

## RESEARCH ARTICLE

# A large source of dust missing in Particulate Matter emission inventories? Wind erosion of post-fire landscapes

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Wind erosion of soils burned by wildfire contributes substantial particulate matter (PM) in the form of dust to the atmosphere, but the magnitude of this dust source is largely unknown. It is important to accurately quantify dust emissions because they can impact human health, degrade visibility, exacerbate dust-on-snow issues (including snowmelt timing, snow chemistry, and avalanche danger), and affect ecological and biogeochemical cycles, precipitation regimes, and the Earth's radiation budget. We used a novel modeling approach in which local-scale winds were used to drive a high-resolution dust emission model parameterized for burned soils to provide a first estimate of post-fire PM emissions. The dust emission model was parameterized with dust flux measurements from a 2010 fire scar. Here we present a case study to demonstrate the ability of the modeling framework to capture the onset and dynamics of a post-fire dust event and then use the modeling framework to estimate PM emissions from burn scars left by wildfires in U.S. western sagebrush landscapes during 2012. Modeled emissions from 1.2 million ha of burned soil totaled 32.1 Tg (11.7–352 Tg) of dust as PM<sub>10</sub> and 12.8 Tg (4.68–141 Tg) as PM<sub>2.5</sub>. Despite the relatively large uncertainties in these estimates and a number of underlying assumptions, these first estimates of annual post-fire dust emissions suggest that post-fire PM emissions could substantially increase current annual PM estimates in the U.S. National Emissions Inventory during high fire activity years. Given the potential for post-fire scars to be a large source of PM, further on-site PM flux measurements are needed to improve emission parameterizations and constrain these first estimates.

**Keywords:** wildfire; dust; PM10

## 1. Introduction

Wildfire smoke is the largest source of primary PM<sub>2.5</sub> (particulate matter with aerodynamic diameter less than 2.5 μm) and the third largest source of PM<sub>10</sub> (PM with aerodynamic diameter less than 10 μm) in the United States (EPA, 2011). Air quality degradation from wildfire smoke is a widespread issue in the western U.S. (Jaffe et al., 2008). This work investigates wind erosion of burned soils as an additional fire-related source of atmospheric PM. In contrast to wildfire smoke, post-fire PM (mineral dust and ash, hereafter referred to as dust) is not as conspicuous an issue since wind erosion events are highly intermittent in time and space and the sources tend to be in remote areas. Yet, there is evidence (Miller et al., 2012; Sankey et al., 2009; Wagenbrenner et al., 2013; supplemental material) that

post-fire landscapes of the U.S. Great Basin (delineated by the gray-colored area in **Figure 1**) are major contributors to atmospheric dust.

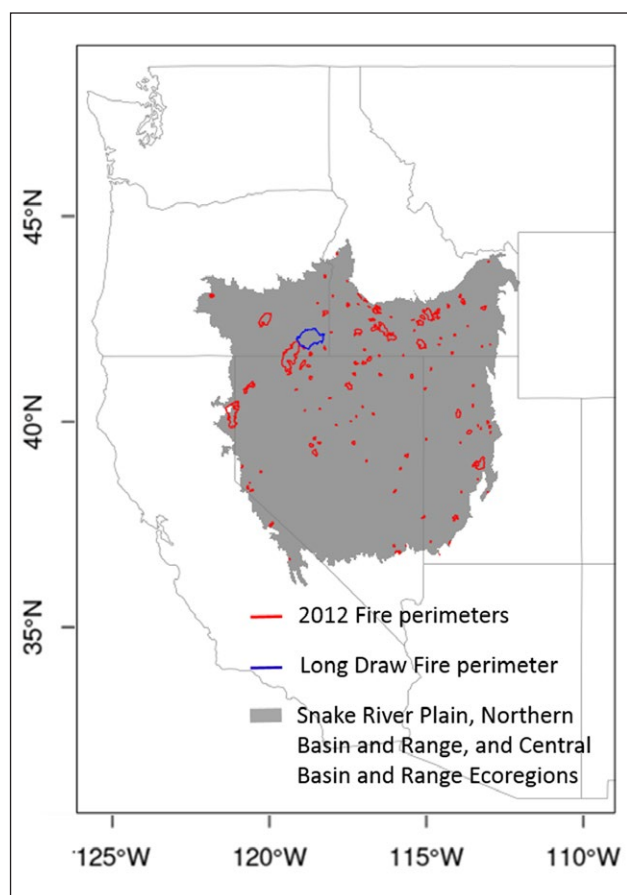
The intense heat of a wildfire can penetrate surface soils increasing erodibility by destroying naturally occurring soil crusts (Ford and Johnson, 2006), increasing soil water repellency (Ravi et al., 2007), and decreasing aggregate stability (Varela et al., 2010). Fires in the Great Basin typically burn quickly and intensely, often consuming all vegetation due to the arid conditions and flammability of the fuels. This leaves dry, loose, bare soil and ash exposed to high winds, also characteristic of this region (Jewell and Nicoll, 2011). Post-fire dust sources are among the strongest atmospheric dust sources reported (Wagenbrenner et al., 2013) and may contribute substantially to the global dust budget.

PM source identification and quantification is important for accurate characterization of atmospheric PM. Atmospheric PM impacts human health (Dockery and Pope, 1994), visibility (Hyslop, 2009), snowpack dynamics including snowmelt (Skiles et al., 2012), snow chemistry (Rhodes et al., 2010), and avalanche danger (Summit County, 2014), ecological and biogeochemical cycles (Field

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**Figure 1: Areas burned by wildfires in the Snake River Plain, Northern Basin and Range, and Central Basin and Range Level III Ecoregions during 2012.** The area burned by the Long Draw Fire is outlined in blue. DOI: <https://doi.org/10.1525/elementa.185.f1>

et al., 2010), precipitation regimes (Sassen et al., 2003), and the Earth's radiation budget (Tegen et al., 1996). But quantification of PM emissions is difficult, particularly for dust, due to the spatial, temporal, and compositional variability of the sources (e.g., Ginoux et al., 2012).

The lack of information regarding dust sources, particularly small-scale or quasi-permanent sources, has hampered understanding and modeling of the global dust cycle (e.g., Ginoux et al., 2012). The best characterized dust sources are topographic depressions containing deep alluvial sediment deposits in arid regions (Prospero et al., 2002). Small-scale and transient sources have been recognized as significant contributors to the global dust budget, but difficulties in identifying emission 'hot spots' have limited their investigation and inclusion in emission inventory estimates (Zender et al., 2003b; Okin et al., 2011). Post-fire sources pose an especially difficult case since they are highly transient, with emissions occurring intermittently for maybe just a year post-fire (Wagenbrenner et al., 2013), and different locations are burned each year.

Additionally, there are potentially specific concerns for post-fire dust. Post-fire dust likely exhibits different optical, chemical, and mechanical properties than typical mineral dust due to the composition of the post-fire material (a mix of ash and soil particles). Little is known regarding

how size distributions and compositional makeup may differ between dust from post-fire sources and mineral dust originating from agricultural or desert sources. Post-fire field measurements suggest that post-fire dust likely contains a higher fraction of  $PM_{2.5}$  than typical mineral dust (Wagenbrenner et al., 2013), likely due to the presence of ash in the surface soil.

In this work we focus on the U.S. Great Basin, where a large number of wildfires occur each year and the landscapes are susceptible to post-fire erosion (Miller et al., 2012; Sankey et al., 2009; Wagenbrenner et al., 2013). Furthermore, dust from the Great Basin has substantial influences on water resources in the U.S. Intermountain West due to dust-snow-albedo effects on snowmelt (Deems et al., 2013; Skiles et al., 2012). The spread of invasive species has increased fuel loadings across the U.S. Great Basin, led to more frequent and severe fires (Balch et al., 2013), and presumably increased post-fire erodibility and PM emissions. Post-fire emissions may provide a positive feedback on the invasive species-altered fire regime loop in this region as the dust emitted from burned areas also contains stores of native plant seedbanks (e.g., Chambers and MacMahon, 1994), which further reduces competition for invasive species.

In this work, we present a new PM emission model based on measurements from Wagenbrenner et al. (2013). To our knowledge, Wagenbrenner et al. (2013) is the only study to-date to report PM vertical flux measurements from a fire scar. The fire scar in Wagenbrenner et al. (2013) was burned by the 2010 Jefferson Fire northwest of Idaho Falls, ID (northeast portion of the area delineated in gray in **Figure 1**). The fire, driven by high southwesterly winds, burned over 100,000 acres of grass and sagebrush in just days. The fire consumed essentially all vegetation and left behind dry, bare soil that was highly erodible. Dust emissions were visible immediately after the fire and elevated emissions were frequently measured until nearly one year post-fire, when vegetation began to re-establish. Additional details can be found in Wagenbrenner et al. (2013).

The objectives of this work are to (1) demonstrate the ability of a new modeling framework to simulate emission and transport of PM during a large post-fire dust event and (2) use the new post-fire PM emission model to provide a first estimate of the contribution of western U.S. fire scars to atmospheric PM. We first provide a case study to demonstrate the behavior of a post-fire wind erosion event and the ability of the new modeling approach to capture this behavior. Then we present a broader emissions modeling study to provide the first estimate of the annual contribution of PM emissions from post-fire dust sources.

## 2. Methods

### 2.1. Models

WindNinja (Forthofer et al., 2014) was used to model the local wind field and PM emissions from the fire scars. WindNinja is a diagnostic microscale wind model that accounts for local mechanical and thermal effects of the underlying terrain on the flow field. Wagenbrenner et al. (2016) found that WindNinja downscaling improved surface wind forecasts in complex terrain.

WindNinja uses a semi-empirical PM emission model (Draxler et al., 2001) parameterized for burned soil. Vertical flux of  $PM_{10}$  is calculated in WindNinja as:

$$F_v = \frac{K\rho}{g} u_* \left( u_* - u_{*t}^2 \right) \quad (1)$$

where  $K$  is the  $PM_{10}$  release factor,  $\rho$  is air density,  $u_*$  is friction velocity, and  $u_{*t}$  is threshold friction velocity.

The  $PM_{10}$  release rate and threshold friction velocity were developed from post-fire field measurements following a 2010 wildfire in the Snake River Plain ecoregion (Wagenbrenner et al., 2013). The  $PM_{10}$  release rate and threshold friction velocity were  $0.0007 \text{ m}^{-1}$  and  $0.22 \text{ ms}^{-1}$  (Table 1). We assumed no re-vegetation and dry, uniform soil conditions representative of the burn scar in Wagenbrenner et al. (2013). Emissions outside of the burn perimeters were assumed to be negligible. The emission model does not account for soil moisture or texture. Given the severe impact fire has on the surface soil and vegetation and the lack of other post-fire PM measurements available, we believe this is a reasonable approximation for an initial attempt to quantify post-fire PM emissions.

Friction velocity is calculated in WindNinja based on a log profile normal to the ground.

$$u_* = \frac{U \kappa}{\ln\left(\frac{z-d}{z_0}\right)} \quad (2)$$

$U(z)$  is the wind speed at height  $z$  above the ground,  $\kappa$  is the von Karman constant,  $d$  is the zero-plane displacement, and  $z_0$  is the roughness parameter. First, the gridded winds are computed in WindNinja.  $U(z)$  is then computed in each cell of a near-surface layer of the computational mesh using a rotated coordinate system where the  $z$ -axis is normal to the ground. Finally  $u_*$  is calculated in each cell of the near-surface layer.  $d$  is set to 0.0 m,  $\kappa$  is set to 0.4, and  $z_0$  is set to 0.01 m, representative of smooth bare soil (Table 1).

The transport and dispersion for the case study were simulated using the AIRPACT-3 regional air quality modeling system for the Pacific Northwest (Chen et al., 2008; Chung et al., 2012). AIRPACT-3 uses the Weather Research and Forecasting (WRF) model (Skamarock et al., 2008) for meteorology and the Community Multiscale Air Quality (CMAQ) model (Byun and Schere, 2006) for chemistry and transport. CMAQ simulates the transport and removal from the atmosphere of PM by wet and dry deposition according to meteorology simulated by WRF. The default AIRPACT-3

modeling system includes all major known sources of PM in the region but, similar to other regional air quality models, does not include post-fire PM emissions, which were added for this work. Based on the measurements of Wagenbrenner et al. (2013), 60% of post-fire  $PM_{10}$  emissions was assigned to coarse-mode PM (PMC or particulate matter with diameters between 2.5 and 10  $\mu\text{m}$ ) and the remainder was assigned to  $PM_{2.5}$ . Post-fire  $PM_{10}$  is calculated as the difference in modeled  $PM_{10}$  concentrations from CMAQ simulations with and without post-fire dust emissions.

## 2.2. A case study

The vast expanse of the Great Basin is sparsely populated, which means fewer people to observe or be immediately affected by post-fire dust events. One of the best documented (Video S1) cases of a population-impacting event occurred on August 5, 2012 when a large dust plume originating from the Long Draw fire scar (Figure 1) in southeast Oregon caused an exceedance of the  $PM_{10}$  National Ambient Air Quality Standard (NAAQS) 150 km downwind in Boise, Idaho. This case study was used to investigate the feasibility of simulating the onset and evolution of a post-fire dust event with a regional air quality model linked with a high-resolution dust emission model.

HYSPLIT backward trajectories originally suggested the Long Draw fire scar as a potential source of the dust plume (e.g., Figure 2). Inspection of NEXRAD base reflectivity in the area at the time of the event further indicated the dust was likely emitted from the burn scar as high winds associated with near-by thunderstorms passed over the area and moved toward Boise, ID (supplemental material).

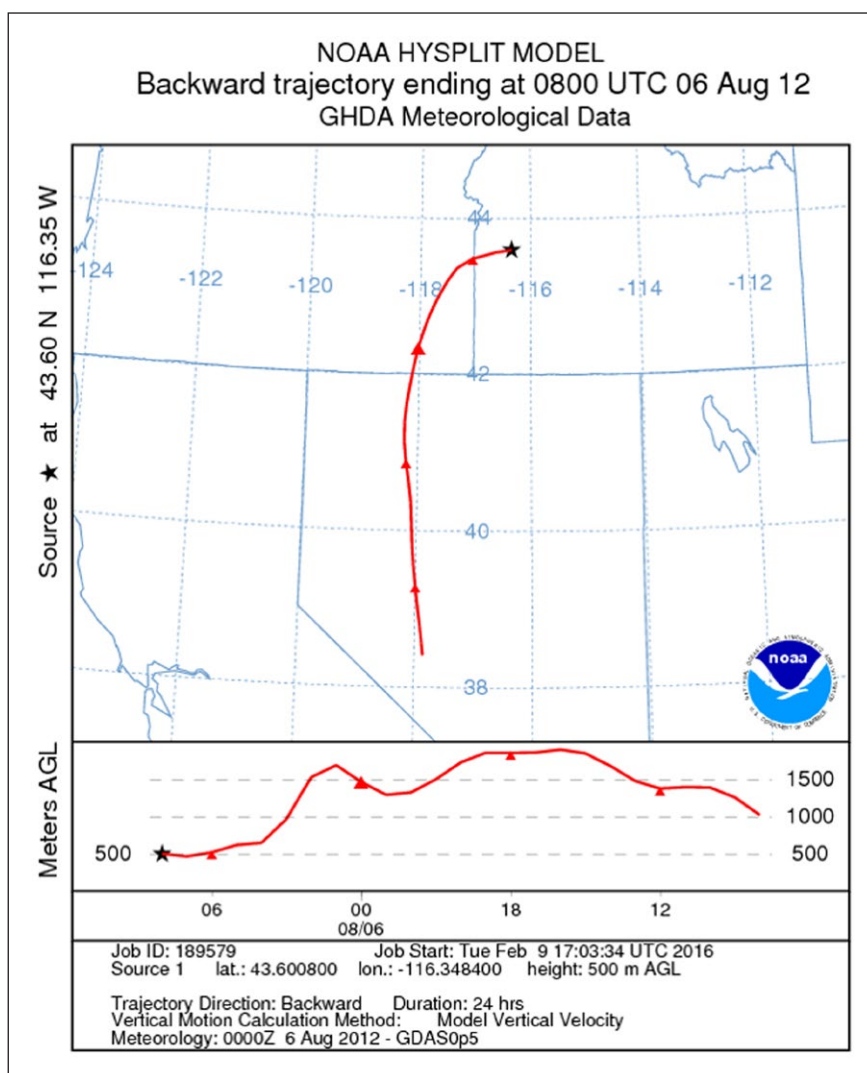
We simulated hourly PM emissions and transport for 24 hours beginning at midnight LT on August 4. Hourly surface winds from a nearby Remote Automated Weather Station (Grassy Mountain RAWS; www.raaws.dri.edu) were used to drive the emission model. The fire perimeter was extracted from the Monitoring Trends in Burn Severity (MTBS84) dataset (Eidenshank et al., 2007). We compared modeled PM concentrations to observed PM data from Boise and Nampa provided by the Idaho Department of Environmental Quality (<http://airquality.deq.idaho.gov>).

## 2.3. Annual emissions estimate

Burn perimeters from wildfires that occurred during 2012 in the Snake River Plain, Northern Basin and Range, and Central Basin and Range Level III Ecoregions (Omernik and Griffith, 2014) were investigated in this study (Figure 1). Fire perimeters and start dates were extracted from the MTBS84 dataset. Only wildfires larger than 1000 acres were included in the analysis. There were 120 burn perimeters covering 1.2 million ha. The year 2012 was chosen because at the time that this work was initiated it was the most recent complete archive of annual fire perimeters and because it included the Long Draw Fire, which we had already chosen to investigate as a case study. This was a particularly active fire year in the U.S. Great Basin (1.2 million ha burned in 2012 vs. 319,651 ha in 2011 and 255,504 ha in 2010) and thus, estimates from this work are representative of a high fire activity year.

**Table 1:** Parameters used in the PM emission model. DOI: <https://doi.org/10.1525/elementa.185.t1>

Parameter	Symbol	Value	Units
von Karman constant	$\kappa$	0.4	–
Zero-plane displacement	$d$	0.0	m
Roughness parameter	$z_0$	0.01	m
$PM_{10}$ release rate	$K$	0.0007	$\text{m}^{-1}$
Threshold friction velocity	$u_{*t}$	0.22	$\text{m s}^{-1}$



**Figure 2: Hysplit backward trajectory for the August 5 dust event observed in Boise, Idaho.** The modeled end time is adjusted (offset of 6 hours based on the WRF-CMAQ results) to account for underestimation of wind speed in the meteorological model. DOI: <https://doi.org/10.1525/elementa.185.f2>

Archived 12-km North American Model (NAM) meteorology was used as input to WindNinja for the emissions estimate work. NAM meteorology was chosen because of its relatively high spatial resolution for regional-scale meteorological modeling and because it was readily obtainable from the National Centers for Environmental Prediction for our temporal and spatial extents. Temporal resolution, determined by NAM forecast output frequency, was one hour. The horizontal grid resolution of the PM emission model was on the order of 100 m but varied according to the domain extent for each fire.

We modeled hourly emissions for each burn scar in **Figure 1** for one year post-fire. The hourly simulations started 24 hours after the fire start date and continued through December 31, 2012, or until the first snowfall, whichever was earlier. Since the grass and sagebrush fuels in the ecoregions investigated in this study typically burn very quickly and accurate temporal fire perimeter information is not available, we assumed the final burn perimeter existed the day following the fire start date. Periods of

precipitation and 24 hours after each precipitation event were omitted from the analysis period.

### 3. Characterization and quantification of post-fire dust

#### 3.1. A case study: 5–6 August, 2012 dust event in Boise, Idaho

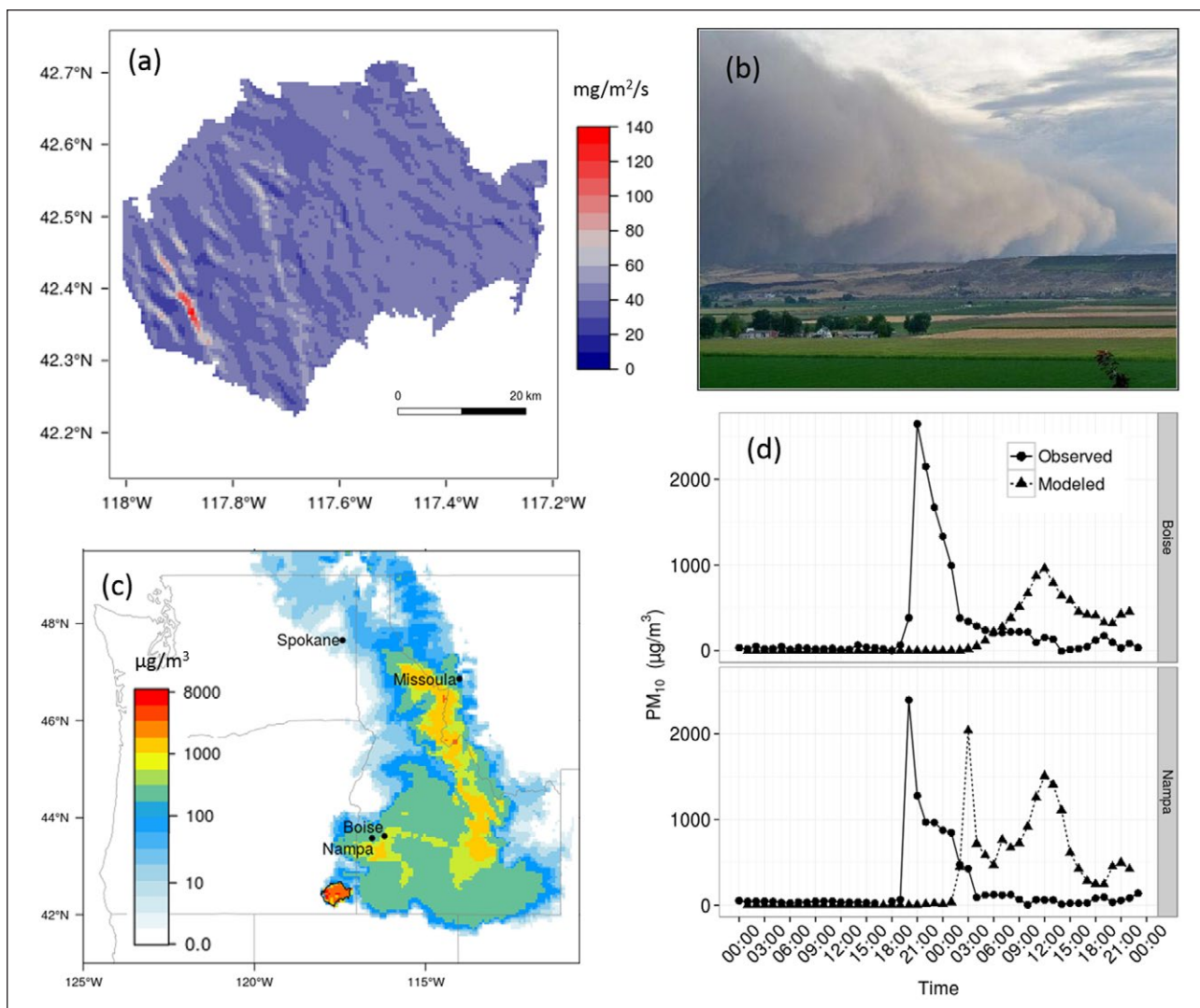
The modeled emissions at the peak of the wind event are shown **Figure 3a**. The modeled emission map clearly shows emission “hot spots” within the burn perimeter due to the mechanical effects of the terrain on the local flow field. The hot spots correspond to areas of higher friction velocities (e.g., due to wind speed-up over a ridge). The modeled emissions varied from 0 to  $140 \text{ mg m}^{-2} \text{ s}^{-1}$  over the domain. The highest predicted emissions were on a ridge oriented from northwest to southeast in the southwest portion of the domain. Although we do not have on-site emission measurements to corroborate the modeled emission pattern, the idea of emission hot spots related to terrain modification of the local wind field has been documented in other studies (e.g., Goosens and Offer, 1997).

**Figure 4b** shows the dust plume approaching Boise, ID around 1800 LT on 5 August. Transport and dispersion modeling captured the general plume trajectory (**Figure 3c**). The modeled transport suggested that PM was lofted high into the atmosphere not far downwind from the burn scar. **Figure 3c** shows high surface concentrations over the burn scar, but lower surface concentrations immediately downwind. The modeled plume eventually covered most of Idaho and extended into parts of Washington, Montana, and Canada (**Figure 3c**). The high PM concentrations along the Idaho-Montana border are due to interception of the dust plume by the high-elevation Bitterroot Mountains (**Figure 3c**).

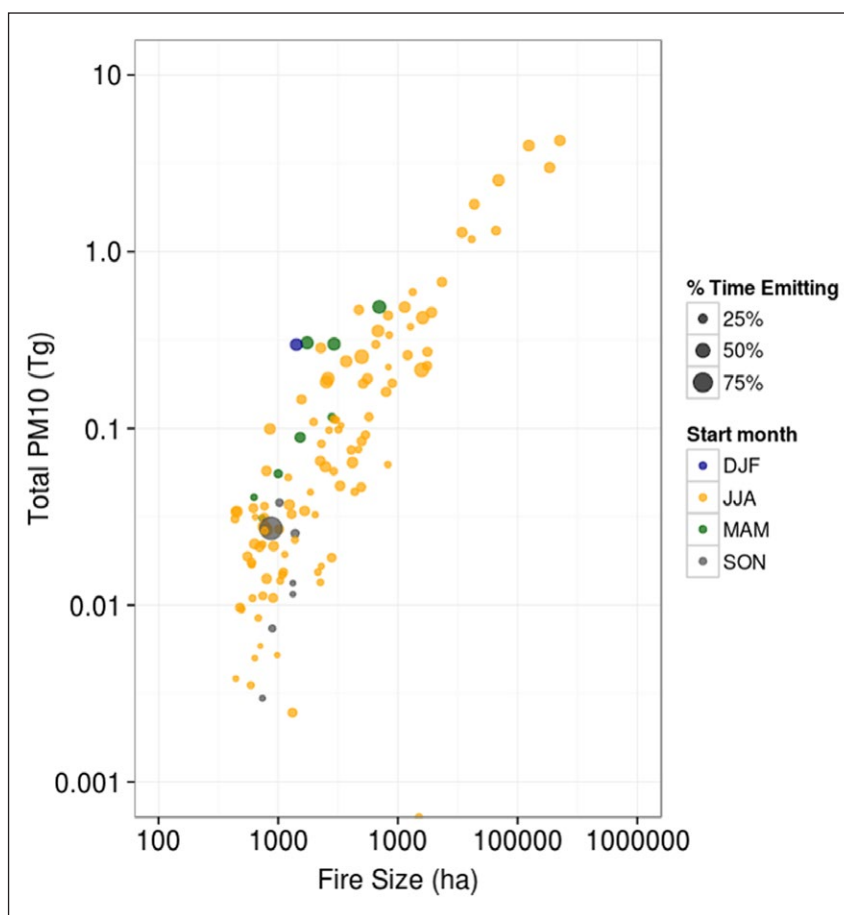
Observed peak concentrations were  $2650 \mu\text{g m}^{-3}$  in Boise at 21:00 (LT) and  $2397 \mu\text{g m}^{-3}$  in Nampa at 20:00 (LT) on 5 August (**Figure 3d**). Elevated concentrations persisted at both locations for about 8 hours. Modeled surface concentrations in Boise and Nampa were lower than, but on the order of the measured concentrations (**Figure 3d**).

Modeled peak concentrations were  $962 \mu\text{g m}^{-3}$  in Boise at 12:00 on 6 August and  $2040 \mu\text{g m}^{-3}$  in Nampa at 03:00 (LT) on 6 August. This indicates that the modeled PM emission rate for this event was reasonable.

The timing of the modeled peak concentrations was delayed due to an issue with the modeled meteorology used to drive the transport model. The WRF meteorological model underestimated the high wind speeds associated with the thunderstorm outflows which drove the emission and local transport of PM (discussed in Section 2.2). This underestimation of wind speed increased the residence time of the modeled plume and is at least partially responsible for the high modeled PM concentrations in Boise and Nampa following the concentration peaks. The 24-hr average concentration was still underestimated for Boise ( $483 \mu\text{g m}^{-3}$  observed vs.  $279 \mu\text{g m}^{-3}$  modeled), but was over-predicted for Nampa ( $385 \mu\text{g m}^{-3}$  observed vs.  $561 \mu\text{g m}^{-3}$  modeled) during 17:00 5 August to 17:00 6 August.



**Figure 3: Modeled emissions from the Long Draw Fire. (a)** A haboob originating from the Long Draw Fire and approaching Boise, Idaho on August 5, 2012. **(b)** Photo credit: ktvb.com. Modeled PM<sub>10</sub> concentrations during the haboob event on August 5, 2012 **(c)** Modeled and observed surface concentrations in Boise and Nampa (~ 34 km west of Boise), Idaho. **(d)** Observed data are provided by the Idaho Department of Environmental Quality (<http://airquality.deq.idaho.gov>). DOI: <https://doi.org/10.1525/elementa.185.f3>



**Figure 4: Total PM<sub>10</sub> emitted from areas burned by wildfire during 2012.** The size of the circle indicates the percent of time (from the day after the fire to the end of the year) during which at least one pixel was emitting PM<sub>10</sub>. The color of the circle indicates the start month of the fire. DOI: <https://doi.org/10.1525/elementa.185.f4>

The general behavior of the post-fire dust event was captured in this modeling effort. WindNinja simulated reasonable PM emission levels when driven by local observed winds, based on comparisons with observed surface concentrations in Boise and Nampa. The dust transport modeling showed relatively good performance for PM surface concentrations although timing was off due to issues with the modeled meteorology. The modeled PM accumulated surface loadings were 32% lower and 53% higher than the observed surface loadings at Boise and Nampa, respectively, and were 64 and 145 times higher than the modeled PM loadings in the absence of PM emissions from the Long Draw fire scar. This case study demonstrates the potential for widespread impacts of post-fire wind erosion as well as the feasibility of accounting for post-fire dust within a regional air quality modeling framework.

### 3.2. An estimate of annual post-fire PM emissions

In this section we describe the modeled annual emissions from the fire scars in **Figure 1**. The modeling approach is described in Section 2.3. Total emitted PM<sub>10</sub> was 32.1 Tg (11.7–352 Tg) including 40% or 12.8 Tg (4.68–141 Tg) of which was estimated as PM<sub>2.5</sub>. The uncertainty ranges in parentheses are based on emission model uncertainties described in Section 3.3. These estimates suggest post-fire

landscapes could be the largest source of PM<sub>10</sub> and PM<sub>2.5</sub> in the continental U.S. and increase annual total PM<sub>10</sub> and PM<sub>2.5</sub> emissions by 171% (62%–1872%) and 231% (84%–2545%), respectively (**Table 2**), during a high fire activity year. PM<sub>10</sub> emissions among the burned areas ranged from 0.003 to 21.0 kg m<sup>-2</sup>, with a mean of 3.49 kg m<sup>-2</sup> and a median of 2.83 kg m<sup>-2</sup>.

Total emitted PM<sub>10</sub> scaled with the size of the burned area (**Figure 4**). The largest fires occurred during JJA, which is the driest season in the Great Basin region; as such, the main window for post-fire dust emissions extended from mid-summer through late fall, depending on the timing of winter precipitation. However, previous field studies have shown that, in some cases, post-fire dust sources can also contribute PM after snowmelt the following spring (Wagenbrenner et al., 2013) and in some cases more than one year post-fire, depending on re-establishment of vegetation in the burned area (Miller et al., 2012). Our simulations did not take this post-winter emission period into account.

The percent of time emitting (PTE) was calculated for each fire as the number of hours from the start date of the fire to the end of the year divided by the number of hours during which at least one model grid cell was emitting PM<sub>10</sub>. The PTE ranged from 0.06 to 91%, with a median of 23%. The strongest source areas had PTEs around 30%,

**Table 2:** Annual PM emission estimates<sup>a</sup>. DOI: <https://doi.org/10.1525/elementa.185.t2>

Region	PM <sub>10</sub> Tg	PM <sub>2.5</sub> Tg
Global <sup>b</sup>		
Mineral dust	1220–1540	308
CONUS <sup>c</sup>		
<b>All sources</b>	<b>18.8</b>	<b>5.54</b>
Mineral dust (non-agriculture)	9.95	1.15
Agriculture	4.08	0.81
Wildfire smoke	2.29	1.93
Other	2.44	1.65
Basin ecoregions		
Wildfire smoke <sup>d</sup>	0.12	0.10
<b>Post-fire dust<sup>e</sup></b>	<b>32.1</b>	<b>12.8</b>
	<b>(11.7–352)</b>	<b>(4.68–141)</b>

<sup>a</sup>Basin ecoregions includes the Snake River Plain, Northern Basin and Range, and Central Basin and Range Level III Ecoregions defined in Omernik and Griffith (2014).

<sup>b</sup>Ginoux et al., 2012 and Zender et al., 2003a;

<sup>c</sup>2011 National Emission Inventory (EPA, 2011);

<sup>d</sup>Urbanski et al. 2011;

<sup>e</sup>This study.

which suggests that the largest contributions to total PM emissions were from occasional high-wind events, rather than persistent, moderate-strength winds (**Figure 4**).

### 3.3. Emission estimate uncertainties

Primary sources of uncertainty in the emission model are the estimated threshold friction velocity and PM<sub>10</sub> release factor. The values for these parameters were chosen based on the work in Wagenbrenner et al. (2013). Our modeling approach assumes that all fire scars in the U.S. Great Basin emit PM in the same manner as the Jefferson Fire scar emitted PM in Wagenbrenner et al. (2013). Wagenbrenner et al. (2013) was the first (and to our knowledge, still the only) study to report vertical dust fluxes from a fire scar; thus, we believe these are the best estimates for post-fire emission parameters at this time. The values used in the current study are within the range reported in Wagenbrenner et al. (2013).

Wagenbrenner et al. (2013) reported two values for threshold friction velocity: 0.20 m s<sup>-1</sup> for the fall and 0.55 m s<sup>-1</sup> for the spring period following snowmelt. A constant value of 0.22 m s<sup>-1</sup> was used in this work. This value is close to the fall value reported in Wagenbrenner et al. (2013). We think this is appropriate, as the increase in threshold friction velocity at the site in Wagenbrenner et al. (2013) was likely induced by accumulated snow and subsequent snowmelt. The burned areas simulated in this work, however, never experienced snow cover and thus, should be more representative of the fall conditions. Additionally, inspection of the modeled friction velocities in this work indicated that a threshold friction velocity of 0.55 m s<sup>-1</sup> would have given unrealistic results. For example, the largest friction velocity modeled on the Long Draw fire was 0.54 m s<sup>-1</sup>. That means that no PM<sub>10</sub> would have been emitted from that site with a threshold friction

velocity of 0.55 m s<sup>-1</sup>, which is not correct. Increasing (decreasing) the threshold friction velocity by 10% to 0.24 m s<sup>-1</sup> (0.20 m s<sup>-1</sup>) decreases (increases) the estimated total emitted PM<sub>10</sub> to 27.9 Tg (37.4 Tg).

Wagenbrenner et al. (2013) reported average PM<sub>10</sub> release rates for three pre-revegetation dust events. The value used in this work (0.0007 m<sup>-1</sup>) is between the late fall and early spring values reported in that study. Increasing (decreasing) the PM<sub>10</sub> release factor to the highest (lowest) pre-revegetation event-averaged value of 0.0066 m<sup>-1</sup> (0.0003 m<sup>-1</sup>) reported in that study increases (decreases) the estimated total emitted PM<sub>10</sub> to 303 Tg (13.8 Tg). Assuming the actual PM<sub>10</sub> release factor was within the pre-revegetation range reported in Wagenbrenner et al. (2013) and the actual threshold friction velocity was within 10% of the fall value reported in the 2013 study, total emitted PM<sub>10</sub> is estimated as 32.1 Tg with a range of 11.7–352 Tg.

For the annual emissions work, we assume uniform, dry, bare soil with characteristics similar to that of the Jefferson Fire scar within all burn perimeters. Essentially, we assume that the impact of the fire overwhelms differences in soil characteristics, including soil texture and moderate changes in soil moisture. Given the lack of PM measurements from fire scars and the documented severe effect of wildfire on surface soils (Ford and Johnson, 2006; Ravi et al., 2007; Varela et al., 2010), we believe this is a reasonable assumption. We do not account for non-erodible areas, such as rocky outcroppings; however, we assume these areas would also not be susceptible to fire and so would not be found within the burn perimeters investigated in this work (at least not at scales larger than the resolution used to map the fire perimeters). We assume the final fire perimeter exists 24 hours after the start date of the fire based on Wagenbrenner et al. (2013). The Jefferson Fire reported in that study burned 109,000 acres in roughly two days in late July 2010. Dust emissions were visible the day after the fire was contained and persisted for almost a year post-fire. This included emissions after precipitation events and following a 3 month period of snow. Emissions did not tail off in Wagenbrenner et al. (2013) until vegetation began to grow back at the site.

Additional evaluation of the PM emission model is desirable; however, lack of on-site post-fire PM flux measurements prohibits direct evaluation of the emission parameterization. This limits model evaluations to indirect metrics, such as downwind atmospheric PM concentrations, in-situ or remotely-sensed observations of dust plumes, or derived parameters such as AOD. We conducted additional analyses of the largest predicted events using Air Quality System (AQS) PM data, HYSPLIT modeling, and satellite imagery (S4). Results confirm that post-fire dust is a potential source of atmospheric PM, but also highlight the limitations associated with using existing networks and remotely-sensed data for dust source attribution. Ultimately, additional on-site measurements of post-fire PM flux are needed to constrain the emission estimates.

#### 4. Conclusions

The Long Draw case study demonstrated the ability of simulating a large post-fire dust event with a high-resolution dust emission model linked with a regional air quality modeling system. The onset and general dynamics of the observed plume were captured by the modeling framework. The emission model predicted emission hot spots within the fire scar that corresponded to areas of higher friction velocities. It was necessary to drive the emission model with on-site wind observations in order to capture the high winds associated with nearby thunderstorms. These high winds were underestimated by the meteorological model used to drive transport and dispersion of the emitted PM, which resulted in mistimed and underestimated peak model concentrations downwind. Despite these issues with the forecast meteorology used to drive the transport model, modeled 24-hr average concentrations ( $279 \mu\text{g m}^{-3}$  at Boise and  $561 \mu\text{g m}^{-3}$  at Nampa) were on the order of the observed 24-hr average concentrations ( $483 \mu\text{g m}^{-3}$  at Boise and  $385 \mu\text{g m}^{-3}$  at Nampa).

Our 2012 emissions estimate indicates that Great Basin fires could generate more  $\text{PM}_{10}$  and  $\text{PM}_{2.5}$  as post-fire dust than all fires combined in the continental U.S. (CONUS) release in smoke plumes during high fire activity years. While this estimate is large, it seems plausible given the source strength reported for the Jefferson Fire scar in Wagenbrenner et al. (2013) and for the Long Draw Fire scar in this case study. The rangeland landscapes of the Great Basin support less biomass per unit area than other landscapes susceptible to fire (e.g., forests). Additionally, less biomass available for fire consumption results in less PM as smoke. Rangeland fires are also some of the largest wildfires in the U.S. since fire spreads quickly through the dry sagebrush and grass fuels and relatively topographically simple terrain (compared to mountainous areas) of this region. Eight of the wildfires in 2012 were larger than 10,000 ha and the three largest fires burned more than 100,000 ha. Larger burned areas result in more exposed soil and ultimately more PM as dust.

Given the estimated substantial contribution of post-fire landscapes to the PM inventory and the likelihood that these sources will persist and possibly grow in the future, it is important to better quantify and characterize post-fire PM emissions. This study provided a first estimate of annual post-fire emissions for a high fire activity year; however, many assumptions were made due to lack of PM measurements in post-fire environments. This work largely relied on the assumption that fire scars in the Great Basin emit PM in the same manner as the Jefferson Fire scar reported in Wagenbrenner et al. (2013). Future studies should focus on narrowing the uncertainty around emission estimates and better characterizing post-fire dust sources. We project that post-fire dust could be an important PM source in steppe ecoregions around the world, such as the Eurasian Steppe belt and the cold Patagonian steppe.

#### Data accessibility statement

Data and code associated with this project are publicly available at <https://github.com/nwagenbrenner/postfire-dust> and <https://github.com/firelab/windninja>.

#### Supplemental Files

The additional files for this article can be found as follows:

- **Supplemental file 1: Video S1.** Video footage of the August 5, 2012 dust event. <https://doi.org/10.1525/elementa.185.s1>
- **Supplemental file 2: Figure S1.** MODIS Terra Satellite imagery of a post-fire dust event. <https://doi.org/10.1525/elementa.185.s2>
- **Supplemental file 3: Figure S2.** Base reflectivity measured by NEXRAD at Boise, ID (KCBX). <https://doi.org/10.1525/elementa.185.s3>
- **Supplemental file 4: Text S1.** Analysis with AQS data. <https://doi.org/10.1525/elementa.185.s4>

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#### Competing Interests

The authors have no competing interests to declare.

#### Contributions

- Performed the dust emission simulations, carried out the analysis, and wrote the paper: NW
- Performed the transport and dispersion simulation: SC
- Conceived of the central idea: BL
- Helped with the analysis: SC, BL
- Edited the paper: SC, BL

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