

The effectiveness of adding fire for air quality benefits challenged: A case study of increased fine particulate matter from wilderness fire smoke with more active fire management

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ABSTRACT

The Lion Fire 2011 (LF11) and Lion Fire 2017 (LF17) were similar in size, location, and smoke transport. The same locations were used to monitor both fires for ground level fine particulate matter (PM_{2.5}). Ground level PM_{2.5} is used to determine the relative smoke exposure from fire management tactics used during LF11 and LF17. The LF11 had a defined and determined perimeter and the fire, largely, grew to the containment lines with meteorological and fire conditions often determining the fire spread and intensity. For smoke management and air quality concerns, the LF17 introduced additional fire in an effort to speed up the burn and take advantage of good dispersal conditions similar to a prescribed fire. The LF17 had 2151 ha of fire added while the LF11 had only 874 ha. While emissions of PM_{2.5} for the LF17 (8062 Mg) were less than half the LF11 (19,105 Mg), ground level concentrations of PM_{2.5} were greater for the LF17 at all smoke impacted sites. The sites of Johnsondale and Camp Nelson experienced the highest concentrations for both fires with an increase mean concentration for the entirety of the fires from 5.8 μgm^{-3} for the LF11 to 26.0 μgm^{-3} for the LF17 at Johnsondale ($p = 0.003$) and 4.9 μgm^{-3} (LF11) to 35.9 μgm^{-3} (LF17) at Camp Nelson ($p = 0.01$). The National Ambient Air Quality 98th percentile daily average increased from the LF11 to the LF17 from 35.0 μgm^{-3} to 57.3 μgm^{-3} at Johnsondale and 28.0 μgm^{-3} to 52.6 μgm^{-3} at Camp Nelson. Adding fire as a tactic for good smoke dispersal to mitigate smoke exposure, as was one of the decision parameters for LF17, increased ground concentrations and exposure of smoke to surrounding communities above what was experienced during the LF11.

1. Introduction

Wildland fire managers, when possible, are increasingly integrating smoke impacts and health concerns to the decision process for managing wildland fire for ecological benefit and fuel reduction. The goal is to use fire tactics (e.g. adding fire) to mitigate exposure while still obtaining the ecological benefits of fire. These decisions are largely based on emissions estimations and dispersion modeling. Decisions are largely based on emissions reductions although emissions are not a good predictor of exposure and the subsequent impacts to human health (Navarro et al., 2016). It is unknown if fire operational strategies ultimately lead to the desired outcome of reduced smoke exposure on surrounding communities.

Wildland fire is increasing across the western United States (Dennison et al., 2014; Miller et al., 2009; Nigro and Molinari, 2019) with fire season expected to lengthen and have an increase in both

severity (Flannigan et al., 2013) and area burned (McKenzie and Littell, 2017). Global warming has increased the severity and probability of extremes in temperature and drought (Diffenbaugh et al., 2017). While research into smoke effects is often focused on extreme fire and smoke events, the role of fire that is within the historic normal of a given environmental system and the trade-offs from altering this smoke cycle are often ignored. While ecosystem and human health are coupled (Everard et al., 2013), smoke management is almost singularly focused on individual events and not trade-offs between increased ecologically beneficial fire over time versus suppression and prescribed fire (Schweizer and Cisneros, 2017). This narrow vision of smoke management is foundational policy for air regulators and land managers and is likely not leading to the desired outcomes of reducing the health burden (Cisneros et al., 2018; Schweizer et al., 2019).

The health and resiliency of the fire prone forests of the California Sierra Nevada are dependent on ecologically beneficial fire (Boisramé

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et al., 2016; Meyer, 2015; van Mantgem et al., 2016). Fire has played an integral role in forest health by routinely consuming fuel and cycling nutrients in the California Sierra Nevada (Kilgore, 1973; Miller and Safford, 2017). The historic policy of fire suppression in the United States has essentially removed the fundamental ecosystem process of frequent low to moderate intensity fire in the Sierra Nevada (Stephens et al., 2018). Suppressing fire has increased fuel load and continuity increasing the risk of a large catastrophic wildfire when suppression fails. Higher temperatures and a longer fire season have enhanced this risk (Collins et al., 2019). Prior to the era of suppression, fire in the Sierra Nevada of California was self-regulated through frequent low intensity burns that regularly removed much of the forest floor fuels. The mosaic recent burns created would largely slow or stop the next burn limiting the size and extent of individual fires.

The era of suppression eliminated much of these routine smoke emissions and stored them as excess fuel loading and created an unsustainable “no smoke” expectation to a segment of the public (Schweizer and Cisneros, 2017). While a large increase in ecologically beneficial fire is needed for a fire resilient forest (Barros et al., 2018; Stockdale et al., 2019), smoke management policy tends to focus only on human health and not the role of smoke in this environmental system. Frequent burning, as was seen historically in Sierra Nevada forests, distributed emissions from wildland fire over time substituting extreme smoke events with more routine smoke events. While many land managers have the option to increase beneficial fire, public opinion about smoke biases the response toward full suppression and prescribed fire.

Smoke from wildland fire is managed largely to reduce immediate exposure to the public with little consideration given to long term health exposure and outcomes (Schweizer and Cisneros, 2017). Paradoxically, increased use of ecologically beneficial fire can reduce smoke exposure (Navarro et al., 2018). Allowing fire that is expected to burn within historic normal size and intensities in the Sierra Nevada can reduce overall smoke exposure when suppression fails and results in a large high intensity fire (Schweizer et al., 2017, 2019).

Fuel treatments, including prescribed fire (the planned lighting of fire to meet management objectives with predetermined acceptable weather, fuel, and safety considerations) or mechanical removal, can be unrealistic or unaffordable in some systems (Bradstock et al., 2012). For added safety and control, prescribed fires often are ignited in times with higher relative humidity (RH) or fuel moisture than can occur during natural ignition fires, which can increase ground level concentrations of particulate matter $< 2.5 \mu\text{m}$ ($\text{PM}_{2.5}$) and alter the timing and extent of ground level exposure (Schweizer et al., 2019). The implementation of fuel treatments (e.g. thinning, prescribed fire, etc.) alone does not solve the fuels issue and will likely require landscape reintroduction of ecologically beneficial fire (Vaillant and Reinhardt, 2017). Nonetheless, fuel treatments are an essential tool around communities and to prepare for re-introduction of fire in areas of extreme fuels.

$\text{PM}_{2.5}$ is a criteria pollutant with both California and federal national standards or thresholds. Anthropogenic emissions cause many areas including the California Central Valley to be in non-attainment of $\text{PM}_{2.5}$ standards. $\text{PM}_{2.5}$ is also an excellent metric to assess exposure to wood smoke (Naeher et al., 2007; Vedal and Dutton, 2006) and is widely used to determine smoke exposure and health impacts (Liu et al., 2015). The health effects from $\text{PM}_{2.5}$ exposure have been researched for many years (Pope and Dockery, 2006).

Areas throughout the Sierra Nevada are largely below federal and state ambient air quality standards for $\text{PM}_{2.5}$ (Cisneros et al., 2014). However, much of this area is considered non-attainment because of current regulatory boundaries. Air quality compliance for rural areas is often determined by monitors in high population density urban areas with high levels of anthropogenic pollution. While wildland fire smoke is an important contributor to $\text{PM}_{2.5}$ in rural areas closer to an individual fire, they are insignificant to attainment/non-attainment of urban sites in California (Preisler et al., 2015). Observed ground level

concentrations of $\text{PM}_{2.5}$ at these locations is a metric that can be used to assess fire management tactics and inform the discussion of health outcomes and smoke exposure that relates to these standards (Schweizer et al., 2017).

According to federal guidance (National Interagency Fire Center, 2009) there are 2 types of wildland fires, planned (prescribed) and unplanned (wildfire). Unplanned wildland fires can include planned ignition under certain conditions. Planned ignition can vary from extremely limited firing operations to broadcast burning (applying fire generally to most or all of an area). While a simple and effective way to explain the planned use of fire during an event consistently to the public, for the understanding of smoke exposure (exposure to smoke estimated by ground level $\text{PM}_{2.5}$ concentrations), this guidance can create a limited view of possible alternatives when fire is used as a tool in wildland fire management during an unplanned event. While not a terminology problem per se, this simplification can lead researchers, air regulators, public health officials, and others not versed in fire management to potential bias in analysis of relative impacts because smoke exposure from unplanned fires is only assessed as a simple binary option of either high intensity catastrophic fire or not.

There is some control for timing and intensity in fire management, but fire will occur. Fires allowed to burn for ecological benefit are effective for managing forest health in the Sierra Nevada (Meyer, 2015). These fires can be managed using various levels of tactics that can be as simple as largely allowing the fire to burn naturally up to natural barriers (rock cliffs, rivers, previous burns, etc.), constructing containment lines, by actively slowing down the fire using water drops, or speeding up the burn by introducing fire through hand or aerial ignition. Fires, at times, can be sped up for smoke management to change the timing of the smoke (e.g. air regulators will prefer burning and releasing emissions during certain atmospheric conditions or times of the day or week). At times, little to no fire is added while at other times, it is possible to manage the fire similar to a prescribed fire where large areas of firing operations are essential if any burning is to occur. The desired smoke dispersal and transport conditions (good dispersal) used in prescribed fire, which is often at higher humidity and fuel moistures, can lead to an increase in $\text{PM}_{2.5}$ exposure to surrounding communities which is more consistent with a high intensity wildfire of the same size (Schweizer et al., 2019). Therefore, understanding the nuances of tactics used to add fire on a given fire is essential to understanding the relative tradeoffs to ground level smoke exposure between individual fires.

Current smoke management is focused on the assumption that, when not full suppression of a large high intensity fire, all fire management tactics will provide similar short and long term smoke exposure. Smoke impacts to air quality can often be framed as a choice between smoke from a high intensity catastrophic wildfire and all other wildland fire, making the assumption that this category accurately reflects all the fire management actions. However, limiting assessment of fire management tactics reduces the efficacy of relative comparison analysis of smoke impacts and the full understanding of the tradeoffs between intended and actual impacts from smoke for both public health concerns. There is a need to understand smoke exposure between different fire management actions to determine if exposure is reduced and results in the best health outcomes and if these actions are more effective than allowing ecologically beneficial wildland fire in designated Wilderness and other protected areas of wilderness.

2. Material and methods

2.1. Study area

$\text{PM}_{2.5}$ concentrations caused by two fires of similar size in the same area both managed for fuel reduction and ecological benefit during different years were analyzed. Transport and dispersion models were used to determine and compare major transport patterns. Smoke

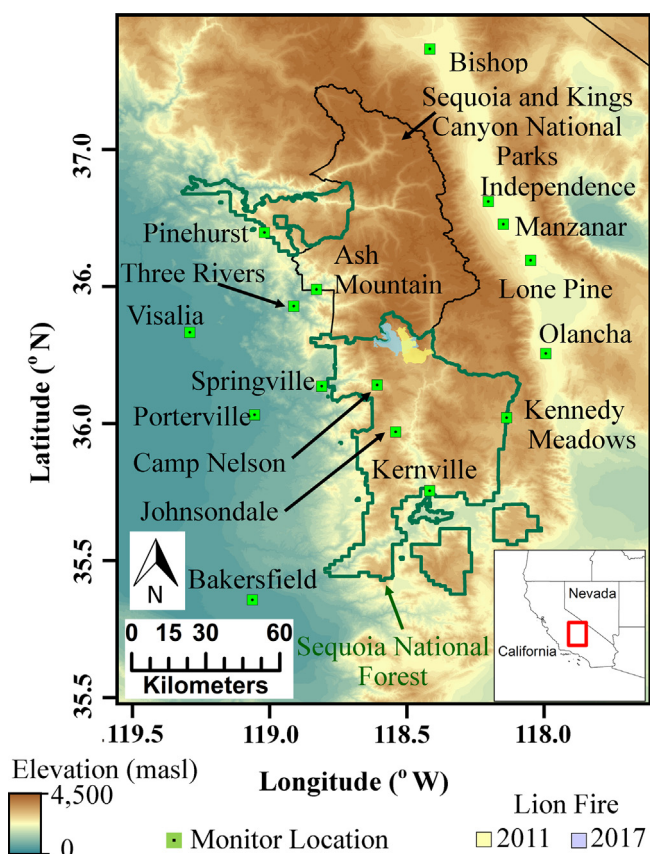


Fig. 1. Fire perimeters and fine particulate monitoring locations.

plumes were further assessed using remotely sensed data to compare spatial extent and transport. Emission estimates were calculated to explore differences in fuel loading and type for smoke exposure differences between the fires.

The Lion Fire 2011 (LF11) and Lion Fire 2017 (LF17) burned in the southern Sierra Nevada largely on the Sequoia National Forest (Fig. 1). Smoke impacts from the LF11 were analyzed and the fire management tactics used on this fire were suggested as a potential strategy to reduce adverse health outcomes in a biome characterized by frequent fire (Schweizer and Cisneros, 2014). LF11 data as analyzed by Schweizer and Cisneros (2014) is used to compare $PM_{2.5}$ concentrations with those of the LF17. The $PM_{2.5}$ monitoring strategy during the LF17 was similar to the LF11 and included permanent and temporary monitors that were placed in the same locations (Fig. 1).

2.2. Smoke transport patterns

Remote sensing data along with dispersion modeling and additional monitoring surrounding the fire were used to determine transport and locate ground level smoke impact for each fire. HYSPLIT, a hybrid model of Lagrangian and Eulerian methodology, normal and frequency forward trajectories computed using archived trajectories (HYSPLIT, 2018) were used to analyze smoke transport from each fire. Personal observations by the authors along with National Oceanic and Atmospheric Administration Hazard Mapping System Fire and Smoke Product (NOAA, 2018) satellite derived smoke density data (HMS) were used to validate transport model runs, explore differences in transport and dispersal between the fires, and to determine the contribution from other wildland fires during the study period. HMS data provide light, medium, and heavy density smoke polygons. HMS can be used to predict where ground level exceeds the expected $PM_{2.5}$ concentration. Exceeding the normal is ~12% and ~36% more likely under medium

and heavy density polygons, respectively; while light concentrations do not show a significant difference to the normal (Preisler et al., 2015). Therefore, medium and heavy density HMS data were used to compare ground level impact spatial extent. The spatial extent of the likelihood of ground level impacts exceeding expected $PM_{2.5}$ was calculated using corrections of 0.12 for medium and 0.36 for heavy density polygons (Preisler et al., 2015). Other BAM and EBAM monitoring data (AQMIS, 2018; NPS, 2018) in the region were used to verify smoke impacts to ground level concentrations of $PM_{2.5}$. These sites include all surrounding monitors and are used to confirm the presence or absence of ground level smoke.

2.3. Emission estimates

Recent fire history for the burn areas was compiled to estimate fuel loading for emission calculations. Emissions calculations were done using a First Order Fire Effects Model (FOFEM, 2017) using vegetation types determined using the National Land Cover Database (NLCD, 2014). Primary cover types in FOFEM were Sierra Nevada Mixed Conifer (SAF 243) and Red Fir (SAF 207) using “Natural” fuel category and, for consistency between fires, model defaults for model parameters (e.g. Season, Dead fuel adjustment factor, etc.). Daily fuel consumption was estimated from 65 to 141 $Mg\ ha^{-1}$ and $PM_{2.5}$ emission estimates from 1 to 3 $Mg\ ha^{-1}$. Emissions estimates were used to assess the smoke output from differences in fuel type and loading.

2.4. Ground monitor data collection and handling

Continuous hourly readings from ground based $PM_{2.5}$ monitors, permanently sited Beta-Attenuation Monitors (BAMs) and temporary Environmental Beta-Attenuation Monitors (EBAMs), were used to quantify smoke impacts to air quality. BAM and EBAM monitors were audited at a minimum of once every two weeks where integrity of flow (leak check), temperature, pressure, and flow audits were performed. Data were validated if the audits were passed and there were no logged errors. Additionally, data quality control and quality assurance included validation of internal RH and flow (Schweizer et al., 2016). The network of $PM_{2.5}$ monitors was used not only to quantify smoke impacts at a given site, but, as importantly, to establish the absence of smoke impacts to a given monitoring site. Monitoring sites were selected to ensure impacts were not missed by gross transport analysis.

$PM_{2.5}$ was measured hourly at ground level (sample at 2 m above ground). $PM_{2.5}$ daily concentrations (24 h mean; minimum of 18 valid hours needed for a valid daily concentration) are used to compare monitoring sites to ensure valid agreement between monitoring methods (Schweizer et al., 2016). Smoke was not present at a site for the entire 24 h each day but would reach high hourly concentrations at a given site at similar times each day due to consistent transport patterns.

Breakpoints for $PM_{2.5}$ Air Quality Index (AQI) were calculated using the United States Environmental Protection Agency thresholds (Good < 12.1 $\mu g\ m^{-3}$, Moderate 12.1–35.4 $\mu g\ m^{-3}$, Unhealthy for Sensitive (UHS) 35.5–55.4 $\mu g\ m^{-3}$, Unhealthy 55.5–150.4 $\mu g\ m^{-3}$, Very Unhealthy (VU) 150.5–250.4 $\mu g\ m^{-3}$, and Hazardous > 250.5 $\mu g\ m^{-3}$), which are based on 24 h average (U.S. Environmental Protection Agency, 2018).

Thresholds for the National Ambient Air Quality Standards (NAAQS) for $PM_{2.5}$ are the maximum amount of $PM_{2.5}$ allowed in outdoor air averaged over a period of three years. There are two primary NAAQS standards to determine attainment of $PM_{2.5}$. One is the NAAQS $PM_{2.5}$ annual average, which is the average of the last three annual means and has a threshold of 12.0 $\mu g\ m^{-3}$. The other is the NAAQS 98th percentile, which is the average of the last three years annual daily mean 98th percentile and has a threshold of 35 $\mu g\ m^{-3}$ (U.S. Environmental Protection Agency, 1999). The annual daily mean 98th percentile for NAAQS exceedance has been estimated as the 5th

highest daily mean at temporary monitors in the Sierra Nevada where the highest PM_{2.5} concentration days attributable to smoke can be used as an indicator of NAAQS compliance at a temporary monitoring site (Preisler et al., 2015; Schweizer et al., 2017; Schweizer and Cisneros, 2014, 2017).

Significance of the differences in concentration of PM_{2.5} between each fire was determined using non-parametric statistical hypothesis testing. Non-parametric tests were chosen largely because of limited sample size for the highest concentration days compared between fires. Additionally, non-parametric tests avoided the assumption of normal distribution and limited the effect of outliers. Two-sample non-paired Wilcoxon test is used to compare PM_{2.5} mean concentrations between the LF17 and LF11 for each of their durations. A paired Wilcoxon test (signed-rank) was used to compare LF17 to LF11 for highest daily PM_{2.5} concentrations (e.g. highest 10 daily means) as PM_{2.5} impacts from wildland fire smoke in the Sierra Nevada have been shown to be most significant during the days of highest concentrations as seen in the NAAQS annual daily mean 98th percentile (Cisneros et al., 2014; Schweizer et al., 2017; Schweizer and Cisneros, 2014). The 10 highest concentration days were used to focus on smoke impacted days to compliment the assessment of smoke impacts for the duration of each fire. Calculations and figures were done using the R Software Environment (R Core Team, 2018).

3. Results

3.1. Fire overviews

The LF11 was started by lightning (36.268 latitude; -118.511 longitude) on 7/8/2011 and burned through 9/2/2011. 8370 ha burned over the entire LF11 with the largest increases in fire growth from July 7 to 31, 2011. LF11 smoke had the largest measured PM_{2.5} impacts at Johnsondale, Camp Nelson, and Kernville (Schweizer and Cisneros, 2014). The LF11 was the only significant (> 1000 ha) fire during the 2011 fire season that increased PM_{2.5} above expected background concentrations. LF11 burned 592 ha more area than LF17 which burned over a shorter time period and occurred later in the calendar year (Table 1).

The southern Sierra had more wildland fires in 2017 than 2011 over the entire season. There were 3 fires that impacted PM_{2.5} monitors other than the natural ignition LF17 (latitude 36.271; longitude -118.486). The Schaeffer Fire, Indian Fire, and Pier Fires all contributed to high concentrations of PM_{2.5} at the monitors used in this study but did not impact during the same time period. The Pier Fire had the closest timing of smoke impacts to LF17. The five days between were typical background concentrations with smoke impacts to PM_{2.5} dropping from 9/17/2017 to 9/20/2017 at the sites most impacted by the Pier Fire (Table S1). While monitoring sites were impacted by smoke at other times, smoke impacts during LF17 were found to be entirely due to the LF17.

3.2. Smoke transport patterns

Spatial extent for HMS medium density was 939 ha for the LF17 and 788 ha for LF11 while heavy density was 254 ha during LF17 and 295 ha during LF11. The total HMS spatial extent for the likelihood of ground level PM_{2.5} exceeding the normal PM_{2.5} was similar between fires with LF17 estimated at 204 ha and 201 ha for LF11. LF11 smoke

transport was generally east and northeast over the Owens Valley into Nevada with smoke drawn into the Kern River drainage at night (Schweizer and Cisneros, 2014). LF17 followed similar general transport and atmospheric ventilation with upper air winds typically from the west with HYSPLIT forward trajectories modeling transport east into and over the Owens Valley. Transport was largely northeast over the Owens Valley for both fires. Smoke settled into the Kern River drainage at night with ground level smoke having increased concentration at local sites surrounding both the LF17 and LF11 (Camp Nelson, Johnsondale, Kennedy Meadows, Olancho, and Kernville).

3.3. Emission comparison

Fuel loading was less for LF17 partly from what had burned in the LF11 and additionally from other fires that burned within the LF17 perimeter. LF11 burned 3695 ha within the LF17 perimeter. There were a total of 8 other fires that burned 1674 ha within the LF17 perimeter from 2003 to 2017. Three fires within the LF17 perimeter re-burned 249 ha of the LF11. Prior to 2003, the last fire was in 1949 and burned 12 ha within the LF11 perimeter. In the last 62 years, 3% of the area within the LF11 perimeter had burned.

While a total of 261 ha had burned from 1949 within the LF11, 3695 ha or 48.3% of the LF17 had been burned since 2003. Twenty six percent of the total area burned in the LF17 (2021 ha) had burned in the LF11. The LF17 perimeter contained 90% of this area of re-burn on 10/3/2017. After this date, the LF17 burned primarily north and south of the LF11 perimeter. Wildfire occurring in the years 2011–2017 reduced fuels and the subsequent emissions for LF17.

LF11 emissions were primarily from Red Fir (53%) and Sierran Mixed Conifer (31%) forests. LF17 emissions were from Sierran Mixed Conifer (48%) and Red Fir (34%). LF17 PM_{2.5} emissions were less than half of the LF11; in part from the smaller overall fire size, the overlap of the burn area from the LF11, and the overall increased area previously burned by other fires in recent years. Emissions were over 2 times larger for the LF11 with the maximum single day emission over 1.5 times larger for LF11 than LF17. The maximum single day of emissions for LF11 was 2612 Mg and 1678 Mg for LF17. The second highest day of emissions on LF17 was less than half (776 Mg) the maximum. LF11 emissions were higher than the maximum day of emissions for the LF17 on 4 days. Emissions estimates were not a good predictor of Daily PM_{2.5}. While emission estimates, as expected, had some linear correlation with ground level concentrations, correlation at the impacted sites (Camp Nelson, Johnsondale, Kennedy Meadows, Olancho, and Kernville) for either fire was weak when trying to compare emissions from different fires for overall effect (Table 2). Using emissions to predict general effect to ground level PM_{2.5} improved when all impacted sites were used particularly over a longer averaging time (rolling 7 day average of emissions for all sites). Rolling 7 day emissions most significantly correlated at Johnsondale and Camp Nelson during the LF11 (Table 2).

3.4. Fire management and ground level PM_{2.5}

LF11 had more limited firing operations (fire personnel using aerial or hand ignition similar to prescribed fire operations) than LF17. There were 7 days of the 53 days when the LF11 was actively burning when fire was added; totaling 874 ha (10% of total area burned). Fire was used to enhance control lines and moved into limited areas to reduce intensity and undesired fire effects. The LF17 used firing operations to ignite more area than the LF11. LF17 used fire tactically more similar to a prescribed fire by using broadcast burning when weather conditions were advantageous. The LF17 used firing operations on 8 days of the 30 days of the fire accounting for 2 times the area (2151 ha and 28% of the total area burned) of the LF11. LF17 fire and air managers primarily used these firing operations to take advantage of good atmospheric dispersal with the ultimate goal of reducing smoke impacts to the

Table 1

Lion Fire 2011 and 2017 fires size, growth, and smoke impact.

Fire	Regional smoke impact	Fire growth occurred	Size (ha)
Lion Fire (LF11)	7/8-9/2/2011	7/9-9/2/2011	8370
Lion Fire (LF17)	9/27-10/19/2017	9/26-10/31/2017	7649

Table 2

Linear regression comparison of Lion Fires 2011 (LF11) and 2017 (LF17) emissions and ground level fine particulate matter (PM_{2.5}) at smoke impacted sites.

Site	Fire	Linear Regression of Daily PM _{2.5} :			
		Concentration vs emissions		Concentration vs rolling 7 day emissions	
		R ²	p-value	R ²	p-value
Kennedy Meadows	LF11	0.155	0.141	0.144	0.129
	LF17	0.058	0.024	0.189	0.016
Johnsondale	LF11	0.072	0.045	0.428	< 0.001
	LF17	0.017	0.533	0.156	0.119
Camp Nelson	LF11	0.021	0.285	0.319	< 0.001
	LF17	0.082	0.031	0.113	0.147
Olancho	LF11	0.075	0.118	0.036	0.284
	LF17	0.039	0.004	0.198	0.014
Kernville	LF11	0.113	0.008	0.227	< 0.001
	LF17	0.083	0.122	0.226	0.008
All sites	LF11	0.063	< 0.001	0.240	< 0.001
	LF17	0.033	0.035	0.152	< 0.001

surrounding area. October 11–15, 2017 all had ignition and fire growth averaged 279 ha per day with the largest growth of 405 ha on October 14, 2017. These days accounted for the highest daily values during the fire for Kernville, Springville (45.5 $\mu\text{g m}^{-3}$), Kennedy Meadows, Camp Nelson, and Johnsondale (Table 3).

Similar to LF11, LF17 had the largest impacts to PM_{2.5} concentrations measured at Johnsondale, Camp Nelson, and Kernville with Olancho, Pinehurst, and Kennedy Meadows also having substantial increases in daily PM_{2.5} concentrations. The highest daily concentrations for these sites were 74.8, 76.7, 81.3, 68.4, 32.5, and 71.5 $\mu\text{g m}^{-3}$ higher, respectively, during LF17 than LF11 (Table 3). Sites with the largest impacts during LF11 (Camp Nelson, Johnsondale, and Kernville) were $\sim 70 \mu\text{g m}^{-3}$ (mean 74.5; SD 4.9) higher on the highest concentration day during LF17 (Table 3). Camp Nelson had the highest daily for both LF11 and LF17. Springville during LF17, similar to LF11, had only 1 of the highest 10 daily PM_{2.5} concentrations attributed to LF11 and LF17 with the highest day being 10.9 $\mu\text{g m}^{-3}$ lower during the LF17.

For the entirety of each fire, mean concentrations were typically higher during the LF17 than the LF11 but the difference was not significant (Table 4). This is consistent with other studies showing the highest daily concentrations are a better indicator of smoke exposure than long term averages that tend to be near background except at the most smoke impacted sites (Preisler et al., 2015; Schweizer et al., 2017; Schweizer and Cisneros, 2014).

PM_{2.5} at the monitoring sites of Porterville and Bakersfield were consistent of expected background (Chow et al., 2006; Cisneros et al., 2014), and LF17, similar to LF11, did not significantly impact these monitors. The sites at Ash Mountain, Springville, Three Rivers, Independence/Manzanar Bishop, and Lone Pine were similar for each fire. PM_{2.5} concentrations were similar between Independence (LF11) and Manzanar (LF17). This was the only smoke impacted site that did not experience a relative increase from LF11 to LF17. This was likely due to

differences in site locations as Manzanar (LF17) was northwest of Independence (LF11) and approximately 6 km further from the fire (Fig. 1).

PM_{2.5} at Visalia had a significant ($p < 0.001$) increase in PM_{2.5} for the LF17 from the LF11. The median daily increase for the highest 10 days was 14.2 $\mu\text{g m}^{-3}$ ($p = 0.002$). This increase could not be attributed to the LF17 as transport models and remote sensing data suggested the increase was not from the LF17 as high days did not correspond when smoke had potential to be at the site. Local sources that normally impact Visalia (Ying and Kleeman, 2006) likely contributed to this increase, which was similar to the $\sim 5 \mu\text{g m}^{-3}$ seasonal increase at Central Valley sites from August to October (Cisneros et al., 2014). Additionally, the pattern described by Cisneros et al. (2014) would imply that background (no smoke) would have been lower during LF17 at all of the smoke impacted sites. Therefore, the background concentrations of PM_{2.5} at the smoke impacted sites during the LF17 would have been lower than during LF11 due to the time of year. Since the increase of PM_{2.5} at Visalia was not definitively from LF17, we have limited discussion of smoke impacts at this site.

PM_{2.5} was significantly increased during the highest 10 days of smoke from the LF17 than during the LF11 at the most smoke impacted sites of Camp Nelson, Johnsondale, Kennedy Meadows, Olancho, Kernville, and Pinehurst (Fig. 2). The highest 10 days of PM_{2.5} median values increased from the LF11 to the LF17 by 9.4 $\mu\text{g m}^{-3}$ to 23.4 $\mu\text{g m}^{-3}$ at these sites (Table 4). These sites all showed an increased mean PM_{2.5} from LF11 to LF17 over the entire fire but we only found a statistically significant increase at Pinehurst where it was 6.2 $\mu\text{g m}^{-3}$ ($p = 0.001$) higher during LF17 (Table 4).

3.5. AQI and NAAQS at smoke impacted sites

All sites surrounding the fires were either similar between fires or saw increased PM_{2.5} from LF17. The most smoke impacted sites were similar for LF11 and LF17 all experienced increased PM_{2.5} from the LF17. Other sites with the potential to be impacted by LF11 or LF17 smoke experienced similar PM_{2.5} with no site experiencing a significant decrease. Therefore, AQI and NAAQS 98th percentile focuses on these 6 impacted sites (Camp Nelson, Johnsondale, Kennedy Meadows, Olancho, Kernville, and Pinehurst).

LF17 was half the duration of the LF11 (Table 1). However, LF17 had poorer AQI conditions. LF17 caused an additional day of very unhealthy and unhealthy for sensitive groups at Camp Nelson. Johnsondale had an additional 3 unhealthy days. Kernville had an additional 1 unhealthy and 2 unhealthy for sensitive groups days during LF17. The LF17 caused 2 additional days of unhealthy and 1 day of unhealthy for sensitive groups at both Kennedy Meadows and Olancho. Pinehurst had 4 additional days of unhealthy for sensitive groups during LF17.

The other fires during 2017 did not contribute smoke during the LF17. These fires did contribute to higher seasonal concentrations during the 2017 fire season; particularly at the Kennedy Meadows and Springville sites. Exposure to smoke for the full 2017 fire season in most rural mountain communities was higher than when only considering the LF17. The LF17 typically contributed only 4–6 of the highest 10 days concentration days for many sites while LF11 was responsible

Table 3

Comparison of differences in fine particulate matter concentrations from the 2011 and 2017 Lion Fires at smoke impacted sites.

Site	Daily PM _{2.5} ($\mu\text{g m}^{-3}$)		Mean	2011 Lion	Standard Deviation	2011 Lion
	Maximum	2017 Lion				
Camp Nelson	215.3	138.7	36.5	16.1	49.0	19.2
Johnsondale	137.4	62.6	33.8	17.9	33.7	12.2
Kennedy Meadows	98.0	26.5	14.7	8.3	21.4	5.0
Olancho	96.3	15.0	14.6	5.8	20.6	2.4
Kernville	136.8	68.4	30.2	20.1	27.7	12.0

Table 4

Comparison of differences in fine particulate matter concentrations from the 2011 and 2017 Lion Fires.

Wilcoxon test: Site	Two-sample non-paired Difference in location in $\mu\text{g m}^{-3}$ (95% conf)	p-value	Signed-rank for 10 highest smoke impacted days Median in $\mu\text{g m}^{-3}$ (95% conf)	p-value
Camp Nelson	4.4 (−0.2/14.4)	0.063	18.2 (7.1/38.9)	0.002
Johnsondale	6.3 (−1.2/16.4)	0.099	21.2 (11.7/42.5)	0.002
Kennedy Meadows	−0.2 (−2.5/3.6)	0.873	9.9 (3.7/37.6)	0.002
Olancho	1.4 (−1.0/4.5)	0.389	23.4 (6.2/43.4)	0.002
Kernville	4.5 (−1.3/12.1)	0.120	9.4 (4.7/36.5)	0.002
Pinehurst	6.2 (2.5/10.8)	0.001	17.1 (11.1/23.4)	0.002
Springville	−7.1 (−9.2/−3.4)	0.003	−0.9 (−4.7/0.9)	0.322
Independence/Manzanar	0.6 (−1.3/2.9)	0.709	−2.8 (−5.2/−0.3)	0.037
Ash Mountain	2.8 (0.2/6.2)	0.035	1.6 (−13.0/6.5)	0.386
Bishop	−1.9 (−2.8/−0.4)	0.008	−1.3 (−2.0/0.9)	0.084
Lone Pine	0.9 (−0.4/2.5)	0.142	−2.7 (−3.6/−1.7)	0.002
Visalia	4.4 (2.1/7.8)	< 0.001	14.2 (10.7/16.9)	0.002
Porterville	1.7 (−1.5/5.7)	0.392	10.7 (7.1/12.0)	0.002
Bakersfield	3.4 (0.0/7.4)	0.055	10.9 (9–11.8)	0.002
Three Rivers	−2.7 (−4.9/2.0)	0.207	1.2 (−1.4/2.6)	0.232

for all of the highest days (Schweizer and Cisneros, 2014). Therefore, for consistency of comparison between LF11 to LF17, annual 98th percentile for 2017 are estimated as the 5th highest day occurring during the LF17 for sites without year round data while a NAAQS annual 98th percentile calculation was used for Kernville (Schweizer and Cisneros, 2014).

The highest daily 98th percentile site (Johnsondale) for LF11 was at the NAAQS threshold ($35.0 \mu\text{g m}^{-3}$) for NAAQS while 4 sites (Camp Nelson, Johnsondale, Kernville, and Pinehurst) were above the threshold during the LF17. 98th percentile was highest during the LF17 at the Johnsondale site ($57.3 \mu\text{g m}^{-3}$) and nearly twice the exposure

level of LF11 (Table 5). $\text{PM}_{2.5}$ 98th percentile was high enough that to be in attainment of the NAAQS standard (3 year average of the annual 98th percentile), the annual 98th percentile at Johnsondale for the next 2 years would need to average $23.8 \mu\text{g m}^{-3}$ or lower. NAAQS 98th percentile for $\text{PM}_{2.5}$ at Camp Nelson, Kernville, and Pinehurst were 52.6 , 48.0 , and $48.2 \mu\text{g m}^{-3}$, respectively. While LF11 smoke impacts were at or below NAAQS, LF17 smoke impacted sites are vulnerable to non-attainment of NAAQS.

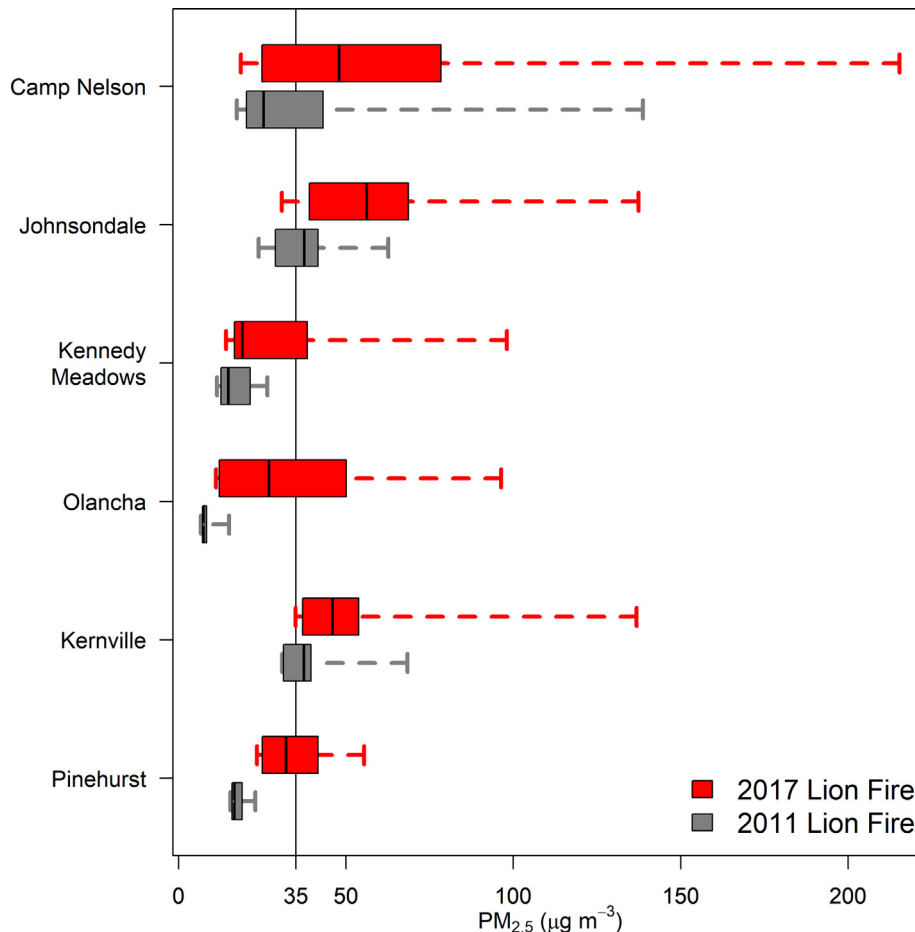
**Fig. 2.** Boxplots of the highest 10 concentration days during the 2017 and 2011 Lion Fires at smoke impacted sites.

Table 5

National Ambient Air Quality Standard (NAAQS) for fine particulate matter (PM_{2.5}) annual and estimated annual 98th percentile for select sites with smoke impacts from the 2011 and 2017 Lion Fires.

Site	98th percentile PM _{2.5} for the year (µgm ⁻³)	
	2011 ^a	2017
Camp Nelson	28.0 ^b	52.6 ^b
Johnsondale	35.0 ^b	57.3 ^b
Kennedy Meadows	17.6 ^b	19.6 ^b
Olancho	7.4 ^b	32.4 ^b
Kernville	34.0	48.0
Pinehurst	19.8	48.2

^a From Table 2 of Schweizer and Cisneros (2014).

^b 5th highest daily mean.

4. Discussion

Determining when and where to use prescribed, full suppression, and allowing, to some extent, fire to burn is a complex decision often with little to no quantitative assessment post fire for desired outcomes for reducing smoke effects on the public. Fire management strategies when allowing a fire to burn for ecological benefit are often determined by human tolerance and predicted smoke impacts with little post burn assessment of these utilized fire management tactics.

Managing fire with these ambiguities and the multiple valid and conflicting ways of framing the problem is difficult and a known challenge to decision making in natural resource management (Brugnach et al., 2011; Sullivan et al., 2018). Smoke considerations are often heavily influenced by public opinion with little quantifiable evidence if the fire management tactics are effectively working to reduce overall exposure and produce the desired public health outcomes. Assessments are needed that quantify both smoke impacts for the single fire event and the consequences of suppressing fire on air quality (Engel-Cox and Hoff, 2005). As importantly, providing the best health outcomes in a smoke prone area requires a thorough understanding of the effectiveness in reducing the public's exposure to smoke through the introduction of fire into the ecosystem or the increased use of fire as a natural process for forest health and ecological benefit.

Fire management actions are never for one single reason. Decisions are often immediate responses for firefighter safety or the protection of life and property. However, when smoke management is the single or most important reason in the decision process, there is a need to understand if the fire management actions are having the anticipated reduction of smoke exposure. Our analysis of smoke impacts to PM_{2.5} suggest smoke exposure is not always reduced by the fire management actions as intended. In this case, the addition of fire during good atmospheric dispersal increased smoke impacts to all sites in the surrounding communities. The transport and dispersal conditions potentially transported smoke to surrounding communities that would have otherwise, during more stable conditions, remained in the undeveloped areas of wilderness nearer to the fire. PM_{2.5} monitors in the same locations for both fires and surrounding the fires showed increased impacts from the fire that more actively used fire largely for air management and public health goals of reducing smoke exposure. Additionally, altering the smoke cycle can change fundamental ecosystem functions involving wildland fire smoke (Kobziar et al., 2018; Kumari et al., 2015; Parmeter and Uhrenholdt, 1975).

Quantifying relative smoke impacts from various wildland fire management actions is difficult. There are many limitations to this single comparison analysis. The LF17 occurred later in the year (October) and the higher elevation (> ~500 m) monitoring sites would have a slightly lower background while the lower elevation sites would have higher background (Cisneros et al., 2014). Monitors were strategically placed but coverage necessarily must be assumed to accurately represent the broader area. These factors make broad generalizations

difficult and warrant further studies and other methods to verify. However, few examples of a natural ignition wildland fire would have such a similar location, size, and intensity of fire with similar PM_{2.5} monitoring. Although gaps in ground based monitors could have missed smoke impacts from LF11, impacted sites were the same for each fire and PM_{2.5} increased at all sites during LF17. While smoke transport was similar for each fire, it was of course not exactly the same and thus certain sites would be expected to see larger smoke impacts while other sites experienced less. Unexpectedly, the LF17 saw an increase in PM_{2.5} at all of the smoke impacted sites increasing the health burden to communities surrounding these wilderness fires. The consistency in these increases suggest smoke mitigation assumptions need to be challenged and strategies used on a wildland fire for smoke management goals need more thorough scrutiny for efficacy.

Emission estimates give some indication of smoke impacts, but were not reliable as a surrogate of ground level exposure to PM_{2.5} at monitors for LF11 or LF17. The use of air regulatory thresholds already in place, such as to the NAAQS PM_{2.5}, have been suggested as an effective way to manage smoke and reduce exposure to it from wildfire in a wilderness or largely undeveloped fire prone forest ecosystem (Schweizer et al., 2017). Returning fire to these landscapes can ameliorate smoke management if the goal is to reduce exposure to smoke. If air quality objectives of LF17 were to reduce the number of days of smoke in the air, and to transport smoke east, this was accomplished. LF11 was more successful if air quality objectives were to reduce the extent and level of ground level smoke exposure from wildland fire to surrounding communities.

For LF17 and LF11, estimates of emissions did not equate to smoke impacts. While the highest single day of LF11 emissions were 1.5 times greater and all emissions were over 2 times greater, the highest daily concentrations and 10 day exposure both were lower during the LF11. Daily AQI was also worse during the LF17 with more Unhealthy days. It is possible exposure to PM_{2.5} could have been reduced had similar emission reductions occurred while using less active fire management tactics on an ecologically beneficial fire in the fire prone landscape of the Sierra Nevada. The atmospheric dispersal during the LF17 increased smoke exposure at our sites, although it did shorten the fire duration by 31 days. LF17 ten highest days of smoke increased at the most impacted sites 94–234 µgm⁻³.

Smoke from wildland fire is going to occur. Allowing fires to burn with less anthropogenic manipulation in remote areas with no life or property at risk has the potential to reduce public exposure to smoke.

5. Conclusions

Fire management tactics used during the LF17 increased the extent and use of firing operations compared to the LF11. Emissions were much less for the LF17, largely due to the reduced fuel loading from the LF11. Adding fire reduced the number of days of smoke during the LF17 but increased exposure to residents of surrounding communities. LF11, with over 2 times the PM_{2.5} emissions, saw significantly reduced exposure to surrounding communities (highest daily averages ~70–90 µgm⁻³ lower); suggesting the tactic of speeding up the LF17 increased air quality impacts and smoke exposure.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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