oregon.

According to the U.S. Department of Health and Human Services, diesel particulate matter (DPM) is "reasonably anticipated to be a human carcinogen" (U.S.DHHS, 2011); in addition, the World Health Organization classifies it as "carcinogenic to humans" (WHO, 2012). In Seattle and other urban areas, DPM is the most important "air toxic" in the metropolitan area and contributes more than 80% of the risk for cancer from airborne air toxics (Keill and Maykut, 2003; PSCAA, 2005). Monitoring and a chemical mass balance model have found average DPM concentrations to range from 1.4–1.9 µg/m^3 for the Seattle area (Keill and Maykut, 2003; Maykut et al., 2003). These concentrations are about 15–20% by mass of the total $PM_{2.5}$, particulate matter with diameter less than 2.5 µm. Sources of DPM include on–road and off–road diesel trucks, ships and rail locomotives.

The U.S. Environmental Protection Agency (EPA) has developed emissions standards for new and remanufactured locomotives (40 CFR part 1033). The emission standards, in g/bhp–hr, decrease steadily for locomotives manufactured between 1973–2001 (Tier 0), 2002–2004 (Tier 1), 2005–2010 (Tier 2), 2011–2014 (Tier 3) and after 2015 (Tier 4) (U.S. EPA, 2013). For Tier 4, locomotives must meet a PM_{10} emission standard of 0.03 g/bhp–hr, or approximately 0.19 g per kg of fuel burned (U.S. EPA, 2009).

A few studies have examined rail yards as sources of air pollutants and have found that diesel fuel combustion is the primary source of $PM_{2.5}$ at these facilities. Galvis et al. (2013) looked at the influence of DPM emissions on $PM_{2.5}$ concentrations near a rail yard in Atlanta. Based on measurements up–wind/down–wind of the facility, they concluded that the average "neighborhood" contribution to $PM_{2.5}$ was 1.7 µg/m^3. They also derived fuel–based emission factors (EFs) of 0.4–2.3 grams DPM per kg of diesel fuel consumed. These EFs are not based on measurements from individual trains but were calculated using three different methods, each of which requires a different set of assumptions. Two studies on a rail yard in Roseville, CA, also found significant enhancements in $PM_{2.5}$ from the facility. Based on upwind/downwind measurements, Cahill et al. (2011) reported an average enhancement of 4.6 μg/m^3. In another study (Campbell and Fujita, 2006), larger contributions for the same facility were reported (7.2–12.2 μg/m^3). Cahill et al. (2011) also showed that the major component of aerosol mass from diesel rail facilities is from very fine PM, with diameters less than 0.26 μm. Abbasi et al. (2013) provide a review of PM concentrations inside trains and near rail lines. They report substantially elevated $PM_{2.5}$ and PM_{10} concentrations, especially in underground rail stations. Gehrig et al. (2007) examined the influence on PM_{10} concentrations from dust associated with electric trains in Switzerland. A number of previous studies have reported EFs for on–road diesel trucks and buses (Jamriska et al., 2004; Zhu et al., 2005; Cheng et al., 2006; Park et al., 2011; Dallmann et al., 2012), but to our knowledge, similar studies have not been reported for diesel rail.

In addition to DPM emissions, trains carrying coal in uncovered loads may emit coal dust into the atmosphere. This has been a topic of some controversy. Rail transport companies are attempting to mitigate this problem (see BNSF Railway, 2013), but few studies have been reported in the scientific literature. We expect that combustion–related DPM and mechanically generated coal dust are associated with very different particle sizes, so size–segregated PM should be able to distinguish these source types (Seinfeld, 1986).

Black carbon (BC) accounts for a significant fraction (44–60%) of $PM_{2.5}$ mass from diesel engines (Bond et al., 2004; Kirchstetter and Novakov, 2007; Ramanathan and Carmichael, 2008). As the major light–absorbing species in atmospheric aerosol, radiative forcing due to BC is important on a global and regional scale (Jacobson, 2001; Ramanathan and Carmichael, 2008). Furthermore, the surface properties of black carbon allow for adsorption and transport of semi–volatile compounds like polyaromatic hydrocarbons (PAHs) (Dachs and Eisenreich, 2000). BC is under scrutiny by health organizations due to its role in adverse effects caused by $PM_{2.5}$, including cardiopulmonary and respiratory disease (Jansen et al., 2005; Janssen et al., 2011; U.S. EPA, 2012).

The City of Seattle has conducted an analysis of potential impacts associated with increasing train traffic. This analysis indicates that the proposed coal export terminal would increase rail traffic by up to 18 additional trains per day if approved (City of Seattle, 2012). Given the lack of information on $PM_{2.5}$ concentrations and human exposure from diesel trains, the controversy over coal dust and the limited information on EFs from diesel trains, we sought to quantify these air quality impacts by addressing the following questions:

(i) What is the exposure to size–segregated PM (e.g., PM_1, $PM_{2.5}$ and PM_{10}) for residents living near the rail lines?

(ii) Can we estimate the potential exposure to size–segregated PM (e.g., PM_1, $PM_{2.5}$ and PM_{10}) for people living near the rail lines if rail traffic increases?

(iii) Do coal trains emit coal dust into the air?

(iv) How do the observed DPM and BC emission factors for locomotives compare with other published EFs?

To address these questions we measured size–segregated PM_1, $PM_{2.5}$, PM_{10}, total suspended particulate (TSP), CO_2 and meteorology at two locations adjacent to rail lines. Because our goal is to quantify the exposure to DPM and coal dust, if present, and the EFs from individual trains, we made 10–second measurements so as to capture the air quality impacts from individual passing trains.

2. Experimental

Measurements on train emissions were made at two sites in Washington State (Figure 1). The first site was located in the residential Blue Ridge (BR) neighborhood (47.70°N, 122.40°W), in the City of Seattle, approximately 10 km north of downtown. The instruments and camera were housed on the patio of a residence, which is approximately 25 meters from two active rail lines. The rail lines are immediately adjacent to the shores of Puget Sound and there are no roads in this direction before the shoreline. A video camera was co–located with the instrumentation and could identify train types both day and night at this site. The second site was located in the Columbia River Gorge (CRG), between the towns of Lyle and Dallesport, Washington (45.67°N, 121.20°W). Here the instruments were housed in a small tent, which was located on a small rock outcropping, approximately 10 meters above and 30 meters north of the rail line. The camera was separate from the other instrumentation and located about 25 meters from the tracks but at a 40–degree angle to it. The lower ambient light levels, the camera angle and distance made it impossible to identify train types at night at this site. The rail lines are along the Columbia River and there were no other roads in that direction before the river. This site was about 100 meters south of Washington Route 14, which is a lightly traveled state highway. The instruments were identical at both the Seattle and CRG sites, except that BC was measured only at the CRG site. The data capture at the Seattle site was greater than 95%, whereas instrument and computer failures at the CRG site gave us lower data capture. At both sites the rail line was essentially flat, with a maximum grade of 1 meter per km in the adjacent few km in either direction.

A third site was used only for comparisons of two different DustTrak instruments with a tapered element oscillating micro-balance (TEOM). This site is one of the regular Seattle air quality monitoring stations operated by the Puget Sound Clean Air Agency (pscleanair.org). The site is located along the Duwamish Waterway in the industrial Duwamish Valley, which has a heavy concentration of diesel trucks, trains and ships, due to its proximity to a major port facility. At this site, a Rupprecht and Patashnick TEOM model 1400AB with Filter Dynamics Measurement Systems (FDMS) 8500 is operated as Federal Equivalent Method (Ray and Vaughn, 2009). An Ecotech M9003 nephelometer was also operated to measure scattering coefficients. The scattering coefficients are converted into $PM_{2.5}$ (μg/m^3) based on a 3–year comparison with a Federal Reference Method. The two DustTrak instruments (described below) were operated at this site in the same way as was done at the train sampling sites. This site was chosen for the DustTrak comparison, as it regularly reports the highest concentrations of $PM_{2.5}$ in the Seattle area, due to the high number of diesel vehicles in the area (Keill and Maykut, 2003).

We measured size–segregated PM measurements using a DustTrak DRX Aerosol Monitor (Model #8533, TSI, Inc., Shoreview, MN). This instrument reports PM mass concentrations in 4 size fractions: PM_1, $PM_{2.5}$, PM_{10} and TSP. Because the DustTrak uses aerosol scattering as the basis for its measurements, the measurements are not identical to a mass–based measurement (Wang et al., 2009). The instrument comes calibrated against standard Arizona road dust (ISO 12103–1), but this will not accurately reflect the scattering efficiency for many aerosol types. This may be especially true for diesel given the small size of particles (Park et al., 2011). Instead, accurate measurements using the DustTrak require a comparison against a mass–based measurement for the aerosol of interest (Moosmuller et al., 2001). A number of previous EF studies have also used the DustTrak to rapidly measure several size fractions of PM and calculate EFs from individual vehicles (e.g., Park et al., 2011; Dallmann et al., 2012), but usually after calibration of the response factor against a mass–based method (Jamriska et al., 2004; Zhu et al., 2005; Cheng et al., 2006). For our study, we calibrated the DustTrak against a mass–

based TEOM measurement (described above). The inlet for the DustTrak was downward–facing stainless steel tubing (5.0 mm i.d.) at a height of approximately 2 meters above ground level. The flow through this inlet was 3.0 liters per minute. Under these conditions, the flow is laminar and we would expect greater than 90% particle transmissions for particles up to 2.5 µm in diameter at wind speeds below 10 m/s (von der Weiden et al., 2009). At higher wind speeds and for larger particle sizes, the sampling efficiency will be reduced. Data were stored as 10–second averages.

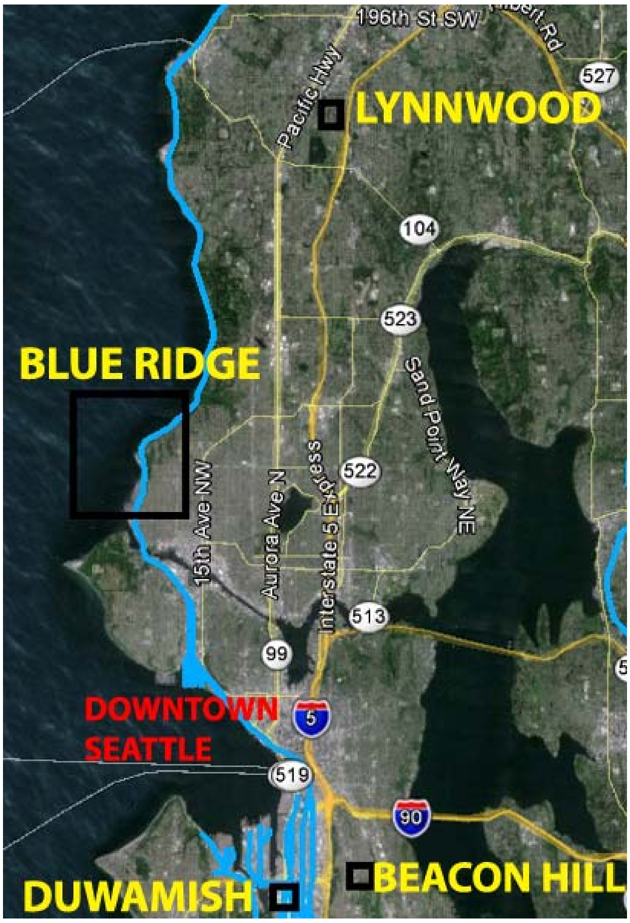

Figure 1. *Map showing air sampling locations in the Seattle area. Major roads and highways are shown by yellow lines and the rail lines are shown in blue. The Duwamish Valley, Lynnwood and Beacon Hill sites are operated by the Puget Sound Clean Air Agency (PSCAA, 2013). At the BR site, PM and CO_2 instrumentation were set up at a residence, approximately 25 meters from the rail lines. The site in the Columbia River Gorge (not shown) is 227 km to the south-southeast of Seattle.*

CO_2 was measured using a Licor–820 (Licor, Inc., Lincoln, NE). Air was pulled through the Licor instrument using a small vacuum pump. The inlet consisted of a 5.0 mm i.d. stainless steel inlet that connected to PFA tubing. The instrument was zeroed by pumping CO_2–free air through it and calibrated with a 395 ppmv standard (Airgas, Inc.). The instrument was calibrated before the Seattle deployment and after the CRG deployment, and the instrument response had drifted by less than 1 ppmv between these times. Data from the DustTrak and the Licor–820 (CO_2, cell temperature and pressure) and the meteorological data were recorded using DAQFactory on a PC. Data were recorded as 10–second averages.

Train types were identified using video taken by a Night Owl camera equipped with infrared night vision (Model CAM–MZ420–425M). The camera was motion activated and controlled using iSpy open source security camera software. At the Seattle site, we were able to classify train types (freight, passenger, etc.) both day and night due to greater ambient light. At the CRG site the camera was able to identify a passing train day or night, but the train type could be identified only in the daytime due to the camera angle, distance and lower ambient light levels.

At the CRG site only, BC measurements were taken using a two–wavelength aethalometer (AE–22, Magee Scientific). BC sampling was performed at 1–minute resolution at 370 nm and 880 nm. Data from the 880 nm infrared absorption signal were used to determine BC loading, as 370 nm is susceptible to absorbance of other organic aerosol from diesel plumes (Wang et al., 2011). The aethalometer measures attenuation (ATN) values (1/m) and determines *BC* concentration (g/m^3) via:

$$BC_0 = ATN/\sigma \tag{1}$$

where, σ is Magee Scientific's calibrated cross–section of $1.4625\text{x}10^4/\lambda$ (at 880 nm, σ=16.6 m^2/g). However, since attenuation diminishes as the BC loading on the filter increases, we apply a correction to the BC concentrations following Kirchstetter and Novakov (2007). Transmission (*Tr*) can be calculated from the attenuation values as:

$$Tr = e^{-ATN/100} \tag{2}$$

The corrected BC loading (ng/m^3) can then be calculated following Kirchstetter and Novakov (2007) as:

$$BC_{corr} = BC_0/(0.88\text{x}Tr+0.12) \tag{3}$$

Both PM_1 and BC EFs are quantified as emissions per kg of diesel fuel burned. These are calculated for each passing train. The EFs for PM_1 and BC are calculated from:

$$\text{EF }(PM_1) = \Delta PM_1/\Delta CO_2\text{x}W_c \tag{4}$$

$$EF\ (BC) = \Delta BC/\Delta CO_2\text{x}W_c \tag{5}$$

where, $\Delta PM_1/\Delta CO_2$ is calculated from the slopes of the regression lines using the 10–second CO_2 and PM_1 data for each passing train. For BC, the ratio $\Delta BC/\Delta CO_2$ is obtained from the one–minute data by subtracting the background concentrations before and after the train passes:

$$\Delta\text{BC}/\Delta\text{CO}_2 = \frac{BC_{Corr,train} - BC_{Corr,baseline}}{[CO_2]_{train} - [CO_2]_{baseline}} \tag{6}$$

CO_2 concentrations are converted to g C/m^3 units using the ideal gas law at 1 atm and 25 °C. W_C is the mass fraction of carbon in diesel fuel (0.87 kg C/kg fuel, Lloyd's Register, 1995; Cooper, 2003), giving overall units on the EF of g PM_1/kg fuel consumed or g BC/kg fuel consumed. Yanowitz et al. (2000) show that more than 95% of the diesel fuel carbon is emitted as CO_2. We chose to use PM_1 in these calculations because this is least likely to be influenced from coal dust or dust from other sources. Using the information presented later in our analysis, one could adjust our EFs for other size fractions.

At the BR and CRG sites, measurements of train emissions and PM were conducted from July 23–August 19, 2013, and August 27–September 2, 2013, respectively. At the Duwamish site, the DustTrak–TEOM comparison was carried out from September 23–October 13, 2013.

3. Results

3.1. Calibration of the DustTrak

Figure 2 shows a scatter plot of the hourly $PM_{2.5}$ concentrations measured by the DustTrak and the TEOM at the Duwamish site. There is an excellent correlation between the TEOM and the DustTrak measurements (R^2=0.83), but the slope is far from 1.0. There was also an excellent agreement between the two DustTrak instruments (R^2=0.99), with a slope of 1.00 and essentially no offset between instruments.

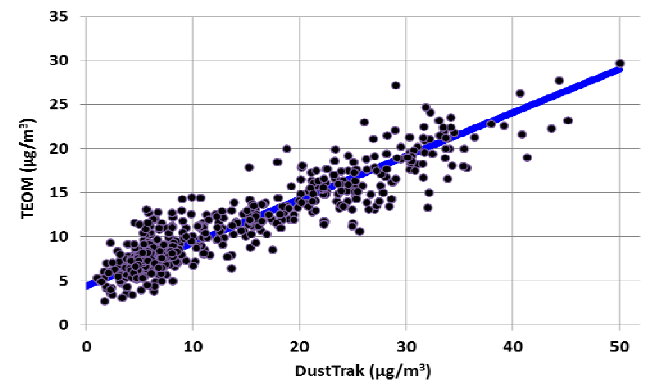

Figure 2. *Comparison of hourly $PM_{2.5}$ concentrations measured by the DustTrak and TEOM-FDMS at the Duwamish site for 9/23/2013 through 10/13/2013. The line shows the RMA regression fit. The regression parameters are m=0.4913, b=4.414 (y=mx+b) and R^2=0.83.*

We determined the regression relationship between the TEOM and DustTrak (serial number 8533131306) using Reduced Major Axis (RMA) regression (Ayers, 2001; Cantrell, 2008):

TEOM $PM_{2.5}$ (μg/m^3)=DustTrak $PM_{2.5}$ (μg/m^3)x0.4913+4.414 (7)

The 95% confidence interval (CI) for the slope and intercept from the RMA regression are 0.47–0.51 and 4.1–4.7, respectively. Our result agrees remarkably well with a similar comparison on DPM by Jamriska et al. (2004), using both a DustTrak and a TEOM, who reported this relationship:

TEOM $PM_{2.5}$ (μg/m^3)=DustTrak $PM_{2.5}$x0.458+4.882 (8)

We also compared the DustTrak with the Ecotech nephelometer at the Duwamish site, to obtain the following relationship:

Ecotech Nephelometer $PM_{2.5}$ (μg/m^3)= DustTrak $PM_{2.5}$ x0.4176+2.926 (9)

The R^2 for the DustTrak–nephelometer regression is 0.98, likely due to the fact that both methods are scattering based. However, the intercept using the nephelometer data is smaller (2.9 μg/m^3 vs. 4.4 μg/m^3) compared to the TEOM, suggesting an uncertainty in the intercept of ±2 μg/m^3. The overall uncertainty in our PM measurements made with the DustTrak is due to uncertainty in the slope (10%) and the intercept (±2 μg/m^3). For the remainder of this paper, we will use the corrected DustTrak data, based on our measured relationship to the TEOM data from the Duwamish site. To maintain consistency with different size bins, we correct all PM concentrations (e.g., PM_1, $PM_{2.5}$, etc.) to the TEOM values using Equation (7).

3.2. Observations of PM and CO_2

Figure 3 shows a time series of PM_1 (μg/m^3) and CO_2 (ppmv) concentrations for a 6–hour period at the Seattle site. We define a “train event” as a single spike or enhancement in PM and CO_2 that is confirmed by the video images. During the period shown in Figure 3 we identified 8 train events. Each train event was confirmed and classified (freight, coal, passenger or other) using the videos. Typical train events last from 1 to 5 minutes, depending on the length of the train, the number of locomotives and meteorology. For each train event, we calculated the regression fit for the following relationships: PM_1–CO_2, PM_1–TSP, $PM_{2.5}$–TSP and PM_{10}–TSP. Figure 4 shows an example of the PM_1/CO_2 relationship for one train event. The slope from the linear correlation ($\Delta PM_1/\Delta CO_2$) is used to derive the DPM EF using Equation (4).

Note that not all trains will be detected by the atmospheric data. For example, if the winds are blowing strongly or are from the wrong direction, our sensors will record only small peaks, or no peaks, in PM_1 and CO_2. These smaller events will generally have a lower PM_1–CO_2 correlation coefficient, so we screened out smaller events with an R^2<0.5 or with ΔCO_2<3 ppmv. This results in 456 train events that passed this QC screen, out of a total of 584 for both sites.

Table 1 shows statistics on the $\Delta PM_1/\Delta CO_2$ slope for the 456 trains we identified at both sites. For the Seattle site, these are separated by train type (freight, coal, passenger and other or unidentified). The distributions are slightly skewed, as shown by the higher means compared to median values. The average slopes range from 0.45 μg/m^3/ppmv for coal trains to 0.59 μg/m^3/ppmv for passenger trains. The difference between passenger and freight trains is statistically significant with greater than 95% confidence. The other differences are not statistically significant. For the CRG site, given the very small number of identifiable coal (3) and passenger (8) trains, we do not report statistics separately for different train types.

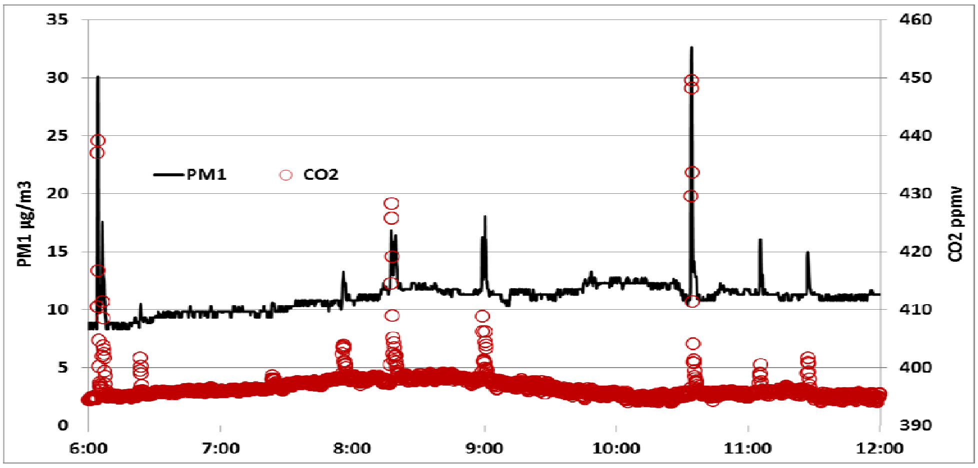

Figure 3. *PM_1 (μg/m^3) and CO_2 (ppmv) data (10-second averages) from the Blue Ridge site for July 25, 2013, between 06:00 and 12:00 local time. During this period, we identified 8 trains from the atmospheric data and confirmed by video images.*

Table 1. *Data on $\Delta PM_1/\Delta CO_2$ slopes ($\mu g/m^3$/ppmv) for different train types at the Blue Ridge location and for all trains at the Columbia River Gorge location. To convert to EFs in g/kg, multiply by 1.81*

	Blue Ridge					CRG
	Freight	Coal	Passenger	Other	All trains	All trains
Count	236	36	93	7	372	84
Average	0.52	0.45	0.59	0.77	0.53	0.51
SD	0.28	0.47	0.35	0.73	0.33	0.36
Median	0.47	0.33	0.53	0.44	0.47	0.45

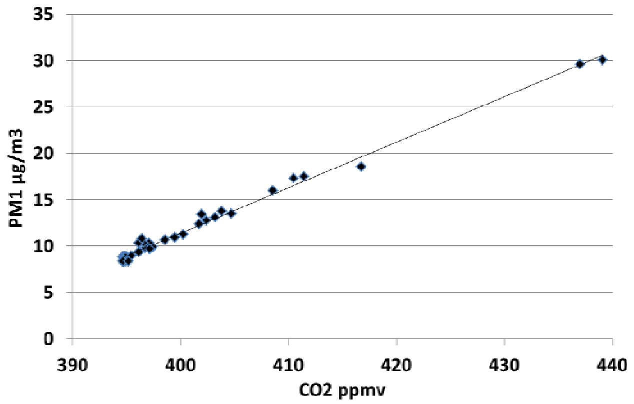

Figure 4. *Linear correlation between PM_1 ($\mu g/m^3$) and CO_2 (ppmv) for the train event starting at 06:03 local time on 7/25/2013 (first event shown in Figure 3). The linear regression parameters are m=0.4888, b=-184.11 (y=mx+b) and R^2=0.989.*

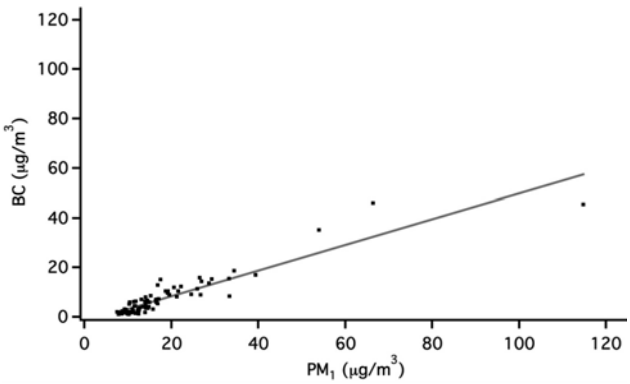

Figure 5. *Correlation of black carbon mass with PM_1 from individual train events. The regression line is y=0.518x-2.009 (R^2=0.85).*

3.3. Emission factors

The average $\Delta PM_1/\Delta CO_2$ slope for all 456 train events was 0.53 $\mu g/m^3$/ppmv, with a 95% confidence interval of ±0.03 $\mu g/m^3$/ppmv. This converts to a PM_1 EF of 0.94 g/kg diesel fuel consumed, with a 95% confidence interval of 0.06 g/kg. Given the uncertainty in the DustTrak calibration factor (Section 3.1), we assign an overall uncertainty of 20% to our mean EF based on the mean PM_1 enhancement. For comparison, an older study by Kean et al. (2000) reports locomotive emission factors using the EPA "NONROAD" model of between 1.8–2.1 g/kg. A 2009 report (U.S. EPA, 2009) projected future emission factors for the fleet averaged, in–use diesel locomotives. The EPA–estimated average EF for 2013 is 1.2 g/kg. A study by Sierra Research (2004) projected a slower reduction in the diesel locomotive EFs, compared to U.S. EPA (2009), and projected a value of 1.5 g/kg for 2013. A study by Galvis et al. (2013) derived EFs for diesel locomotives of 0.4–2.3 g/kg, depending on the assumptions made. Given the uncertainty in our EF, our average value is consistent with the values given for the 2013 time frame.

At the CRG site, the observed BC and PM_1 measurements on 84 trains reveal that on average, 52% of the PM_1 is BC (Figure 5). This is broadly consistent with previous measurements of black carbon in diesel engine particulate emissions in trucks; Hildemann et al. (1991) report 55% black carbon, and Watson et al. (1994) report 45% in diesel engine particulate emissions, in both cases larger than the fraction observed in gasoline emissions. BC to $PM_{2.5}$ ratios at 47–52% have also been reported for diesel train emissions in Atlanta (Galvis et al., 2013). The average BC EF of 0.66 g_{BC}/kg_{fuel} (see the Supporting Material, SM, Figure S1) suggests that rail BC emissions are similar to those reported from heavy–duty diesel trucks, 0.54 g_{BC}/kg_{fuel} (Dallmann et al., 2012). Because only very few coal trains were identified at the CRG site, there was insufficient data to clearly identify coal dust in the BC time–series data.

3.4. Size distributions

The DustTrak measures PM concentrations in four size ranges. For the majority of trains measured, the mass fraction of the TSP concentration was dominated by particles smaller than 1 µm diameter. To compare the size distributions from different train types, we required that R^2>0.5 for the PM_1–TSP correlation for an event to be included in this analysis. This yielded 449 train events from the Seattle site for this analysis, out of 487 possible events. Data on the PM size distributions are given in Table S1 (see the SM). Note that because we use different QC criteria than for the PM_1/CO_2 slope, the number of train events included in this analysis is different from the number shown in Table 1. For these 449 events, the average PM/TSP mass fraction for all trains was found to be 0.86, 0.89 and 0.97 for PM_1, $PM_{2.5}$ and PM_{10}, respectively.

For some trains, there was evidence for larger particles present. Figure 6 shows the measured PM_1, TSP and CO_2 concentrations for a coal train that passed the Seattle site starting at 9:56 local time on August 13, 2013. For this train event, the first peak, between 9:56 and 9:57, shows an excellent correlation between PM_1 and CO_2 (R^2=0.98) and nearly all aerosol mass is due to particles less than 1 µm diameter (PM_1≈TSP). However, there is a second peak in PM_1 at 9:59 without a corresponding CO_2 peak. For this peak, TSP is now significantly larger than PM_1, indicating the presence of larger particles. It is important to note that our inlet likely excludes a significant fraction of larger particles, so our measured TSP concentrations are likely an underestimate. We examined the data to see if there was a statistically significant difference between the PM fractions by train type. Figure 7 shows the average PM size fraction (PM_1/TSP, $PM_{2.5}$/TSP and PM_{10}/TSP) separated by train types for the Seattle site. On average, the PM fractions show that coal trains emit larger particles into the air. These PM_1/TSP fractions, 0.87, 0.77 and 0.88 for freight, coal and passenger trains, respectively, are significantly different at a *P* value of <0.02. Though we did not collect PM samples for chemical analysis, it seems highly likely that the relative contribution of larger particles due to the total PM mass consist of aerosolized coal dust from the uncovered coal trains.

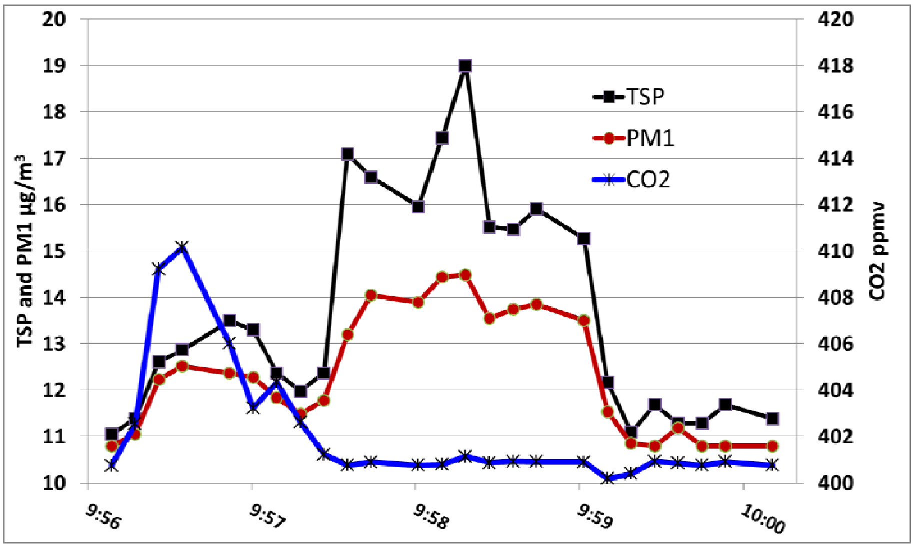

Figure 6. *PM_1 and total suspended particulate (TSP) in µg/m^3 and CO_2 (ppmv) concentrations during passage of a coal train at the Blue Ridge site at 9:56 (PDT) on August 13, 2013.*

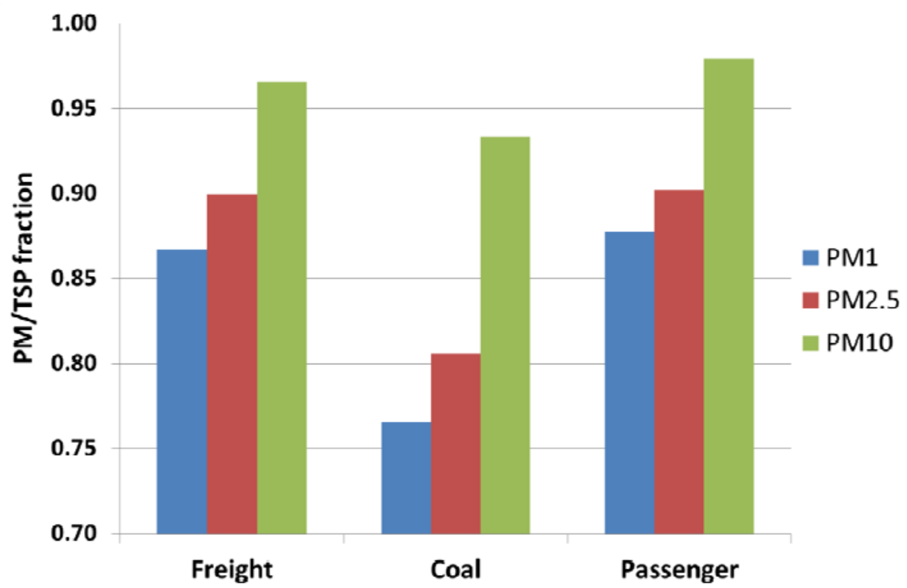

Figure 7. *Fraction of total suspended particulate mass in each size range as measured by the DustTrak at the Blue Ridge site. For this comparison there were 296, 49 and 104 freight, coal and passenger trains, respectively, which had an R^2 for the PM_1-TSP relationship of >0.5. For the PM_1/TSP fraction, the coal trains are significantly lower than freight and passenger trains ($p<0.01$), indicating that a larger fraction of the total PM mass is greater than 1 µm, compared to the other train types.*

3.5. $PM_{2.5}$ exposure due to trains

We measured average $PM_{2.5}$ concentrations at the Seattle and CRG sites of 11.0 and 7.4 µg/m^3, respectively. The lower concentrations at the CRG site reflect the fact that this region is characterized by higher wind speeds and the fact that our site was on a bluff overlooking the river and railroad tracks. At the Seattle site, the instruments were located only a few meters higher than the tracks and the local topography likely creates a greater barrier to dilution of the train emissions. An additional factor is the cold temperatures of Puget Sound (10–12 °C), which cause a stable layer near the surface of Puget Sound (Mass, 2013).

Figure S2 (see the SM) shows the daily average $PM_{2.5}$ concentrations for the BR site during our measurement period, along with three other sites in the region operated by the Puget Sound Clean Air Agency. The site locations are shown in Figure 1. The daily variations at all sites are well correlated and reflect regional $PM_{2.5}$ source/sink relationships. For example, all sites had lower $PM_{2.5}$ on August 1–2, 2013, when cooler, wetter and windier conditions prevailed across the region. During the 4–week measurement period, the average $PM_{2.5}$ concentrations at the Blue Ridge, Lynnwood, Beacon Hill and Duwamish sites were 11.0, 4.3, 6.6 and 11.1 µg/m^3, respectively. The BR and industrial Duwamish Valley sites show similar concentrations, despite the fact that there are no major roads or industries near the BR site. It is possible that marine vessels could have contributed to the enhanced $PM_{2.5}$ observed at the BR site. To evaluate this possibility, we examined $PM_{2.5}$ concentrations at four marine sites along Puget Sound for the same time period using data from the Puget Sound Clean Air

Agency (PSCAA, 2013). For the sites at Bremerton, Oak Harbor, Port Angeles and Port Townsend, the average $PM_{2.5}$ concentrations during this same time period were 5.0, 3.2, 3.5 and 5.0 μg/m^3, respectively. Thus it appears that marine shipping cannot explain the much higher concentrations observed at the BR site. We attribute the additional $PM_{2.5}$ to the presence of the nearby rail line and trains. The difference between the measured PM at the BR site and the average of the four marine sites (11.0–4.2 μg/m^3) represents the additional $PM_{2.5}$ at the BR site due to diesel trains (6.8 μg/m^3).

The enhancement in $PM_{2.5}$ is not only due to the "spikes" that occur as a train passes, but also the residual that accumulates in the local airshed. The topography in the Puget Sound region may also exacerbate the accumulation of $PM_{2.5}$ from trains. This is because the rail line runs approximately north–south (see Figure 1) in the same direction as the prevailing summer winds, and at the foot of a 50–100 meter–high bluff that further limits mixing. These factors contribute to the $PM_{2.5}$ enhancement due to rail traffic.

We can estimate possible impacts of increasing rail traffic on $PM_{2.5}$ concentrations at the BR site. We assume that the $PM_{2.5}$ enhancement due to trains, 6.8 μg/m^3, is linearly related to the total train traffic. Using this assumption, a 50% increase in diesel train traffic would increase the $PM_{2.5}$ due to trains to 10.2 μg/m^3. When added to the regional background (4.2 μg/m^3), this would bring the $PM_{2.5}$ concentrations at the BR site up to approximately 14 μg/m^3, which is higher than the new U.S. National Ambient Air Quality Standard (NAAQS) of 12.0 μg/m^3 (annual average). It is important to note that compliance with the NAAQS is based on three years of data, and thus our one month of observations cannot indicate compliance. But nonetheless, this calculation suggests that a 50% increase in rail traffic will put some residences, such as our BR home, near or over the NAAQS. This assumes that each train contributes uniformly to the PM exposure. While our diesel emission factors for coal and freight train types were statistically indistinguishable, the train length, number of locomotives and fuel consumption may vary for different train types. Thus a more complete estimate of future impacts on air quality from rail traffic should consider these factors.

4. Summary

We measured the PM_1 emission factors for over 450 trains at two sites in Washington State and the resulting $PM_{2.5}$ exposure (μg/m^3). For 84 of these trains, we also measured the black carbon emission factors. Our measurements demonstrate that rail traffic emits substantial quantities of diesel exhaust and that the $PM_{2.5}$ concentrations are significantly enhanced for residents living close to the rail lines. Future growth in rail traffic will increase the $PM_{2.5}$ exposure and, for some homes, may result in concentrations that exceed the U.S. NAAQS. Our results also show that after passage of coal trains there was a statistically significant enhancement in larger particles, compared to other train types. These larger particles most likely consist of aerosolized coal dust. Our study addresses exposure to residents who live close to the rail lines. Future studies should examine several questions that were not addressed by our study:

(i) How does the concentration of $PM_{2.5}$ vary with proximity to the rail lines?

(ii) What are the total emissions from rail traffic and what is the net contribution to $PM_{2.5}$ across the broader Seattle metropolitan area?

(iii) What are the health effects associated with $PM_{2.5}$ and coal dust from rail traffic?

Acknowledgements

The authors would like to acknowledge the financial contributions of over 270 individuals via microryza.com and a donation by the Sierra Club. These donations were made as gifts to the University of Washington. The authors are also grateful to several anonymous individuals who let us use their home and/or property to conduct these measurements. BA and JLF would like to thank Tony Hansen and Magee Scientific for the generous loan of the AE–22 instrument used to conduct these experiments.

Supporting Material Available

BC emission factors calculated for each train event (Figure S1), Daily average $PM_{2.5}$ for four sites in the Seattle metropolitan region (Figure S2), Fraction of total suspended particulate (TSP) mass in the PM_1 and $PM_{2.5}$ size ranges, for each train type (Table S1). This information is available free of charge via the Internet at http://www.atmospolres.com.

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SUPPORTING MATERIAL

Diesel particulate matter emission factors and air quality implications from in-service rail in Washington State, USA

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Content

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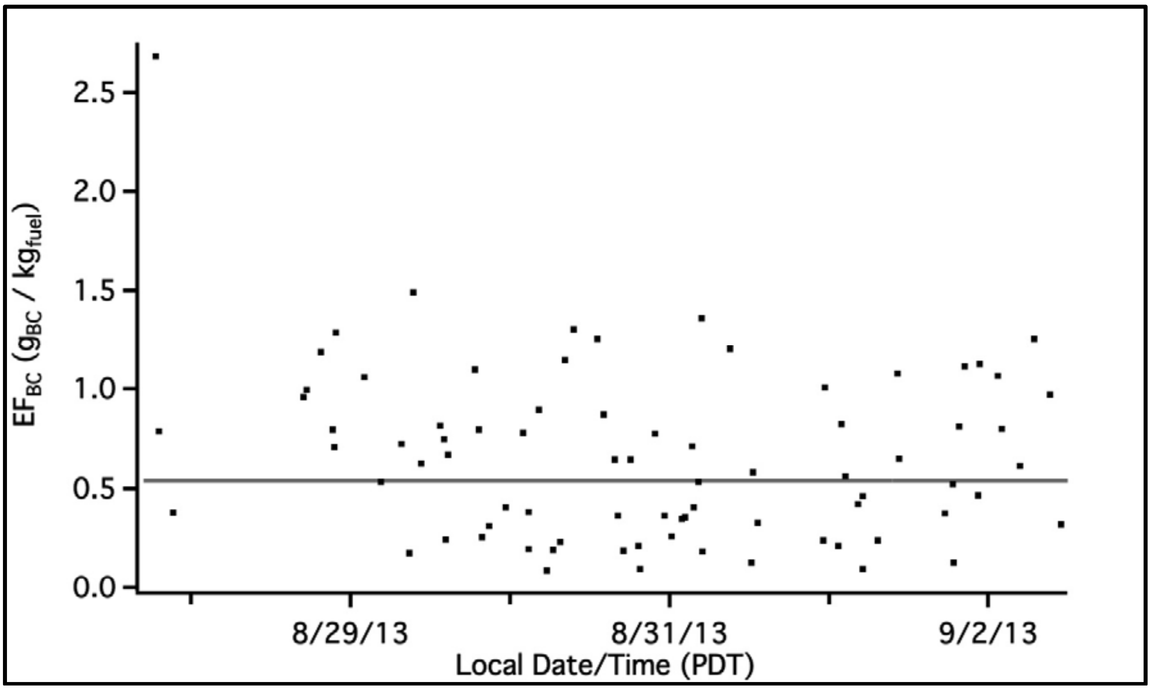

Figure S1. BC emission factors calculated for each train event (dots) show substantial variation, but the mean value of 0.66 g/kg is similar to a recent study on diesel truck engines (0.54 g/kg, horizontal line, Dallmann, et al., 2012).

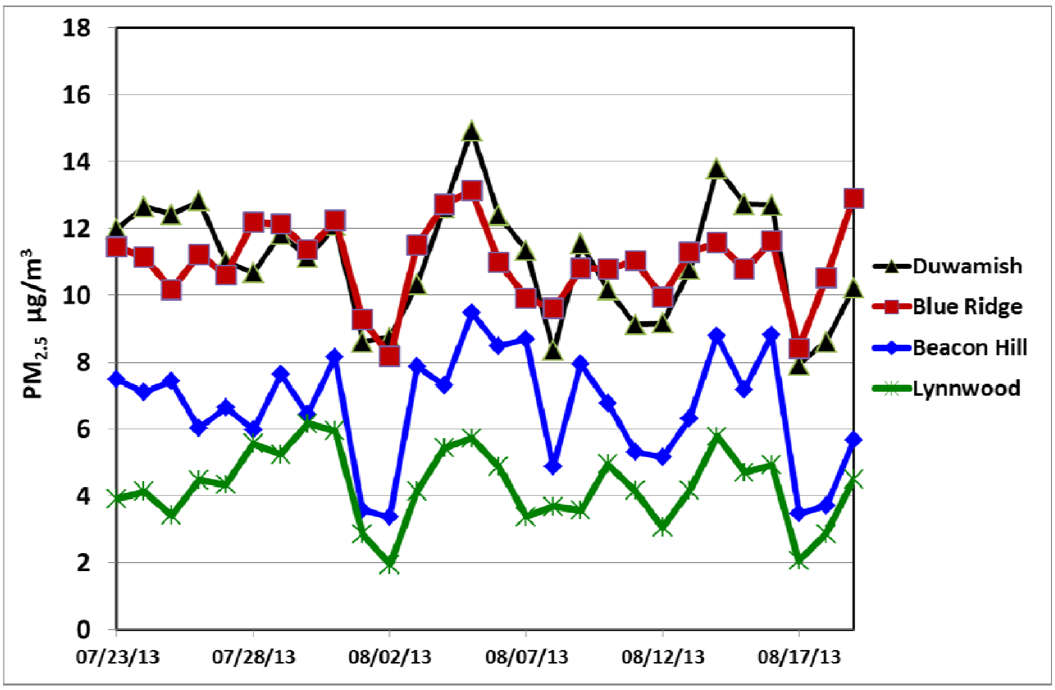

Figure S2. Daily average $PM_{2.5}$ for four sites in the Seattle metropolitan region (see Figure 1 for locations). During this time period the average concentrations for the Duwamish, Blue Ridge, Beacon Hill and Lynnwood sites were 11.1, 11.0, 6.6 and 4.3 $\mu g/m^3$, respectively.

Table S1. Fraction of total suspended particulate (TSP) mass in the PM_1 and $PM_{2.5}$ size ranges, for each train type. Data is for the Blue Ridge site only

	Freight	Coal	Passenger
Count	296	49	104
PM1/TSP Average	0.87	0.77	0.88
PM1/TSP Standard Deviation	0.14	0.19	0.12
PM2.5/TSP Average	0.90	0.81	0.90
PM2.5/TSP Standard Deviation	0.13	0.17	0.11