Title
Fine Particulate Matter and Wildland Fire Smoke: Integrating Air Quality, Fire Management, and Policy in the California Sierra Nevada

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Publication Date
2016

UNIVERSITY OF CALIFORNIA, MERCED

Fine Particulate Matter and Wildland Fire Smoke: Integrating Air Quality, Fire Management, and Policy in the California Sierra Nevada

A dissertation submitted in partial fulfillment of the requirements for the degree of Doctor of Philosophy

by

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in

Environmental Systems

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2016

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Table of Contents

List of abbreviations

APCD	Air Pollution Control District
AQI	Air Quality Index
BAM	Met One, Inc. Beta-attenuation Monitor
CAA	the Clean Air Act
CARB	California Air Resources Board
EBAM	Met One, Inc. Environmental Beta-attenuation Monitor
EPA	United States Environmental Protection Agency
FEM	Federal Equivalency Method
FLM	Federal Land Manager
FRM	Federal Reference Method
HMS	Hazard Mapping System Fire and Smoke Product
HYSPLT	Hybrid Single Particle Lagrangian Integrated Trajectory Model
MODIS	Moderate Resolution Imaging Spectroradiometer
NAAQS	National Ambient Air Quality Standards
NASA	United States National Aeronautics and Space Administration
NOAA	National Oceanic and Atmospheric Administration
NPS	National Park Service, Department of the Interior
$PM_{2.5}$	Fine particulate matter less than 2.5 microns in aerodynamic diameter
RAWS	Remote Automatic Weather Station
SIP	State Implementation Plan
USFS	United States Forest Service, Department of Agriculture
WUI	Wildland urban interface

List of Figures

List of Tables

Acknowledgements

I would like to thank my wife Deb along with the rest of my family and friends for their support. I would also like to thank my committee members: Ricardo Cisneros for his help and input through the entire process; Samuel Traina for his steadying influence and many years of advising; Teamrat Ghezzehei for his input and interest; and Glenn Shaw for his insight and collaboration. Thanks also to everyone who helped me on my educational journey which included the National Park Service and US Forest Service. Numerous people from these agencies helped me learn the combination of land management, politics, integrity, and science necessary for success.

I would like to acknowledge my co-authors for the publications included in this dissertation: Ricardo Cisneros and Glenn Shaw for the fine particulate monitor comparison used in Chapter 2, accepted for publication in *Atmospheric Pollution Research*; Ricardo Cisneros for the Lion Fire fine particulate matter manuscript in Chapter 3, published in *The Journal of Environmental Management*; and Ricardo Cisneros for the policy recommendation paper from Chapter 5, accepted for publication in *Air Quality, Atmosphere and Health*.

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Abstract

Title: Fine Particulate Matter and Wildland Fire Smoke: Integrating Air Quality, Fire Management, and Policy in the California Sierra Nevada
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Wildland fire is an important component to ecological health in the Sierra Nevada. It is essential to understand smoke impacts from full suppression policy that has produced a smoke averse public if this natural process is restored to the landscape. Smoke is easily visible and has air quality impacts easy to assess with fine particulate matter ($PM_{2.5}$) monitoring. Little research has been done to understand the benefits of managing ecologically beneficial wildland fire smoke. This dissertation looks at particulate matter in the Sierra Nevada in the context of fire management through prescribed, managed, and full suppression. Mobile particulate monitors, widely used as temporary smoke monitors throughout California, are assessed for their validity in comparative analysis with federal compliance monitors. The 2011 Lion Fire, a managed fire on federal wilderness, is used as a case study for smoke impacts. Fine particulate matter data from urban areas of the Central Valley to rural communities near the fire are analyzed for human health exposure. A permanent fine particulate monitoring site mid-elevation in the Sierra Nevada is used to assess wildland fire smoke impacts across the landscape over time. Implications and improvements to fire and air policy and regulations are discussed to attempt to merge short and long term public health goals.

Chapter 1.0 Smoke in the Sierra Nevada

Fire is a natural process integral to the Sierra Nevada environment determining vegetation distribution and structure (Kilgore, 1981; Swetnam, 1993; Swetnam et al., 2009). Smoke is an inevitable consequence. Native American tribes have a long history attributing fire and smoke to successful landscape management including active and widespread use of fire to increase desired results (Levy, 2005). As C.H. Merriam chief of the U.S. Division of Biologic Survey wrote in 1898 of smoke and visibility in the Sierra Nevada (Cermak, 2005):

> "Few see more than the immediate foreground and a haze of smoke which even the strongest glass is unable to penetrate"

Numerous other accounts attest to much more smoke was encountered during European settlement of the Sierra Nevada. Wildland fire smoke in California in the late 19th century was said to "choke up the atmosphere" and with any increase "…our farmers will be able to cure bacon and ham without the aid of a smokehouse" but "Nobody seemed to care; it was all public land, and what is everybody's business is nobody's business" (Cermak, 2005). There was a largely indifferent attitude to wildland fire and smoke among the mountain residents during European settlement of the Sierra Nevada with a general belief that it was an essential process (Cermak, 2005).

The benefits of wildland fire slowly moved out of favor as land was developed and industrialized (Cermak, 2005). Fire as an ecological process necessary in the Sierra Nevada has become better studied and understood in the 20th century. Fire has been widely accepted as an important ecosystem component (Beaty and Taylor, 2008; Collins et al., 2007; Kilgore, 1973; Miller et al., 2012; Pausas and Keeley, 2009) while smoke research has largely focused on air quality and impacts to public health (Adetona et al., 2016). As population increases in California and more people move into the wildland urban interface (WUI), wildland fire policy and air regulatory policy will likely continue to conflict (Jacobson et al., 2001).

Suppression of wildland fire has been the normal management action in the United States since the 19th century. Increased fuels and a changing climate are creating a post suppression era where large high intensity destructive wildfires are becoming more common. Wildland fire in the Sierra Nevada, CA and throughout much of the American west is expected to increase in activity in the post suppression era with increased large fire frequency, longer duration fire, and a longer fire season (Westerling et al., 2006). Fire suppression policy dating back 150 years' has created western forests with an abundant fuel loading problem (Steel et al., 2015). The fuel loading problem implies an emissions problem. A return of fire to the Sierra Nevada landscape necessitates confronting a possible increase of emissions from this backlog of fuels.

Smoke has not been recognized as a necessary part of modern Sierra Nevada communities while fire ecologists sound the alarm on fuels and fire. The spatial extent of wildland fire during years currently considered "extreme" may actually be closer to historic (pre-Euro-American settlement) normal burn area (Stephens et al., 2007). Policy and the public may not be ready to adapt. Meanwhile, climate change has increased the length of the fire season (Flannigan et al., 2013; Westerling and Bryant, 2008) and can be expected to continue or further extend the annual pattern as large wildland fire emissions in California are expected to increase with future climate scenarios (Hurteau et al., 2014). While fire science is pointing to increased fire, a disconnection in the science and policy exists (Ayres et al., 2016) with a smoke averse public and limited research on relative risk of wildland fire management actions (Gaither et al., 2015; Smith et al., 2016). Ecologically beneficial fire, or fire the size, intensity, extent, and effects historically experienced in the ecosystem, will be limited by public willingness to breathe smoke that may in part be rectified by understanding the health implications of smoke emissions and exposure under prescribed, managed, and full suppression scenarios.

Wildland fire burns have the ability to limit subsequent fire spread and lead to self-regulating landscapes (Parks et al., 2015). Suppression may very well be an unsustainable policy for landscape level land management. Additionally, fire suppression policy may have created an unrealistic expectation of smoke free air in areas which historically have seen abundant fire (van de Water and Safford, 2011) and smoke. While minimal research has been conducted on risk management coupling human and natural fire-prone forests systems similar to other natural hazards (Spies et al., 2014), smoke is almost completely ignored.

Emissions from large wildland fires analyzed using satellite data document air quality impacts (Langmann et al., 2009) with smoke toxicity (Wegesser et al., 2009) and the negative impacts of large wildland fires on human health being widely published (Tham et al., 2009) creating concern for exposure and public health. Although it is important to understand large high-intensity “extreme” wildland fire smoke events, failure to understand air quality impacts from ecologically beneficial wildfire can give a biased understanding. Extreme suppression policy beginning in the early 1900s has led to generations unaccustomed to smoke impacts from fire-adapted forests that historically have burned much more frequently.

Policy decisions have a profound effect on human and ecological health. Federal land managers (FLMs) throughout the United States have multiple acts and policies that regulate their actions. Policies allowing natural processes that emit regulated pollutants can seemingly be in contradiction with public health. In the Sierra Nevada of California, the essential ecosystem process of wildland fire in areas set aside for conservation is one such process that is in apparent conflict with air regulations.

Chapter 1.1 Smoke ecology

Fire ecology is focused on the role of fire factors such as intensity, fuel consumption, and nutrient cycling (Harvey et al., 1980; Wuerthner, 2006). The role of smoke in the ecological health of fire adapted plant and animal communities has been less studied but provides some awareness of the role wildland fire smoke has in ecosystem function.

Smoke ecology and seed germination research provides some context (Jefferson et al., 2014). Seed germination traits are little studied and often overlooked by ecologists but can provide an important component to understanding plant communities (Jiménez-Alfaro et al., 2016). Smoke has been associated with forest health as a germination cue (Baxter and Van Staden, 1994; Brown, 1993; Moreira et al., 2010; Thomas et al., 2007; Van Staden et al., 2000) and higher emergence of seedlings (Hidayati et al., 2012). Smoke and heat, for some species, does not enhance germination after fire disturbance (Tsuyuzaki and Miyoshi, 2009). Smoke likely has impacts to the environmental system that go beyond the fire burn area.

Plant species in the Sierra Nevada respond to smoke (Abella et al., 2007; Keeley and Fotheringham, 1998, 1997; Keeley et al., 2005). Smoke enhanced seed germination benefits restoration of habitat in fire prone areas (Roche et al., 1997) and is important to community ecology (Jiménez-Alfaro et al., 2016). Smoke inoculated seed research, even though stimulated in large part from agricultural benefit, helps contextualize the role of smoke in ecosystem health.

Along with smoke enhanced and inhibited germination, smoke can stimulate pollen growth in a number of plant species (Kumari et al., 2015). Additionally, the presence of smoke increases water soluble organic carbon (Mayol-Bracero et al., 2002) which can also impact germination (Dayamba et al., 2010). Wildland fire emissions can have an important role in cloud condensation nuclei activity (Mayol-Bracero et al., 2002) naturally altering local atmospheric conditions. Certain species of insects have been shown to swarm in smoke plumes where smoke plumes may serve as "mating points" and the burnt area where oviposition occurs (Klocke et al., 2011). Sensors on others point to their use of smoke to find suitable mates and egg laying locations (Álvarez et al., 2015). Additionally, smoke has been shown to reduce fungi (Parmeter, J.R. and Uhrenholdt, 1975) and inhibit germination of mistletoe (Zimmerman and Laven, 1975). There is clearly a role for smoke in forest function.

Phosphorus and nitrogen increase 5 to 60 times above background from smoke (Spencer et al., 2003). These changes have been shown to persist at least 4 years (Mast and Clow, 2008). Fire emissions as ash or aerosol blanketing an area would have an impact to the system. Extreme fire accounts for much of the largest changes in stream water (Rhoades et al., 2011). That is not to say that smaller fires would influence forest chemistry. Prescribed fire can raise soil pH and release calcium (Stephens et al., 2004). The presence of smoke and the associated "pollutants" that blanket and flood a fire prone area repeatedly can be assumed to have some consequence to forest health. While lakes in the Sierra Nevada show eutrophication and changes in acid neutralizing capacity in response to deposition and anthropogenic air pollution (Heard et al., 2014; Sickman et al., 2003), the link with atmospheric deposition from fire should not be discounted as important to these oligotrophic lakes.

Smoke ecology is an emerging research area. It is difficult to understand the natural role and function of smoke in a fire adapted ecosystem particularly after years of full suppression. Smoke likely had a more subtle impact on forest health than fire but is likely is currently being undervalued when regions in and around fire adapted environmental systems formed and evolved under a "flood" of smoke. Further study is needed to determine the role of wildland fire smoke in ecosystem health.

Chapter 1.2 Smoke and human health

The impacts to human health from wildland fire emissions (smoke) must be understood and incorporated into any discussion, management, or policy surrounding the use of wildland fire for ecological benefit. Smoke exposure can have an increase mortality while increased morbidity is more common (Youssouf et al., 2014). The economic cost to health effects from exposure to smoke is difficult to determine and may require a unique approach when comparing to anthropogenic air pollution (Kochi et al., 2010). The impacts to human health from smoke are difficult to specifically determine as duration and dispersal confound the quantification of population level impacts. This is further complicated when assessing relative impacts from altering fire suppression policy.

Air pollutants from a wildland fire are dependent on fuels, can be complex near the flame front, and interact with anthropogenic sources (Alves et al., 2010; Hosseini et al., 2013; Statheropoulos and Karma, 2007). Smoke emission can be more toxic than urban emission during large high intensity fires (Wegesser et al., 2009) but there is limited understanding of the causal factors of smoke composition including fuels, fire size and intensity, and chemicals introduced when agricultural areas and houses burn. The same fire can produce large variability in smoke composition even at the same monitoring site (Wigder et al., 2013). The variability of plume chemistry during transport along with varying dispersal conditions makes understanding individual plume toxicity challenging. It is then difficult to determine the net effects of forest fires on human health (Fowler, 2003). Population level impact assessment may provide the best estimation of the costs and benefits of smoke exposure until the underlying science of specific plumes is understood.

Particulate matter less than 2.5 microns in diameter ($PM_{2.5}$) is a large portion of emissions from wildland fire (Clinton et al., 2006) and is easily transported over long distance (Bein et al., 2008; Dokas et al., 2007) having a large impact on air quality (Fowler, 2003; Langmann et al., 2009). $PM_{2.5}$ is one of the best ways to assess smoke exposure (Naeher et al., 2007; Vedal and Dutton, 2006). Particulate matter is the most frequently studied pollutants when studying wildland fire smoke impacts in part because it can be 10 times higher than non-fire background concentrations (Liu et al., 2015). Transport and dispersal of smoke is represented in $PM_{2.5}$ exposure often far from the fire (Sapkota et al., 2005). Smoke transport can easily be detected by remote sensing (Hoff and Christopher, 2009; Wang and Christopher, 2003). Quantifying ground level concentrations of $PM_{2.5}$ using remote sensing is difficult (Toth et al., 2014; Yao and Henderson, 2013). Remote sensing and modeling can increase remote sensing estimates of ground level $PM_{2.5}$ (Li et al., 2015; Reid et al., 2015; Yao et al., 2013). Remote sensing can be used to indicate exceedances from the normal of ground level $PM_{2.5}$ concentrations due to smoke in the Sierra Nevada but ground based monitors are necessary for accurate quantification (Preisler et al., 2015).

$PM_{2.5}$ often best exhibits impacts to health providing strong evidence of acute respiratory responses particularly in persons with pre-existing respiratory conditions (Adetona et al., 2016). Large wildland fires during 2003 in southern California increased deaths from smoke exposure (Kochi et al., 2012) as high density urban areas experienced heavy smoke from these high intensity unnatural fires. The short term high concentrations of $PM_{2.5}$ from prescribed burning are also a health concern (Haikerwal et al., 2015). High concentrations of $PM_{2.5}$ will be found with any fire but reducing the spatial extent can limit exposure to populations of concern. While few associations between wildfire emissions and mortality have been observed, associations with subclinical effects have been established (Youssouf et al., 2014), but major and minor health outcomes due to wildland fire smoke need to be better identified (Kochi et al., 2010).

Regional forecasting using remote sensing may eventual lead to the best understanding of fire activity impacts to human health by identifying which fires are most likely to impact a given location (Price et al., 2012). Linking landscape ecology and epidemiological perspectives is important to reintroduction of ecologically beneficial fire into the modern world. For example, in Australia, it was noted that daily asthma presentation increase may be avoided while allowing some fire by using an airshed threshold for particulate matter (Bowman and Johnston, 2005). Urban center are an additional complexity when managing a fire prone landscape. Asthma presentations increase incrementally as particulate matter increases from smoke over these urban areas (Johnston et al., 2002). Implicit in this study is the exposure variability from routine versus large high intensity fires and the importance of weighing the relative risk of managed fire against uncontrolled wildland fire from suppression in a fire prone landscape. A metric where public health impacts from regional fires are used to estimate impacts from the number and size of fires over a given season may provide a way to reintroduce measured landscape level ecologically beneficial fire in the Sierra Nevada.

Wildland fire smoke impacts will depend heavily on level of emissions, transport, and receptor distance from the fire. The economic impacts to health can be substantial when urban areas are impacted by large high intensity fires instead of smaller fires (Rittmaster et al., 2006). Megafires can result in increased asthma emergency room visits and hospital admissions and significant economic cost (Jones et al., 2016). Protecting public health from smoke is directly dependent on controlling fire emissions. Wildland fire emissions will increase human exposure to air pollutants. Health risks of population exposure to wildland fire smoke are evident even at lower levels of $PM_{2.5}$ (Johnston et al., 2006). Fire management policy will determine the levels and extent of exposure. Controlling timing and quantity is essential. Timing and dispersal can be used to mitigate some of the health impacts of increased wildland fire (Tian et al., 2008). Complete suppression would theoretically provide solution but in practice does not work. It is apparent after over 100 years of suppression in the United States that at best full suppression is a delaying tactic that can be better said to mortgage smoke exposure to subsequent generations.

Chapter 1.3 Smoke and fine particulate matter in the Sierra Nevada

Particulate pollution is an important impact from fire on air quality (Langmann et al., 2009). Fine particulate matter ($PM_{2.5}$) with an aerodynamic diameter of less than 2.5 µm generated by wildland fire has significant health implications (Statheropoulos and Karma, 2007). $PM_{2.5}$ monitoring data is now available throughout California but is biased toward urban areas because of the Clean Air Act (CAA, 2004) and the minimum monitoring requirements from 40 C.F.R. 58 Appendix D to select densely populated areas. In addition to this urban federal and state compliance required monitoring, the US Forest Service and National Parks Service provide supplemental monitoring of more rural areas closer to wildland fire emissions. $PM_{2.5}$ monitoring in support of federal wildland fire management in the Sierra Nevada includes 5 permanent sites with additional temporary sites dependent on fire location as illustrated for deployment during the Lion Fire in 2011 (Figure 1.1).

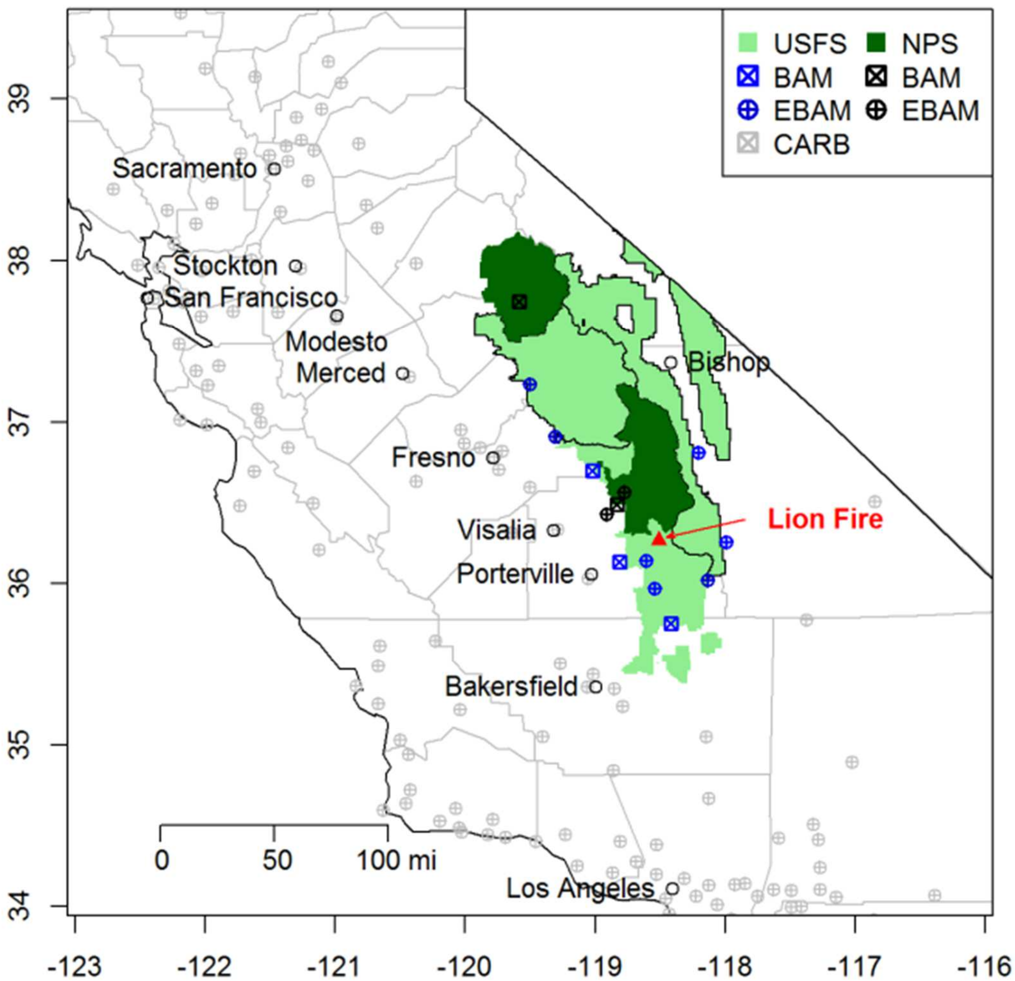


Figure 1.1. 2011 Lion Fire temporary smoke monitoring with environmental beta-attenuation monitor (EBAM) and permanent beta-attenuation monitor (BAM) sites managed by the U.S. Forest Service (USFS), National Park Service (NPS), and the California Air Resources Board (CARB).

Advances in remote sensing are increasing the applicability of this data for air quality determination. Computer vision algorithms continue to improve and assist in early smoke detection and verification of smoke density (Favorskaya et al., 2015; Zhao et al., 2015). Aerosol optical depth from satellite data has shown promise to assess ground level $PM_{2.5}$ (Hoff and Christopher, 2009). Remotely sensed aerosols occurring during the most extreme days of smoke can estimate exposure during these large events (Yao and Henderson, 2013). The National Oceanic and Atmospheric Administration (NOAA) Hazard Mapping System (HMS) can aid ground based monitoring and effectively determine the spatial extent of wildland fire smoke (Preisler et al., 2015).

Remote sensing and ground monitoring are essential to understanding wildland fire smoke impacts across the landscape. Data between monitors of different designs is currently being utilized to determine smoke impacts to $PM_{2.5}$ from wildland fire. Monitoring equipment is assumed to be sufficiently similar to allow for hourly, daily, and annual data comparison.

Determining agreement between monitor types helps to provide more precise comparative analysis of $PM_{2.5}$ concentrations that better inform public health and fire management decisions. Knowing the impacts to air quality and public health from wildland fires of varying size and intensity is essential to determining the impacts of wildland fire.

Chapter 1.4 Fire management policy

Fires have been widespread and frequent over a long period of history shaping the present environment (Scott, 2000). Wildland fire was largely seen as an integral way to manage forested land throughout much of the west by Native American tribes (Anderson, 1999, 1996). European-American settlers first moving west saw the importance of continuing these practices (van Wagtendonk, 2007). Losses of life in large wildfires such as the Peshtigo Fire (1871) in Wisconsin and the Santiago Canyon Fire (1889) in California began to instill the philosophy of suppression into fire management that would be important to the foundation of American fire-fighting policy.

Current wildland fire management and policy is a product of the 1910 fire season where 78 people died and over 8 million hectares burned and modern suppression policy originated (Silcox, 1910). This fire season also known as the “Big Burn” or “Big Blowup” was only 5 years after the United State Forest Service (USFS) was established. USFS policy to put all fires out as quickly as possible was questioned. Although light burning similar to Native American practices was used by settlers and some argued for the necessity of it being a part of sound forest policy (Koch, 1935), overwhelmingly, questioning of the policy was not over burning but how to use modern techniques and management to fully suppress wildland fire (Greeley, 1920). Reliance on private lumber companies and lack of USFS coordination was seen as a major obstacle to forest protection and health through suppression (Allen, 1910). Cooperation was seen as what failed in the now almost exclusive held perspective of policy makers that full suppression was the only way to protect forested lands. The Weeks Act (1911) designated the USFS as the agency for federal cooperation in fire suppression and was strengthened by the Clarke-McNary Act of 1924 (Southard, 2011) while the Protection of Timber Owned By United States From Fire, Disease, Or Insect Ravages (16 USC 594) was the National Park Service (NPS) equivalent.

By the mid-1930s, the policy to contain and control all fires by 10 a.m. had been adopted by the USFS and full suppression was largely in place. In this era, wildland fire was seen as an evil that could be stopped with enough money, sound tactics, and advances in science and technology. This policy was solidified during and immediately after World War II when all fire was considered evil (Figure 1.2) and the public perception of complete suppression began despite the essential need of fire in the forest (Kauffman, 2004).

Figure 1.2. 1940s era U.S. Forest Service public campaign likens forest fires to death and destruction.

The use of wildland fire began to gather interest in the 1960s as fire management cost increased and research began to demonstrate benefits (Kilgore, 1973; Parsons and DeBenedetti, 1979). In 1963, the "Leopold Report" argued that western parks should be maintained as nearly as possible to the condition when the first Euro-American settlers arrived (Leopold et al., 1963) and began to inspire policy makers to include fire management. The Wilderness Act (1964) allowed for the natural process of fire to occur and started a move to include ecologically beneficial and prescribed fire to move from fully controlled to some form of management.

The large land management organizations (typically federal, state, and tribal governments and agencies) are diverse in their missions and goals to safe and effectively manage fire at a landscape level. This creates and immediate and fundamental hurdle to a simplified one-size-fits-all policy where easy solutions for one agency are contradictory to other agencies legislative authority. The United States Forest Service (USFS) and National Park Service (NPS) frequently are located adjacent to one another spatially, but have different mandates and mission goals. The NPS, a part of the U.S. Department of Interior, is fundamentally a conservation agency with an obligation to allow natural processes to function while the USFS, a part of the U.S. Department of Agriculture also is required to incorporate sustainable harvest over much of the land they manage. Timber harvest and other anthropogenic uses are authorized in the USFS while the NPS is required to preserve the ecological integrity of the land area they manage by eliminating to the greatest extent possible anthropogenic impacts.

The need for greater agency cooperation began to enter policy after the 1988 Yellowstone fires. The 1995 "Federal Wildland Fire Management Policy & Program Review" reflected the need to integrate fire into landscape level management. Extensive fires in 2000 led to the "Management the Impact of Wildfires on Communities and the Environment: A Report to the President in Response to the Wildfires of 2000" to reduce risk in the Wildland Urban Interface (WUI).

The "Review and Update of the 1995 Federal Wildland Fire Management Policy" (2001) forms the basis of current wildland fire policy with the "Interagency Strategy for the Implementation of Federal Wildland Fire Management Policy" (2003) detailing the implementation.

The "Guidance for Implementation of Federal Wildland Fire Policy" (2009) is currently the primary policy direction. Current policy includes a "single cohesive federal fire policy" that directs agencies to consider long-term benefits of fire in relation to risks with the number one guiding principle being firefighter and public safety. The second guiding principle is the essential role of wildland fire as an ecological process needs to be incorporated into the planning process. Further guiding principles include requiring risk management to include the cost of either allowing or suppressing fire and the inclusion of consideration of public health and environmental quality into the decision process. All guidance is to be underpinned with fire management plans based on the "best available science."

Science based fire management plans in the political environment faced by policy makers can be conflicting. While air quality is a rather modern concern, fire has long been understood to perform many beneficial ecosystem functions (Kilgore, 1981) including helping to maximize carbon sequestration in fire-prone areas (Hurteau et al., 2008). Recurring wildland fire additionally limits fire spread and substantially reduces fire progression under extreme weather conditions (Parks et al., 2015) and may provide an avenue to control emissions and the subsequent health impacts. But, past fire management policy dominated by anthropogenic factors has primarily been intended to prevent or contain wildland fire with the consequence of reducing ecological integrity in fire adapted ecosystems (Dellasala et al., 2004).

Future policy will likely continue to be based almost exclusively around anthropogenic concerns unless the entrenched disincentives of current policy are overcome and proactive use of managed fires is supported (North et al., 2015a). Without understanding the impacts from wildland fire smoke under a typical fire regime it is easy to understand how suppressing all emissions for public health would appear to be sound policy. Unfortunately this is a short term solution where future emissions are essentially ignored and priority is given to restricting wildland fire emissions as much as possible with the assumption that future fire will not occur. Sound policy requires differences between these competing scenarios be quantified.

Fire suppression limited smoke when widespread air quality monitoring began provided some of the first regulatory data. This time of low emissions coupled with an increased fuel loading has created a backlog of fuels and wildland fire emissions with limited historic monitoring context. Suppression limited wildland fire emissions during the initial stages of systematic widespread air quality monitoring likely has led to inappropriate base-line estimates of air quality exposure to areas in and adjacent to the fire adapted ecosystem of the Sierra Nevada. Smoke impacts were historically much more frequent throughout the Sierra Nevada (van de Water and Safford, 2011) but the cooler slower burning wildland fire needed to sustain the Sierra Nevada ecosystem mosaic potentially made the spatial extent of these historic wildland fire smoke events smaller.

The potential for air quality regulations to limit fire management options has long been recognized as an impediment to the use of ecologically beneficial fire (Sneeuwjagt et al., 2013). Smoke will likely be a greater concern with increased fire use and acceptability of smoke levels declines (Shindler and Toman, 2003). Numerous overarching policy considerations have been offered in the public, research, and policy sectors to help ameliorate the coming together of fire and air policy. Often air quality concerns center around current regulation with wonderfully written regulatory language that has little to no practical field applicability. Fire management needs a clear path to implementation where air quality impacts are well defined by regulators and provide a quantifiable way to manage smoke for the best health outcomes in both the short and long term.

Chapter 1.5 Air management policy

As human caused impacts to natural systems and processes were beginning to be understood, the continued unregulated polluting of the air was recognized to be detrimental to ecosystem and thus human health. The atmosphere has a finite ability to buffer air pollution being emitted through urbanization and modernization of society. Air quality in the United States of America is regulated through a multifaceted approach using Federal, State, and local laws to assure compliance with clean air requirements established by the Clean Air Act (CAA).

The Air Pollution Control Act of 1955 provided initial federal legislation and funded research about air pollution to better understand the impact of anthropogenic emissions to both human and ecosystem health. With an increased understanding of anthropogenic pollution impacts to human and ecological health further legislation was created to address growing concerns. The U.S. CAA was passed in 1963. The CAA was established to improve the air quality of the nation from the excessive loading of anthropogenic emissions by creating a national program to minimize air pollution. The primary goal of the CAA is to regulate anthropogenic emissions of air pollution to protect human and ecological health through air pollution prevention and control, emission standards for vehicles used in transportation, noise pollution, acid deposition control, and stratospheric ozone protection.

As the CAA has been further defined and developed since its inception. The Air Quality Act of 1967 addressed interstate transport of air pollutants. The CAA of 1970 established National Ambient Air Quality Standards (NAAQS), requirements for State Implementation Plans (SIPS). The Clean Air Act amendments of 1977 (Public Law 95-95) addressed visibility as an air quality related value in Class I Wilderness Areas and Prevention of Significant Deterioration (PSD). The Clean Air Act Amendments of 1990 (Public Law 101-549) assures federal conformity to state and local regulations and identified 189 Hazardous Air Pollutants.

Oversight of the CAA is primarily the responsibility of the U.S. Environmental Protection Agency (EPA) with implementation delegated to state and local regulators. The EPA sets National Air Quality Standards (NAAQS) for criteria pollutants which include ground-level ozone (O_3), particulate matter (both particulate matter with an aerometric diameter of less than 10 µm (PM_{10}) and less than 2.5 µm ($PM_{2.5}$)), lead, nitrogen dioxide (NO_2), carbon monoxide (CO), and sulfur dioxide (SO_2). Federal Reference Methods (FRMs) dictate site equipment and maintenance to ensure data between all sites is comparable. Federal Equivalency Methods (FEMs) are equipment and operating procedures that meet all the requirements of data quality that were originally established by the FRM.

The primary responsibility for assuring air quality meets federal standards is delegated to the individual state. States are required to develop State Implementation Plans (SIPs) to ensure compliance with NAAQS standards by specifying how the standards will be achieved and maintained. States are responsible for meeting CAA and other relevant federal regulations. California is divided into air pollution control districts (APCDs) with regulatory oversight from the California Air Resources Board (CARB). Additionally, California has additional state mandated regulatory requirements under the California Clean Air Act of 1988. Title 17 of the California Code of Regulations (CCR, 2016) provides direction for the regulation of agricultural and prescribed burning.

The NAAQS standards established by the EPA for $PM_{2.5}$ are the annual mean and 24 hour for $PM_{2.5}$. The annual mean is the previous 3 year mean of the annual mean daily concentrations where the threshold is 12.0 µg m^{-3}. The 24 hour is a 3 year mean of the 98th percentile of daily means for a given year with a threshold of 35 µg m^{-3} (U.S. Environmental Protection Agency, 1999). These values are the regulatory thresholds that determine federal attainment or nonattainment. California has a state standard threshold for annual arithmetic mean for $PM_{2.5}$ of 12 µg m^{-3}. The southern Sierra Nevada is largely a part of the San Joaquin Valley Air Pollution Control District (SJVAPCD) and in nonattainment for $PM_{2.5}$ (Figure 1.3).

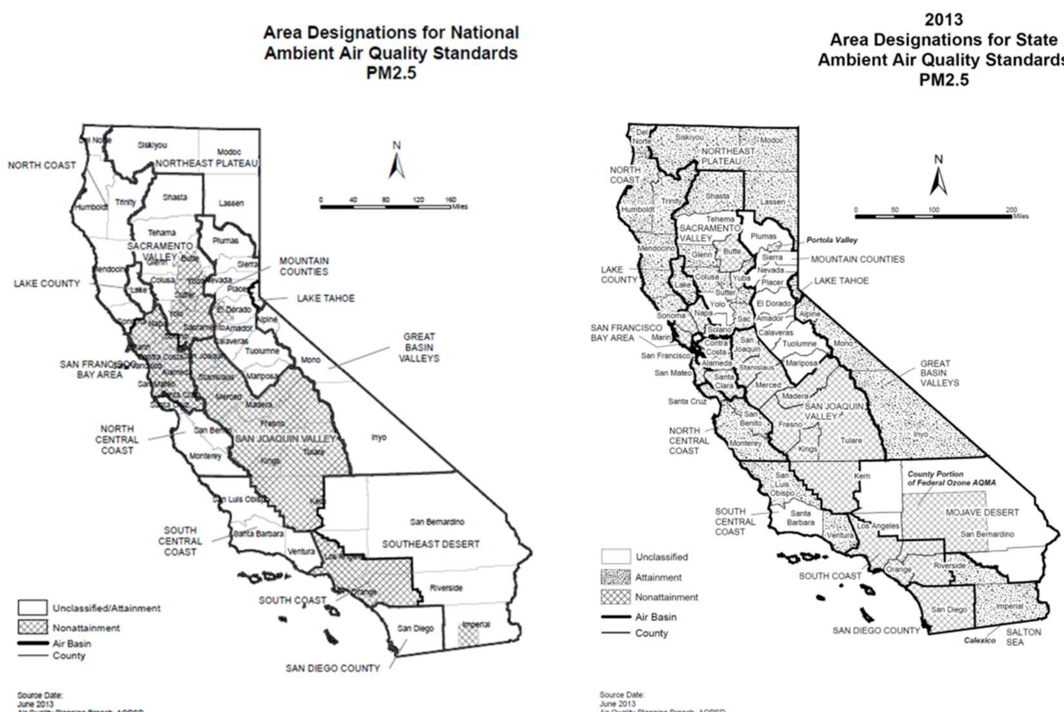


Figure 1.3. California air districts and 2013 federal and state attainment for fine particulate matter ($PM_{2.5}$).

In addition to state and federal compliance, the EPA has established the Air Quality Index (AQI) to provide a simplified rating of local current air quality. The AQI is calculated for major air pollutants (ozone, particulate, carbon monoxide, sulfur dioxide, and nitrogen dioxide) focused on immediate (few hours or days) health effects. The AQI is meant for the public and provides an easy estimate of air quality. The AQI has 6 categories (good, moderate, unhealthy for sensitive groups, unhealthy, very unhealthy, and hazardous) where the highest individual pollutant AQI value for a given day is the reported AQI. AQI for $PM_{2.5}$ from smoke can be estimated as 1-3 hour, 8 hour, and 24 hour AQI (Lipsett et al., 2013).

AQI is an estimate of daily exposure to an air pollutant where effects and precautions can be taken to minimize exposure while regulatory thresholds are an indicator of background levels of exposure to pollutants with thresholds to limit impacts to public health. Both thresholds can be used to determine relative human health impacts to $PM_{2.5}$ from wildland fire smoke.

Chapter 1.6 Objectives

Forest and air management policy is often in conflict. This is readily apparent in fire adapted ecosystems where smoke is routinely present. Fire has been a major natural mechanism in the Sierra Nevada Mountains of California providing evolutionary pressure which has shaped this ecosystem. As population has boomed throughout California, more people are living in and immediately adjacent to this fire adapted ecosystem creating a conflict not only between the immediate destruction of life and property from wildland fire but additionally subjecting larger populations to the exposure of wildland fire smoke. Wildland fire smoke may be the most reviled non-destructive byproduct of any natural process. Smelling smoke in the air immediately makes many people deem they are experiencing hazardous air quality even when smoke impacts are undetectable in background ambient concentrations.

The use of resource objective wildfires effectively restores and maintains fire-adapted ecosystems of the Sierra Nevada and their use should be expanded particularly in remote areas to mitigate the negative consequences of suppression (Meyer, 2015). While ecological benefits from fire are well established, smoke impacts are more difficult to quantify. Fuel loading, fire size, and distance from the fire are important to understanding impacts (Moeltner et al., 2013). Controlling fire size and intensity through increased use of ecologically beneficial fire may prove to be an effective tool in smoke management. There is potential for wildland fire the size and intensity historically seen in the Sierra Nevada to be managed while adhering to federal health standards. Increased wildland fire has also provided data suggesting that timing and dispersal can be used to mitigate some of the health impacts (Tian et al., 2008). Modeling of smoke plume dispersal and movement has improved to provide better estimates of forest fire exposure (Yao and Henderson, 2013). Further understanding of the spatial extent of smoke and public exposure levels under various fire size and intensity scenarios would help inform wildland fire policy about the role of ecologically beneficial fire.

The objectives of this study are: (1) to determine the relative extent and intensity of ground level $PM_{2.5}$ from wildland fire smoke in the Sierra Nevada and surrounding rural and urban areas and (2) determine the relative impacts from full suppression vs. prescribed vs. managed wildland fire on rural communities and urban areas. The hypotheses are that mobile monitors used in the Sierra Nevada for wildland fire smoke monitoring provide data with a precision adequate for comparison with federal regulatory monitors and can be used to assess ground level impacts to $PM_{2.5}$; smoke impacts from managed ecologically beneficial fire the size and intensity historically seen in the Sierra Nevada will be significantly less than megafire impacts; and a single monitoring site will provide a consistent estimate of smoke impacts over a large area and be an effective smoke management tool. The following specific questions are considered:

1. Does temporary particulate monitoring currently being widely used in the Sierra Nevada for wildland fire smoke monitoring have a good enough agreement to allow for comparison to FEM/FRM monitors?
2. What are the impacts to $PM_{2.5}$ from a managed fire of the size and intensity historically experienced in the Sierra Nevada?
3. What are the current impacts to $PM_{2.5}$ at a representative site over multiple years of monitoring?

These questions are addressed sequentially in Chapters 2 through 4 with the intention to put in context the impacts to air quality from wildland fire emissions and allow for a more nuanced and critical look at policy as it converges in air and fire management and regulation.

The introduction in this chapter (Chapter 1) is intended as a general introduction with additional specific topic introductions given in subsequent chapters. Robustness of comparative analysis between permanent regulatory monitors and the temporary fine particulate monitors currently widely deployed in the Sierra Nevada during wildland fires is discussed in Chapter 2. In Chapter 3, the natural ignition Lion Fire (2011) is used as a case study to determine fine particulate matter concentrations from wildland fire smoke and the impacts to immediate and long term human health in the surrounding area during this ecologically beneficial managed fire. Chapter 4 includes a single representative Sierra Nevada site to assess smoke impacts from 2006-2015 and the differences from emissions using different management strategies for planned (prescribed) and unplanned events (full suppression and managed for ecological benefit). Air and fire policy recommendations from these findings are presented in Chapter 5.

Differences in air quality impacts through smoke management of prescribed, natural ignition, and full suppression fires are quantified to gain a better understanding of smoke generated from the fire adapted ecosystem of the Sierra Nevada. Understanding of relative differences in smoke impact between fire size and intensity will be used to assess the current intersection of wildland fire and air quality management policy with the goal of improving smoke management in a fire adapted ecosystem.

Chapter 2.0 Limitations of agreement

$PM_{2.5}$ monitoring is widespread throughout California. The Met One, Inc. Beta-Attenuation Monitor (BAM) is an FEM monitoring technique that is used almost exclusively in California compliance monitoring of $PM_{2.5}$. Temporary wildland fire smoke monitoring is primarily accomplished using the Met One, Inc. Environmental Beta Attenuation Monitor (EBAM). The EBAM is designed to be portable and easily deployed and does not meet FEM/FRM requirements.

In addition to California regulatory monitors, the USFS Region 5 (California) utilized BAMs at 3 permanent sites in the Sierra Nevada in Sequoia and Sierra National Forests. The NPS additionally has a BAM at Yosemite National Park and another at Sequoia National Park. The USFS and NPS both have numerous EBAM temporary smoke monitors. USFS and NPS monitor smoke impacts to $PM_{2.5}$ on the extensive federal lands throughout the Sierra Nevada. This monitoring is used during a wildland fire to supplement the California urban $PM_{2.5}$ monitors predominantly in the rural areas throughout the Sierra Nevada.

Side by side comparisons of BAMs and EBAMs show hourly measurements can lack precision (Figure 2.1 (a and b)). Although precision can still vary even when averaging over 24 hours (Figure 2.1 (c)), mean concentrations are reliable enough to give near real time information to public health officials (Figure 2.1 (d)).

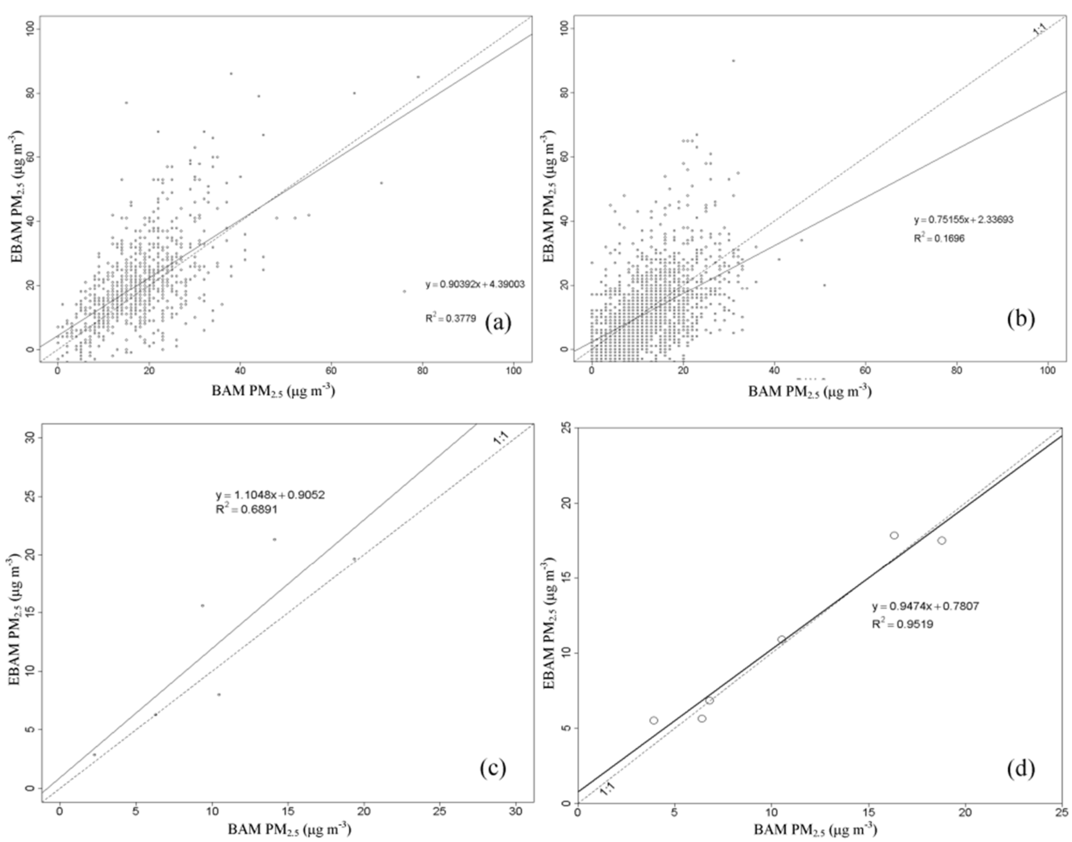


Figure 2.1 Environmental beta attenuation monitor (EBAM) versus beta attenuation monitor (BAM) side by side comparative measurements for hourly (a and b) and 24 hour (c and d) measurements.

Simple side by side comparisons are useful for initial indications of comparability. These comparisons help indicate problem areas where caution is needed. One obvious issue between these monitors comes when ambient relative humidity (RH) is high. High RH can increase error in a particulate sample by absorbing water during the sampling period. To offset this issue both the BAM and EBAM include an inlet heater that keeps the internal RH below a set point (typically 40% in USFS and NPS equipment). The EBAM is susceptible to high RH which can be partially corrected by removing hourly values when the internal RH is greater than 40% (Figure 2.2).

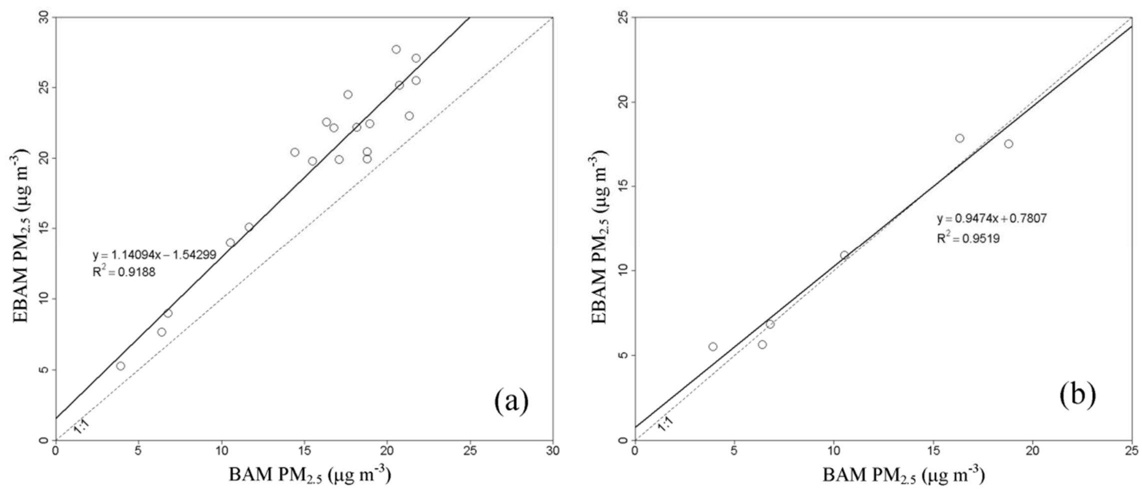


Figure 2.2 Beta attenuation monitor (BAM) versus environmental beta attenuation monitor (EBAM) 24 hour comparison for all data (a) and data corrected for internal relative humidity (b).

Agreement between monitors is important. Although at times presence or absence of smoke using $PM_{2.5}$ as an estimate necessitates only needing a rough estimate, comparing AQI and NAAQS standards require accurate and precise readings for more rigorous analysis. Using both the BAM and EBAM in smoke monitoring can at times trigger management actions at levels as low as 35 μg m^{-3}. Meticulous comparison helps to fundamentally understand the extent to which conclusions can be drawn when doing analysis using these 2 monitors.

Chapter 2.1 A comparative analysis of temporary and permanent beta attenuation monitors: The importance of understanding data and equipment limitations when creating $PM_{2.5}$ air quality health advisories.

Reprinted from the Atmospheric Pollution Research, Don Schweizer, Ricardo Cisneros, and Glenn Shaw, A comparative analysis of temporary and permanent beta attenuation monitors: The importance of understanding data and equipment limitations when creating $PM_{2.5}$ air quality health advisories, Copyright (2016), with permission from Elsevier.
DOI: 10.1016/j.apr.2016.02.003

Abstract

Mobile particulate monitors are being widely used for smoke monitoring throughout the western United States. While this provides valuable additional data for public health decisions, quantifying the field performance of this equipment is necessary to understand measurement limitations when being compared with federal compliance instruments. Met One Instruments, Inc. Environmental Beta Attenuation Monitors (EBAMs) were co-located at permanently established Beta Attenuation Monitor (BAM) sites to determine agreement under normal field operating conditions. Monitors were assessed for agreement between fine particulate matter ($PM_{2.5}$) measurements. The instruments correlated for hourly (R^2 0.70) and daily (R^2 0.90) means. Mean difference for EBAM to BAM comparison showed the EBAM over-predicting the BAM by 24% (3 $\mu g\ m^{-3}$). Hourly concentrations fluctuated more in the EBAM. Daily mean concentrations were the most equitably comparable measurement for these monitors. Increases in relative humidity (RH) were associated with increased disagreement between monitors. When EBAM internal RH was below 40%, R^2 increased (0.76 hourly, 0.93 daily). The EBAM produced higher hourly AQI estimates. As a result of this study, it is advised to invalidate hourly data when the internal RH is greater than 40% and to only use daily AQI estimates to limit the EBAM AQI over-prediction.

Keywords: Particulate monitoring equipment; $PM_{2.5}$; health advisories; accuracy; precision

Introduction

Beta attenuation mass monitors are widely used to measure fine particulate matter ($PM_{2.5}$) throughout the United States of America. The Met One Instruments, Inc. model BAM-1020 (BAM) is used at many sites to assess $PM_{2.5}$ concentrations and determine compliance with state and federal air quality standards. The environmental beta attenuation monitor (EBAM) model from Met One Instruments, Inc. is intended for rapid temporary deployment and is not a federal reference or equivalency method approved for $PM_{2.5}$ compliance monitoring in the USA. BAMs are designed to operate in a temperature controlled enclosure while the EBAM is self-contained.

Both BAMs and EBAMs provide continuous hourly measurements of particulate matter concentrations using the relationship between attenuation of beta particles and particle deposition on a glass filter tape (Macias and Husar, 1976). There have been many studies that compare and contrast different particulate monitors that typically use regression to determine accuracy between monitors (Chung et al., 2001; Hains et al., 2007; Liu et al., 2013; Shin et al., 2011; Takahashi et al., 2008; Tasić et al., 2012) and include attempts to increase agreement through correction factors (McNamara et al., 2011). Various methods can help summarize agreement and offer different perspectives on the data (Haber et al., 2010; Liao, 2003; Lin et al., 2002). The use of correlation alone does not measure agreement and can be inappropriate to determine closeness between two measurements (Altman and Bland, 1983). This can be true for closely related measurements and if the points are clustered along the 1:1 line (Bland and Altman, 1995).

BAMs are widely used throughout California for compliance determination and to advise public health officials of air quality. EBAMs are regularly deployed throughout California to help air and public health regulators during wildland fires. This smoke monitoring is used to supplement the compliance sites and help public health officials in the determination of air quality impacts in areas typically long distances from the nearest permanent air quality monitor. Widespread use of the EBAM during smoke and other air pollution events has led to routine use of the data produced by an EBAM and its subsequent use as fully representative of BAM data when assessing public health exposure. While this supplemental data is of great value to public health exposure determination in less populated areas, understanding the comparability of data from these two similar monitors when operating in typical field conditions is of critical importance.

$PM_{2.5}$ monitoring equipment including the BAM and EBAM can be expected to perform at different precision and reliability (Baldauf et al., 2001; Takahashi et al., 2008). Error can be introduced from difference in monitor design or parts (Liu et al., 2013). Appropriate use of $PM_{2.5}$ data for public health protection and the determination of attainment and exceedance of federal and state air quality standards need include an understanding of the bias between mass measuring techniques (Chow et al., 2006).

Effects from field conditions can impact measurement agreement between instruments. Temperature has been shown to cause differences even between monitors that meet federal reference method (FRM) or federal equivalency method (FEM) requirements (Zhu et al., 2007). Volatilization from filter samples has been widely documented to reduce correlation between measurements (Chow et al., 2005; Hains et al., 2007). Relative Humidity (RH), both ambient and internal heater controlled, influence BAM $PM_{2.5}$ measurements (Huang and Tai, 2008). To offset effects from humidity, BAMs and EBAMs use an inlet heater which is designed to keep the internal humidity at or below a programmed set point.

The U.S. Environmental Protection Agency (EPA) has established the Air Quality Index (AQI) system to disseminate health impacts the public may experience over the course of hours to days. EBAMs are used to better assess AQI information disseminated to the public during temporary events (i.e. wildland fire smoke) by providing additional ground based monitoring in areas often long distances from federal reference monitors. Quite often EBAMs are deployed to less populated areas where air quality monitors, typically placed in the more populated urban areas, may not accurately reflect localized air quality conditions from a temporary short duration emission event or sited to capture specific transport of temporary emissions that may be missed by a permanent monitor.

During wildland fire used for fuel reduction and ecological benefit, data analysis of particulate matter measurements necessarily must be much more nuanced as any additional $PM_{2.5}$ exposure to a population is critical to effective smoke management. $PM_{2.5}$ concentrations in this analysis are typical of scenarios across federally managed lands in the Sierra Nevada, California during a smoke event from a wildland fire the size and intensity historically seen in this ecosystem (Schweizer and Cisneros, 2014). Low concentrations are typical in these scenarios and often burning can be subjected to management actions including full suppression on the basis of an individual temporary monitor reaching a single hour of 35 $\mu g\ m^{-3}$. We attempt to quantify with what certainty the data from mobile equipment can be used to make these difficult fire management decisions when in direct comparison to BAMs used for regulatory compliance.

Understanding the reliability and precision of monitoring systems under field conditions is important when attempting to accurately represent data using different measurement techniques. We attempt to quantify the differences between the BAM and EBAM while operated under conditions typical of field deployment in California during 2006, 2009, and 2011. In this paper we are attempting to understand BAM and EBAM agreement and highlight limitations that exist when comparing measurements. In particular we endeavor to determine the efficacy of using the portable EBAM when determining air quality impacts to human health when being utilized as a temporary monitor during a wildland fire or similar temporary emission scenario.

Methods

Monitor Locations

BAMs and EBAMs were co-located at 5 locations (Figure 2.3). A large high intensity wildland fire or other event that would create high $PM_{2.5}$ conditions was not encountered during any of these deployments. Thus, the following comparison does not include the high concentrations and ranges experienced during a full suppression wildland fire smoke event when correlation between equipment may be adequate for air managers to simply understand where the largest impacts are occurring and precision is less important.

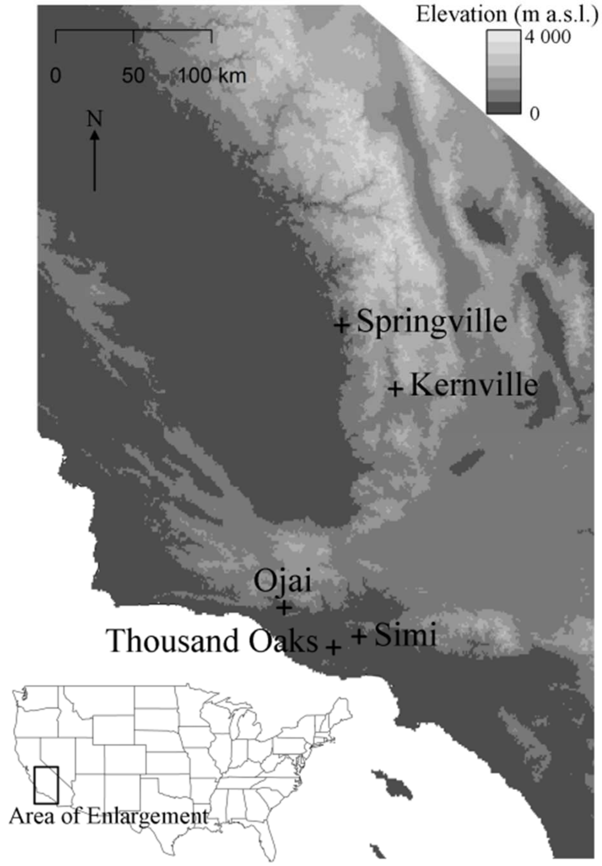


Figure 2.3 Site Locations (California, U.S.A).

Instrument Descriptions

The BAM is designated by the U.S. Environmental Protection Agency (EPA) as an FEM with hourly measurements having a standard range of 0-1 000 µg m^{-3}, resolution of ±0.1 µg m^{-3}, and 24 hour average lower detection limit less than 1.0 µg m^{-3} (Met One Instruments, 2008a). The BAM is suitable for meteorological conditions in California (Chung et al., 2001) and is widely used throughout the state for regulatory monitoring of $PM_{2.5}$.

EBAMs have a measurement range of -5-65 530 µg m^{-3}, a data resolution of 1.0 µg m^{-3}, an accuracy of ±10% of the indicated value for hourly measurements (or 2.5 µg m^{-3}), and a 24 hour average lower detection limit less than 1.2 µg m^{-3} (Met One Instruments, 2008b). The EBAM does not meet FEM requirements. The EBAM is not appropriate for compliance determination but is a more portable and less expensive version of a BAM and used for temporary monitoring of particulate matter. The EBAM hourly measurement is obtained by using two 4 minute counts. The first count in minutes 2-5 at the top of the hour establishes a zero reading while minutes 57-60 establish the total count for the hour. Tape advance can be set so a sample is accumulated over multiple hours on the same sample with the first and last 4 minute counts coming at the top of each hour. A negative value can thus be determined if the count from the beginning of the hour to the end of the hour reduces which may indicate error in the hourly sample that is being measured by the individual hour reading negative. This negative hourly value is included in our calculations when stated to determine if the negative value increased comparative accuracy when calculating 24 hour mean concentrations.

Instrument Deployment and Maintenance

Two EBAMs were run simultaneously at Thousand Oaks and Simi Valley sites in 2009 while at other sites one EBAM was used for comparison. Sampling at each site was from 1-3 months duration and all monitoring occurred between April and November (Table 2.1). Ojai and Simi were sampled in 2006; Simi, Thousand Oaks, and Springville in 2009; and Kernville in 2011. EBAMs were installed with inlets as near as possible to the same vertical height (within 0.5 m) and 1-2 m distance from BAM inlets. Internal RH set points on the BAMs and EBAMs were set to 40 or 45%. Tape advance was set between 1 and 8 hours. Protocol for equipment function included integrity of the flow (leak check), temperature, and flow audits which were performed at a minimum of once every two weeks. Data was considered valid if there were no errors logged by the instrument and all audits were passed. To correct for the noise band of several micrograms on the BAM (Met One Instruments, 2008a), the occasional negative values for the BAM were set as zero for all calculations. EBAM hourly values were calculated setting negative values to zero and including the negative values as noted in the text, figures, and documents as a separate calculation. EBAMs frequently produced a negative number before and/or after an hourly value that was much higher than the BAM. Therefore, the negative hourly values were included in one set of calculations to determine the significance of including negative hourly EBAM measurements when comparing 24 hour mean values between instruments.

Table 2.1 Site locations and monitoring dates.

Site	Latitude (N)	Longitude (W)	Start	End
Kernville	35.75512	-118.4175	8/9/2011 12:00	11/28/2011 12:00
Springville	36.13588	-118.811	7/10/2009 12:00	8/4/2009 11:00
Thousand Oaks 1	34.21014	-118.8705	9/9/2009 13:00	10/26/2009 10:00
Thousand Oaks 2	34.21014	-118.8705	8/5/2009 15:00	10/26/2009 9:00
Thousand Oaks 3	34.21014	-118.8705	8/5/2009 15:00	9/4/2009 3:00
Simi 1	34.2764	-118.68375	8/5/2009 12:00	9/9/2009 9:00
Simi 2	34.2764	-118.68375	8/5/2009 11:00	9/9/2009 9:00
Simi 3	34.2764	-118.68375	4/19/2006 13:00	5/19/2006 12:00
Ojai	34.44804	-119.23131	4/20/2006 12:00	5/12/2006 11:00

Measurement Effects from Site Conditions

Meteorological conditions were assessed using linear modeling with the difference (BAM-EBAM) in $PM_{2.5}$ hourly concentration (PM_{diff}) for individual explanatory variables of internal (heater controlled) relative humidity (RHi), ambient relative humidity (RHx), ambient temperature (t) in degrees Celsius (C), wind speed (WS) in meters per second (mps), and wind direction (WD) in degrees. Multiple linear regression was used with PM_{diff} described by multiple meteorological site conditions as:

$$PM_{diff} = t + RHx + RHi + WS + WD$$

Statistical calculations and graphics were produced using the software environment R (R Core Team, 2015).

Correlation and Agreement

Regression of BAM and EBAM measurements was used to determine the linear relationship and variance between these two methods of measuring $PM_{2.5}$. Mean difference between EBAM and BAM measurements were used to assess the agreement between hourly, 3 hour mid-point mean, and daily mean averages. The mean difference (*m*) was calculated (EBAM-BAM) along with the standard deviation of the differences (*s*). The *m* was considered the bias between the 2 measurements with the levels of the limits of agreement determined as $m+2s$ and $m-2s$ (Bland and Altman, 1986) with smaller values being better numerical agreement between EBAM and BAM measurements. Daily averages required 18 or more valid hourly readings.

AQI

The AQI was calculated to compare the effectiveness of data being used in smoke advisories. AQI was used to determine any differences when communicating to the public. The AQI reporting system has 6 categories (good, moderate, unhealthy for sensitive groups, unhealthy, very unhealthy, and hazardous) with thresholds depending on a given pollutant. Daily and 1 (and 3) hour $PM_{2.5}$ AQI is calculated using established breakpoints from the EPA and California Office of Environmental Health and Hazard Assessment (Lipsett et al., 2013). The daily or 24 hour breakpoints for $PM_{2.5}$ are good 0-12, moderate 12.1-35.4, unhealthy for sensitive groups 35.5-55.4, unhealthy 55.5-150.4, very unhealthy 150.5-250.4, hazardous 250.5-500 $\mu g\ m^{-3}$. For 1 and 3 hour $PM_{2.5}$ exposure breakpoints of good 0-38, moderate 39-88, unhealthy for sensitive groups 89-138, unhealthy 139-351, very unhealthy 352-526, and hazardous >526 $\mu g\ m^{-3}$ are used.

Results

Hourly concentrations of $PM_{2.5}$ at all sites were between 0 and 93 $\mu g\ m^{-3}$ for BAMs and -5 and 102 $\mu g\ m^{-3}$ for EBAMs (Table 2.2). Hourly concentrations measured by the EBAM routinely exceeded the BAM with EBAM hourly maximum concentrations typically greater than those of the BAM (the difference between the maximum values recorded at the instruments ranged from 7-39 $\mu g\ m^{-3}$). The exception was at Kernville where the BAM hourly maximum was 11 $\mu g\ m^{-3}$ greater than the EBAM. Daily mean concentrations were 0.9-40.1 $\mu g\ m^{-3}$ for BAMs and 0.1-48.2 for EBAMs. The EBAMs typically measured the highest daily mean concentrations which were 0.5 to 26.4 $\mu g\ m^{-3}$ above the co-located BAM. Kernville and Simi (during the sampling in 2006) had maximum BAM daily mean concentrations higher (2.0 and 6.0 $\mu g\ m^{-3}$ respectively) than the EBAM (Table 2.2).

Table 2.2 Site specific summary statistics for hourly (a) and daily (b) concentrations including correlation (R^2) between the beta attenuation monitor (BAM) and environmental beta attenuation monitor (EBAM).

(a) Correlation and comparison of hourly $PM_{2.5}$ at each site.

		Mean		Median		Max		Min		Standard Deviation	
Site	R^2	BAM	EBAM	BAM	EBAM	BAM	EBAM	BAM	EBAM	BAM	EBAM
Kernville	0.7180	9.6	12.6	8	12	93	82	0	-5	9.1	9.5
Springville	0.7440	9.2	18.0	9	19	28	49	0	-5	4.5	8.7
Thousand Oaks 1	0.6894	9.4	10.7	8	9	46	76	0	-5	6.5	12.8
Thousand Oaks 2	0.6828	11.4	10.5	10	9	51	90	0	-5	7.1	11.4
Thousand Oaks 3	0.7339	14.1	26.2	14	23	51	90	0	-5	7.4	18.3
Simi 1	0.8194	17.1	22.8	17	20	79	102	0	-5	9.3	15.1
Simi 2	0.8000	17.1	19.9	17	18	79	86	0	-5	9.3	13.5
Simi 3	0.6993	19.6	16.5	19	14	67	99	0	-5	11.7	15.6
Ojai	0.7412	15.4	14.1	15	14	29	46	0	-5	5.1	9.3
all data	0.7042	12.5	15.1	11	14	93	102	0	-5	9.1	9.5

(b) Correlation and comparison of daily $PM_{2.5}$ at each site.

		Mean		Median		Max		Min		Standard Deviation	
Site	R^2	BAM	EBAM	BAM	EBAM	BAM	EBAM	BAM	EBAM	BAM	EBAM
Kernville	0.9432	9.6	12.8	8.6	12.4	30.8	28.8	0.9	6.5	5.3	4.3
Springville	0.9737	9.3	18.3	9.5	18.4	12.3	20.9	3.6	14.5	2.3	1.6
Thousand Oaks 1	0.9100	9.5	11.3	9.0	11.0	23.3	39.5	2.3	1.4	4.8	8.3
Thousand Oaks 2	0.9704	11.5	11.1	11.8	11.6	22.5	28.0	2.4	0.1	5.2	5.6
Thousand Oaks 3	0.9459	14.3	26.5	14.8	25.8	21.8	48.2	3.1	7.6	5.0	11.8
Simi 1	0.9798	17.3	22.2	18.5	22.1	27.8	42.3	3.9	5.4	6.0	8.5
Simi 2	0.9764	17.3	20.2	18.5	19.2	27.8	40.1	3.9	6.3	6.0	8.9
Simi 3	0.9835	19.6	17.1	20.2	16.6	40.1	34.1	2.7	6.3	10.0	8.2
Ojai	0.9964	15.6	14.4	15.5	14.8	19.3	19.8	11.4	8.8	2.9	3.5
all data	0.9432	12.6	15.4	11.2	13.8	40.1	48.2	0.9	0.1	6.6	8.2

Site condition impacts

WS and RHi had the strongest correlation to PM_{diff} (Table 2.3) with higher RH leading to a pattern with noticeable loss of agreement while WS was more evenly distributed (Figure 2.4). RH is understood to impact $PM_{2.5}$ measurements with ambient temperature impacting agreement between gravimetric and beta attenuation sampling through loss attributed to volatilization of particulate nitrate and organic compounds from sample heating particularly above 20 ºC (Zhu et al., 2007). We found differences between the BAM and EBAM to be primarily correlated to a RHi above 40%. Atmospheric temperature even at higher temperatures (>=20 C) showed little correlation (Table 2.3). Since both the BAM and EBAM use a similar system of sample heating, we expected the differences between these samplers would be limited to the effectiveness of the EBAM to control RHi.

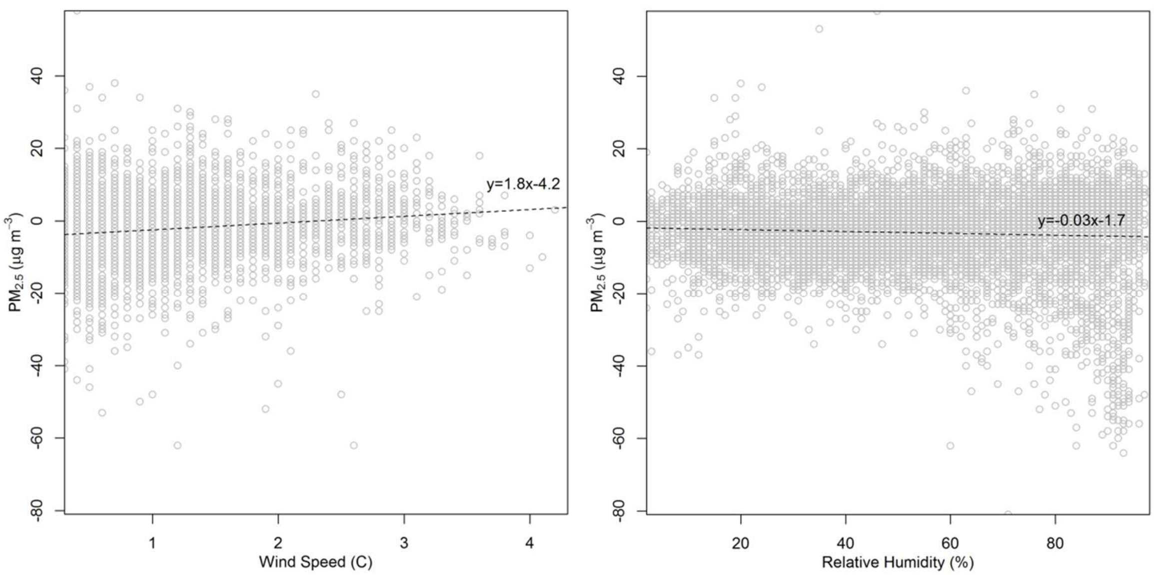


Figure 2.4 Beta attenuation monitor (BAM) to environmental beta attenuation monitor (EBAM) difference in hourly concentration of fine particulate matter ($PM_{2.5}$) as a function of wind speed and relative humidity.

Table 2.3 Assessment of individual meteorological explanatory variables on difference in fine particulate matter ($PM_{2.5}$) hourly concentrations between the beta attenuation monitor (BAM) and the environmental beta attenuation monitor (EBAM).

Variable	R^2	p-value
Temperature (ºC)	0.0011	0.001
RH	0.0037	$p<0.001$
Internal RH	0.0192	$p<0.001$
Wind Speed	0.0229	$p<0.001$
Wind Direction	0.0015	0.013
Internal RH >= 40%	0.1247	$p<0.001$
Temperature >= 20	0.0003	0.215

Figure 2.5, showing the partial residual plots from the multiple linear regression model, illustrates the potential impact of higher RHx and particularly RHi >45% on agreement between the BAM and EBAM. The impact of RHx and RHi for our data was in agreement with the published results for these types of monitors (Huang and Tai, 2008; Takahashi et al., 2008).

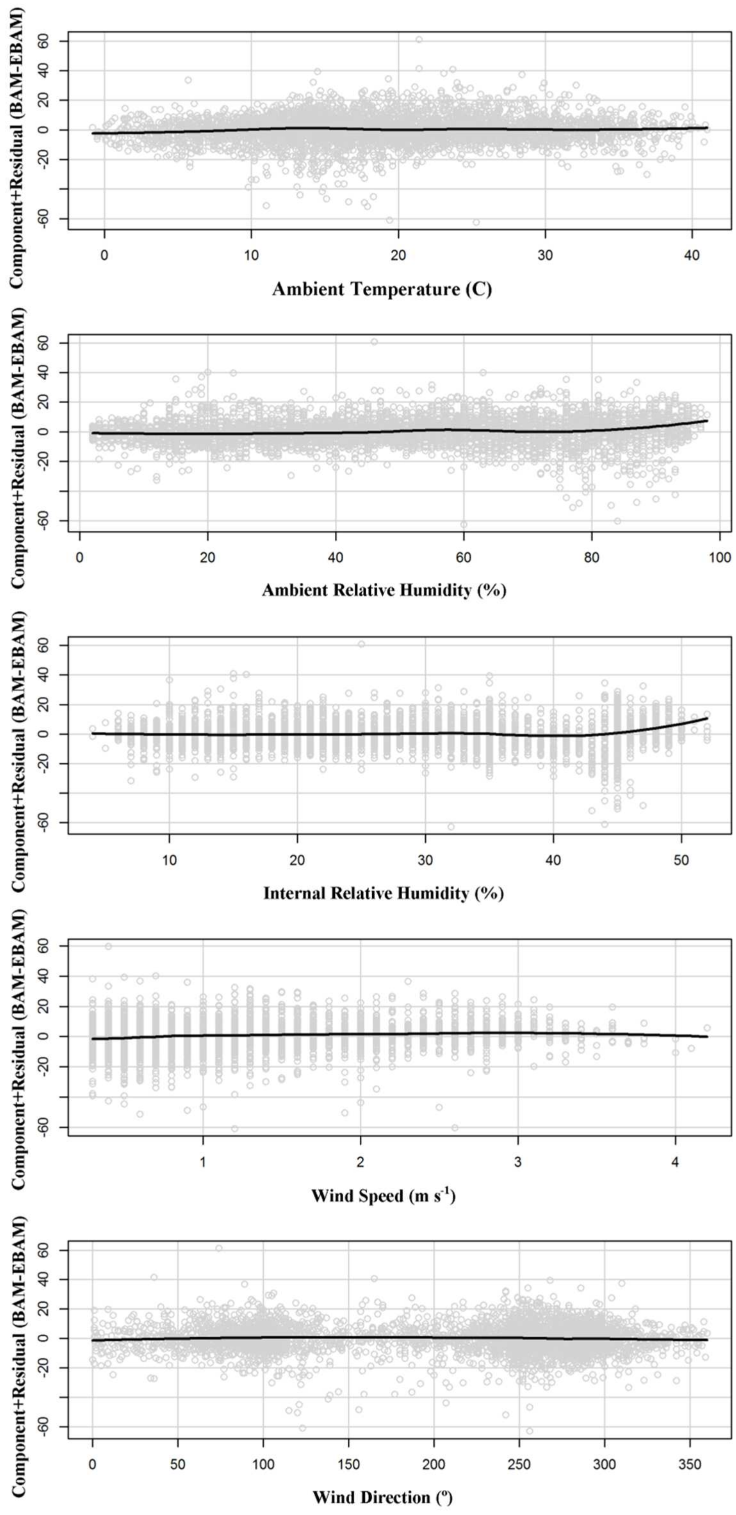


Figure 2.5 Partial residual plots with locally weighted scatterplot smoothing (LOESS) line for site environmental variables.

Correlation

Co-located sampling of ambient concentrations of $PM_{2.5}$ between the BAM and EBAM had strong correlation at each individual site with a range of R^2 for hourly data of 0.6828 - 0.8194 ($p<0.001$) and for daily of 0.9100 - 0.9964 ($p<0.001$). Correlation was strong when all sites were analyzed together with hourly measurements having an R^2 of 0.7042 ($p<0.001$) and daily values with an R^2 of 0.9039 ($p<0.001$). Best correlation of BAM and EBAM data found was an R^2 of 0.7663 for hourly and 0.9334 for daily ($p<0.001$) when the internal RH of the EBAM was below 40% and negative EBAM hourly values were replaced by zero (18 or more hourly values were required for a daily mean). Additionally, correlation of BAM and EBAM 3 hour mean (midpoint) $PM_{2.5}$ (minimum of 2 hourly values) resulted in an R^2 increase from 0.7663 for the hourly EBAM to 0.8230 using the 3 hour mean (Table 2.4).

Table 2.4 Beta attenuation monitor (BAM) and environmental beta attenuation monitor (EBAM) comparison statistics[a] for hourly, 3 hour, and daily data.

Description of hourly data used (daily mean requires 18 valid hours unless otherwise stated)	R^2		
	Hourly	3 hour	Daily
All data	0.7042	-	0.9039
Daily mean with no hourly minimum	-	-	0.8983
Negative EBAM hourly values replaced with zero	0.7178	0.7685	0.9075
Ambient RH<60%	0.7594	-	0.9275
Ambient RH<60% and EBAM hourly values of negative replaced with zero	0.7714	0.8184	0.9267
EBAM internal RH <40%	0.7550	-	0.9325
EBAM internal RH <40% and EBAM hourly values of negative replaced with zero	0.7663	0.8230	0.9334

[a] Probability (p) < 0.006 for all R^2 values.

Mean Difference Agreement

Internal RH had the largest effect on both hourly and daily agreement. Mean difference calculations (EBAM-BAM) all showed a positive bias of ~3 $\mu g\ m^{-3}$ with standard deviation of the differences decreasing when higher EBAM internal humidity was removed (Table 2.5). Upper and lower limits of agreement for hourly values were closest when EBAM internal RH was below 40% and zero was used in place of negative values. When comparing hourly BAM readings to mid-point 3 hour mean concentrations of the EBAM, the EBAM over-predicted the BAM. Replacing negative EBAM hourly values with zero and removing hourly EBAM values with internal RH below 40% showed an increase in agreement. The use of 3 hour mean concentrations helped to reduce the mean difference between monitors. Best agreement for this data was with daily mean concentrations where hourly EBAM internal RH <40% was removed (Table 2.5).

Table 2.5 Mean difference and limits of agreement statistics with upper and lower limits of environmental beta attenuation monitor (EBAM) to beta attenuation monitor (BAM) comparison.

	Mean Difference	Standard Deviation	Number of samples	Lower limit	Upper limit
All hourly data	2.8	10.8	8972	-18.8	24.4
Negative EBAM hourly values replaced with zero	3.1	10.4	8972	-17.8	24.0
EBAM internal RH <40%	2.4	8.5	6293	-14.6	19.4
EBAM internal RH <40% and EBAM hourly values negative replaced with zero	2.6	8.2	6293	-13.8	19.0
3-hour mean	3.2	9	8976	-14.7	21.1
3-hour mean with EBAM internal RH <40% and EBAM hourly values negative replaced with zero	2.8	6.7	6295	-10.6	16.1
Daily mean	2.8	5.3	367	-7.8	13.5
Daily mean negative EBAM hourly values replaced with zero	3.2	5.2	367	-7.2	13.5
Daily mean with EBAM internal RH <40% and EBAM hourly values negative replaced with zero	3.1	3.2	193	-3.4	9.6

As ambient humidity increased above 65%, hourly mean difference and standard deviation increased (Figure 2.6a). Mean difference with external humidity between 55% and 60% was 1.7 µg m^{-3} with a standard deviation of 8.6 µg m^{-3}. When external humidity increased to 60-65%, mean difference and standard deviation increased to 2.4 µg m^{-3} and 10.9 µg m^{-3} respectively. Mean difference and standard deviation were highest when external humidity was 90-95% with a mean difference of 6.9 µg m^{-3} and standard deviation of 18.0 µg m^{-3}. EBAM hourly values were higher throughout all levels of external RH except when ambient RH was 95-100% where the EBAM hourly value was less than the BAM (Figure 2.6a). Although the EBAM typically is over-estimating the BAM, when ambient RH is near 100% the EBAM can underrepresent BAM readings representing the complex impacts of temperature, RH, and inlet heating on PM measurements particularly when one monitor is operating at or near ambient conditions (EBAM) and the other is in an enclosed climate controlled environment (BAM).

Increased ambient RH is in part corrected by the inlet heater controlled internal RH of both the BAM and EBAM. This internal RH, when operating at levels < 45%, had an hourly mean difference typically between 2 and 5 µg m^{-3}. When internal RH was below 5% the BAM was over predicting the EBAM (mean difference -4.4 µg m^{-3}) although the smallest number of hourly samples (28) fell into this group. Mean difference was 2-3 µg m^{-3} when internal RH was 5-

40%, increased to 5 µg m^{-3} at 40-45%, fell to -4 µg m^{-3} between 55-60% then increased to 5 µg m^{-3} at 55-60% then increasing to a maximum of 35 µg m^{-3} when internal RH was 70-75%. Standard deviation of the mean difference was ~8-9 µg m^{-3} when the internal RH was below 40% (6 µg m^{-3} at 0-5%; 9 µg m^{-3} at 5-15% and 30-40%; 8 µg m^{-3} at 15-30%) then increased to 13 µg m^{-3} for 40-45%, 7 µg m^{-3} for 55-60%, and 14 µg m^{-3} at 60% and higher (Figure 2.6b). When internal RH was kept below 40% agreement between the BAM and EBAM was the most consistent.

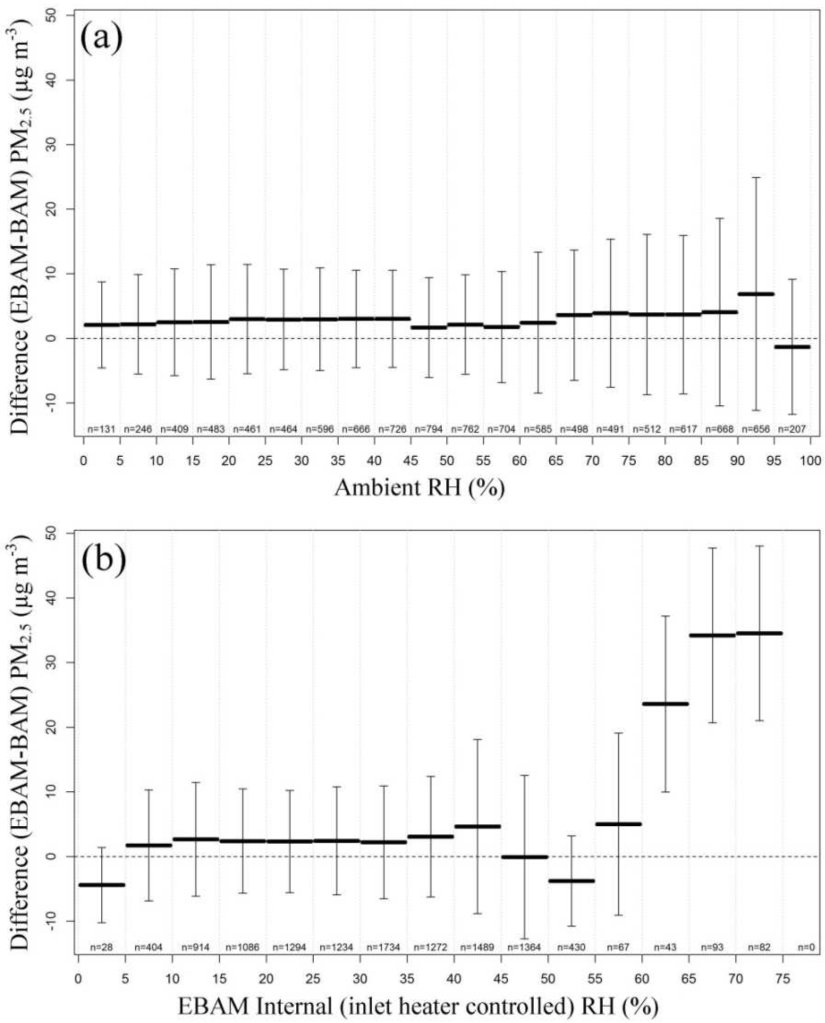


Figure 2.6 Agreement using difference from the mean (+/- 2 standard deviation error bars) of environmental beta attenuation monitor (EBAM) and beta attenuation monitor (BAM) with (a) similar external relative humidity (RH) and (b) similar internal (inlet heater controlled) RH. RH levels have a range of +/- 2.5% with n number of hourly values.

Relative humidity was the cause of the largest discrepancies between EBAM and BAM measurements. High ambient RH (>90%) produced the largest differences in measurements and standard deviations. Inlet heating of the EBAM to control internal RH reduced high ambient RH impacts. When internal RH was below 40%, EBAM measurements were most consistent when comparing to the BAM (Table 2.5).

AQI Agreement

Assessing AQI for BAM to EBAM showed an overall increase in AQI when using data from an EBAM. None of the hourly measurements, 3 hour mean, or daily mean was above an AQI of unhealthy for sensitive groups. Removing hourly values when the internal RH of the EBAM met or exceeded 40% reduced the disparity between AQI category estimations but also resulted in the loss of roughly a quarter to a third of all measurements.

AQI for hourly BAM to EBAM (when both the BAM and EBAM recorded an hourly reading) was typically categorized good (8 862 (98.8%) BAM; 8 502 (94.8%) EBAM) with 109 (1.2%) BAM to 460 (5.1%) EBAM moderate and 1 (<0.1%) BAM to 10 (0.1%) EBAM unhealthy for sensitive groups. AQI hour counts were 6 215 (98.8%) BAM to 6 160 (97.9%) EBAM good, 77 (1.2%) BAM to 128 (2.0%) EBAM moderate, and 1 (<0.1%) BAM to 5 (0.1%) EBAM unhealthy for sensitive groups.

Using 3 hour mean concentrations in comparison resulted in 8 986 (98.8%) BAM to 8 676 (95.5%) EBAM good, 96 (1.2%) BAM to 403 (4.5%) EBAM moderate, and 0 (0.0%) BAM to 3 (<0.1%) EBAM unhealthy for sensitive groups values. When internal RH was below 40%, AQI estimates were more consistent (good 6 329 (98.8%) BAM, 6 293 (98.3%) EBAM; moderate 72 (1.2%) BAM, 108 (1.7%) EBAM) with 0 (0.0%) BAM and 1 (<0.1%) EBAM unhealthy for sensitive groups.

Daily AQI comparison resulted in 203 (55.3%) BAM to 138 (39.5%) EBAM good, 164 (44.7%) BAM to 138 (59.1%) EBAM moderate, and 0 (0.0%) BAM to 5 (1.4%) EBAM unhealthy for sensitive groups. Controlling for internal RH (including only hourly data where the internal RH was below 40%) produced 136 (70.5%) BAM to 85 (44.0%) EBAM good days, 57 (29.5%) BAM to 108 (56.0%) EBAM moderate, and no unhealthy for sensitive groups. Removing these hours reduced the number of daily comparisons by 47% (367 to 193 days) and invalidated the highest daily AQI (unhealthy for sensitive groups) estimates which were only found with the EBAM data.

A plot of the distributions, using a normal (or Gaussian) kernel function with bandwidth 0.9 times the minimum of the standard deviation and the interquartile range divided by 1.34 times the sample size to the negative 1/5 power (Silverman, 1986) for a kernel density estimation as a non-parametric representation of the density of the hourly $PM_{2.5}$ variable, helped us to visualize the increase in hourly measurements when comparing an EBAM to a BAM (Figure 2.7). The overall hourly EBAM distribution is noticeably higher than the BAM. The impact of setting negative hourly EBAM values to zero illustrates the shift in values. Including only EBAM hourly values where the internal RH is less than 40% produces a distribution where the highest hourly values (> ~30 $\mu g\ m^{-3}$) run closer to the BAM.

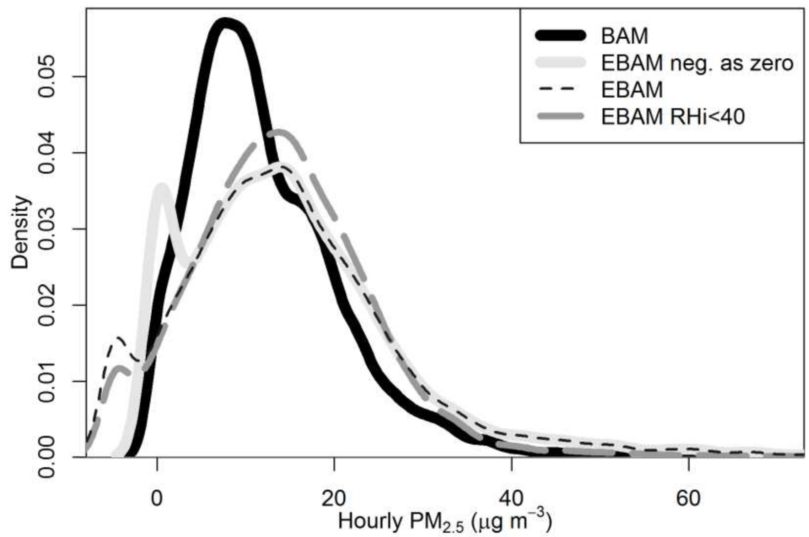


Figure 2.7 Distribution of hourly fine particulate matter ($PM_{2.5}$) using kernel density estimation.

Discussion

Temporary mobile particulate monitors are being widely utilized for smoke monitoring throughout California. The EBAM is an excellent source allowing for mobile, temporary monitoring of $PM_{2.5}$. The EBAM has been widely deployed during emergency events and provides an indispensable source of additional data for public health protection that otherwise would not be available. Analysis in this paper is intended to document and understand limitations of using the EBAM; in particular when being used for smoke management decisions through direct comparison to monitors used for federal and state compliance.

The BAM is widely used by California air regulators as the “gold” standard for compliance monitoring, thus the EBAM (or any other monitor) being used for smoke monitoring and management has a default assumption of comparability. EBAMs are particularly useful when monitoring large, high intensity, full suppression wildfires when “conservative” estimates allow public health officials to error on the side of caution and indeed our analysis suggests that using EBAMs is an excellent way to provide additional targeted data and over-represent ground concentrations of $PM_{2.5}$ when compared to BAMs. This monitoring assumption can be useful when determining compliance and impacts from large full-suppression wildfires (Preisler et al., 2015) while consistent over-estimation may be hindering a more nuanced policy approach to smoke management desperately needed for effective wildland fire management in the western United States (North et al., 2015a). Wildland fire being used for fuel reduction and ecological benefit through prescribed and natural ignition often localizes smoke impacts heavily dependent on non-regulatory temporary monitors (Schweizer and Cisneros, 2014) where even slight increases in the estimated background concentrations can determine fire management actions and perpetuate entrenched disincentives for full suppression (North et al., 2015a). While this policy debate involving fuel loading, fire ecology, and smoke management continues (Boer et al., 2015; North et al., 2015b; Thompson et al., 2015), understanding smoke monitoring equipment strengths and weaknesses will help inform the discussion.

The supplemental data being generated by EBAMs provides an invaluable addition to the urban FEM monitoring network during a wildland fire. Current advisories and assessment of smoke impacts on public health at many times are reliant on the EBAM. Although these instruments are providing data in areas where it is not feasible to establish a permanent monitor, our data suggests the EBAM is over representing the exposure levels and is of particular concern at lower concentrations where EBAM measurements are being relied on for smoke management actions. This is particularly apparent when higher humidity and more stagnant air masses may be combining to introduce error into the EBAM hourly measurements.

RH is an important meteorological component to wildland fire dynamics with increasing RH reducing burn severity (Collins et al., 2007) and often aiding containment of extreme wildfires. Although high humidity may not be present in all wildland fire events, evening and overnight RH can exceed 40% (when BAMs and EBAMs begin humidity control heating) even during large high-intensity wildfires (Peterson et al., 2015). Higher RH is often typical at the end of a managed fire where emissions are primarily generated from interior smoldering during burn down. RH is also a component to a prescribed fire plan and is typically stipulated for levels that both moderate fire behavior and provide desired fuel consumption. Ambient RH can be above 40% for much of a prescribed fire (Knapp et al., 2005) with lower RH being specified for the desired burn intensity and higher RH utilized as a controlling mechanism. RH can frequently exceed the 40% internal RH set point for the heater controlled sample of the BAM and EBAM during a wildland fire.

High RH can be a particular concern at the end of a full-suppression or managed natural ignition wildland fire when meteorological conditions can include high humidity and precipitation events. Error introduced from high RH almost solely manifests as a higher hourly $PM_{2.5}$ readings on the EBAM. Using this data can lead to incorrectly including high values in data analysis of smoke impacts. Although an erroneously high measurement may be considered conservative in the protection of human health, and indeed be helpful to punctuate the seriousness of an extreme air quality event, the application of consistently over-estimating $PM_{2.5}$ in a fire adapted ecosystem where smoke is inevitable can easily create unintended consequences at lower concentrations where misrepresentation of ground level $PM_{2.5}$ results in suppression biased smoke management actions.

Over-estimation of smoke impacts to $PM_{2.5}$ for the EBAM in large part can be remedied by both using 24 hour average concentrations while assessing impacts and exposure of a given population and giving extra consideration to internal RH of the EBAM before including individual hourly values in the calculation. Hourly AQI consistency between monitors was increased simply when we only included hourly data when the internal RH was <40% and improved further with 3 hour averaging.

It is important to recognize limitations and uncertainties when comparing fine particulate monitors. $PM_{2.5}$ measurements included in this study reflect the accuracy and precision expected when temporary monitors are used in the field to measure $PM_{2.5}$ at lower concentrations (hourly concentration range 0-102 $\mu g\ m^{-3}$). With agreement encompassing a range that includes about half the hourly data points and a quarter of the range of daily values, the possibility exists that at high concentrations of $PM_{2.5}$ the effect is similarly large and would encompass a wider concentration disparity which could lead to even greater differences between instruments when determining AQI for high levels of smoke.

Data presented in this study agree with the documented effects from humidity on PM mass measurements (Heber et al., 2006; Tsyro, 2005). For best agreement between monitors, heater controlled internal RH should be held below 40% and special consideration should be given when ambient RH rises above 65% with any readings where the ambient RH is above 90% being used with caution as the high external RH is likely reducing the effectiveness of the internal heater manifesting in the EBAM over-estimating $PM_{2.5}$ concentrations. When the RH remains above the EBAM inlet heater set point for extended periods of time this effect seemed more pronounced.

The hourly concentrations in this data are relatively low (maximum BAM 93 $\mu g\ m^{-3}$, EBAM 102 $\mu g\ m^{-3}$) with good AQI. Our data shows instrument selection and RH have an impact on calculated AQI for $PM_{2.5}$. AQI estimates during hours of high humidity typically were higher for an EBAM than a BAM giving rise to erroneously high estimates from areas relying on data from an EBAM. Concentrations of $PM_{2.5}$ are frequently much higher during incidents with large emissions from full suppression wildfire. More study is needed to compare particulate monitors at higher concentrations to ensure AQI is being adequately reflected between temporary and permanent monitors.

While EBAMs provide useful information to help determine specific hour or hours during a day when air quality poses the largest threat to human health, the hourly data from EBAMs should be used with caution when comparing to BAM measurements. Although inlet heating to control relative humidity can minimize this effect (Huang, 2007), EBAM inlet heaters did not always keep internal RH below 40%. Invalidating hourly EBAM data with high internal RH hours when calculating daily averages helped increase agreement with BAM measurements. Additionally, we saw an over-estimation of AQI when EBAMs are compared to BAMs. This is particularly true using hourly measurements for AQI. We recommend using hourly (or 3 hour mean) with extreme caution when providing near real time public health advice. Daily AQI is a much more appropriate matrix when using EBAMs for assessing air quality for $PM_{2.5}$.

Conclusions

Mobile monitors such as EBAMs are an incredibly useful source of data for research and public health protection during short duration events. The increased use of mobile monitors during temporary events such as wildfire makes understanding the limitations of mobile monitors important to providing accurate information to the public.

The EBAM is susceptible to over-estimation of $PM_{2.5}$ concentrations when ambient RH is high. Agreement between BAMs and EBAMs are weakest when internal RH is above 40% with EBAMs typically over-estimating $PM_{2.5}$ concentrations that additionally result in elevated EBAM estimations of AQI category. Co-located monitoring during wildland fire smoke events would be useful in determining the presence and extent of this impact at higher $PM_{2.5}$ concentrations than were presented in this study.

The co-location comparisons in this study suggest EBAM hourly measurements are not precise enough to warrant comparison to an EPA certified BAM site. Additionally, EBAM hourly AQI estimates should be used with extreme caution as hourly (and 3 hour mean concentrations) often result in over-prediction even at the lower concentrations included in this study. Thus, it is recommended to use only daily AQI estimates when using the EBAM. EBAM daily mean concentrations calculated using only hourly $PM_{2.5}$ concentrations when the internal RH is below

40% are the most appropriate measurements to use when comparing $PM_{2.5}$ concentrations and AQI estimates to BAM data.

Chapter 3.0 Smoke impacts from wildland fire managed for ecological benefit

Wildland fire smoke monitoring is increasing in sophistication. Incorporating all forms of monitoring and modelling is essential to understanding smoke impacts (Kochanski et al., 2015). Ground based monitors are essential to understanding human health impacts by quantifying exposure but are limited spatially. While widespread monitoring occurs in the more urban areas, $PM_{2.5}$ monitoring for wildland fire smoke in the Sierra Nevada relies heavily on data from mobile EBAMs. There are some limitations using the temporary EBAM to monitor $PM_{2.5}$ from wildland fire smoke. Limitations can be reduced by including only data when the internal RH below 40% and ensuring data when ambient RH is high.

The Sheep Fire (2010) which started in Kings Canyon National Park helped to improve deployment. EBAMs deployed on the Sheep Fire focused primarily on the west side (Figure 3.1).

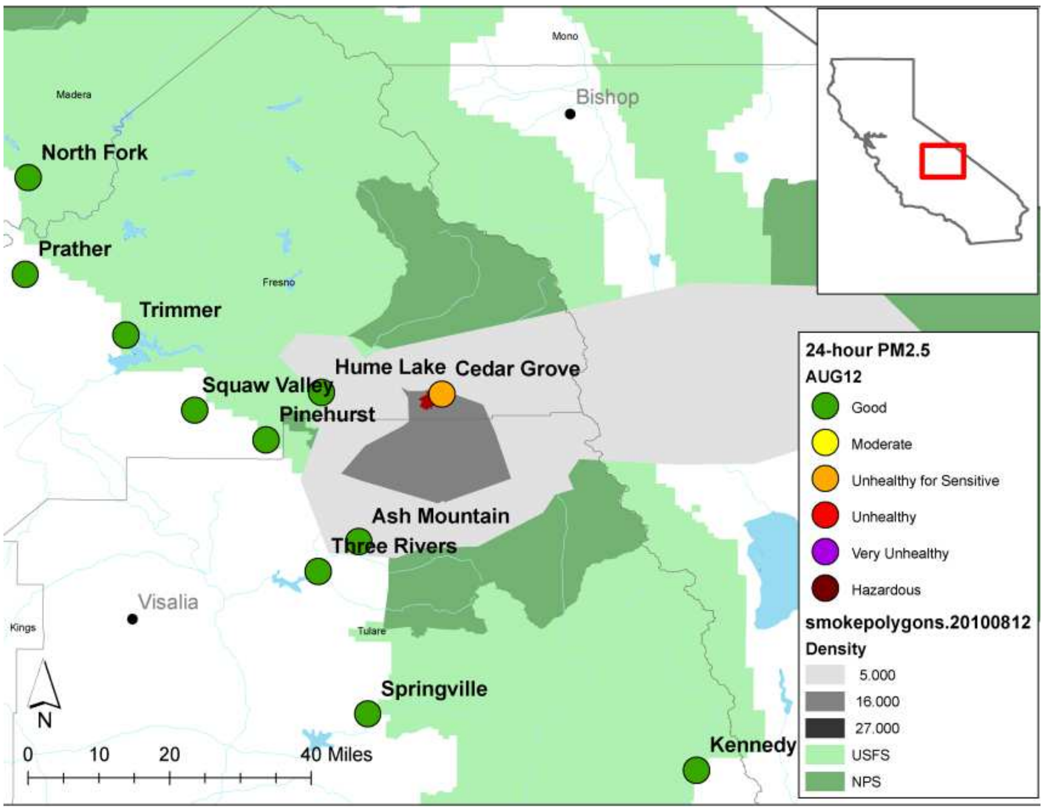


Figure 3.1 Sheep Fire (2010) monitoring showing U.S. Forest Service and National Park Service monitoring sites, fine particulate matter ($PM_{2.5}$) air quality, and Hazard Mapping System Fire and Smoke Product smoke density polygons on August 12.

The following case study uses monitoring from an ecologically beneficial managed fire. EBAMs were placed around the fire and at locations where transport was expected to supplement BAMs. This fire was used because of the extensive monitoring and the ability to strategically place monitors at the earliest stages of the fire. These units remained in place after the fire to give a better indication of individual site background levels of $PM_{2.5}$. This managed fire was used because of the more strategic deployment of equipment allowed for a more thorough analysis.

Chapter 3.1 Wildland fire management and air quality in the southern Sierra Nevada: using the Lion Fire as a case study with a multi-year perspective on $PM_{2.5}$ impacts and fire policy

Reprinted from the Journal of Environmental Management, 144, Don Schweizer and Ricardo Cisneros, Wildland fire management and air quality in the southern Sierra Nevada: using the Lion Fire as a case study with a multi-year perspective on $PM_{2.5}$ impacts and fire policy, pages 265-278, Copyright (2014), with permission from Elsevier.
DOI: 10.1016/j.jenvman.2014.06.007

Abstract

Management of fire is an important and controversial policy issue. Active fire suppression has led to a backlog of fuels, limited the ecological benefits of fire, and reduced short-term smoke impacts likely delaying these emissions to future generations over a larger spatial extent. Smoke impacts can be expected to increase as fire size and intensity increase and the fuel backlog is consumed; whether through reintroduction of fire under desirable conditions or through stand replacing fire. Land Management Agencies would like to increase the use of naturally ignited fires to burn during favorable conditions as a way to reduce catastrophic fires. This study provides information about the levels of air quality impacts expected from these types of fires and discusses some of the policy controversies of managed fire that propagate inconsistencies between agencies and enter the public discourse. The Lion Fire, a primarily low intensity 8,370 hectare fire that was extensively monitored for Particulate Matter less than 2.5 microns ($PM_{2.5}$), is used to quantify impacts to air quality. $PM_{2.5}$ monitoring sites are used to assess exposure, public health impacts, and subsequently quantify annual air quality during a year with a fire that is within the historic normal fire size and intensity for this area. Ground level $PM_{2.5}$ impacts were found to be localized with 99% of the hourly Air Quality Index readings in the moderate or good category for the sites impacted by the fire. $PM_{2.5}$concentrations at sites nearest the fire were below annual federal air quality standards for $PM_{2.5}$ with annual 98th percentile at the most impacted sites (Johnsondale, Kernville, and Camp Nelson) of 35.0, 34.0, and 28.0 μgm^{-3} respectively. Smoke impacts to $PM_{2.5}$ concentrations were not found to reach the populated Central Valley. The findings suggest that this type of fire can be implemented with minimal public health impacts thus allowing an opportunity for air and fire managers to alter policy to allow additional burning in an area with severe anthropogenic air pollution and where frequent widespread fire is both beneficial and inevitable. The more extensive air quality impacts documented with large high intensity fire may be averted by embracing the use of fire to prevent unwanted high intensity burns. A widespread increase in the use of fire for ecological benefit may provide the resiliency needed in Sierra Nevada forests as well as be the most beneficial to public health through the reduction of single dose exposure to smoke and limiting impacts spatially.

Highlights:

- Smoke impacted sites $PM_{2.5}$ was below federal annual and 98th percentile thresholds.
- $PM_{2.5}$ AQI in monitoring locations were typically good or moderate during the fire.
- Smoke impacts were primarily localized.
- Smoke impacts from a low intensity managed fire can be largely mitigated.
- Allowing fires of this magnitude may reduce regional smoke impacts to air quality.

Introduction

Wildland fire management is an important subject for policy makers to address. It is necessary for some ecosystems to thrive but, depending on environmental and weather conditions, may cause detrimental impacts to air quality and to human health. However, some wildland fires and the subsequent smoke impacts to human health may be necessary to best mitigate the extreme air quality events from large high intensity fires. By proactively managing fire under desirable conditions, improved ecological health can provide a resilient and robust forest system in the Sierra Nevada that can help lessen public health impacts from anthropogenic emissions while minimizing air quality impacts from wildfire emissions.

Fire has a significant role in the formation and health of forests in the California Sierra Nevada (Kilgore, 1981). Before active suppression of wildfires, slow moving low intensity ground fires dominated this ecosystem providing an ecological pressure that shaped the fire-adapted and tolerant species native to this area. The natural process of frequent, low intensity burns is essential to maintaining healthy landscape level populations and overall forest health (Ferrenberg et al., 2006; Keeley et al., 2005; Moreira et al., 2010; Parmeter, J.R. and Uhrenholdt, 1975; Swetnam et al., 2009; Zimmerman and Laven, 1975). Fire adaptation and tolerance are necessary for the continued health of this ecosystem particularly in the face of climate change (Stephens et al., 2010).

Wildfire activity has increased with longer wildfire seasons coupled with larger size and longer duration wildfires (Westerling et al., 2006) since the radical suppression tactics utilized over the last century and a half. High severity stand-replacing fire has been increasing in the Sierra Nevada since the mid-1980s, with mean and maximum fire size and annual area burned at or above those before the national policy of suppression was instituted (Miller et al., 2009). Total burned areas in recent years, which currently are considered “extreme fire years,” are typical of historically normal burn years (Stephens et al., 2007). Although the burned area is similar, emissions of current fires can be greater because of excessive fuel loading from past fire suppression and increased fire intensity caused by warmer and dryer conditions.

Fire policy by Land Management Agencies (LMAs) has been changing over the last two decades. Past suppression policies are now recognized as one important factor leading to the catastrophic and unnatural forest fires presently occurring. Prescribed Burning (PB) started to be implemented in federally protected lands in California in an attempt to re-introduce fire as a natural process in the forest. In California, the National Park Service started implementing PBs in the 1960s and the U.S. Forest Service started in the 1990s. After years of PB, it is apparent that the small scale burning typical of PB (<200 hectares) will not lead to the landscape level restoration of fire sought by LMAs. Thus LMAs are proposing wider implementation of managed fire (MF). A MF is typically started with a natural ignition (lightning) and of larger size than PB. MFs utilize smoke dispersal, meteorological, and fuel conditions which allow for the safest implementation. MF is different from fires that use full suppression tactics in that larger areas are allowed to burn, when there is no threat to life or property, to maximize the beneficial effects of fire while reducing fire cost and increasing firefighter safety.

The Lion Fire was a naturally ignited fire in California that was allowed to burn as a MF. LMAs suggested that this type of fire (MF) could be implemented successfully allowing for fire to return to the landscape without causing ecological damage and at the same time impacts to air quality can be minimized to the extent that air pollutants in the most smoke impacted areas are below federal thresholds.

The Lion Fire is representative of a fire the size and intensity which historically occurred and is needed to restore and sustain the natural role of fire in the Sierra Nevada (Beaty and Taylor, 2008). Historically fire in this forest type burned on average between 11-40 years (van de Water and Safford, 2011), but this area had not burned in approximately 90 years, therefore fuel loads were greater producing larger emissions that would not have occurred with the historically lower fuel loads from more frequent fire.

Wildland Fire, Smoke, and Ambient Air Quality

Burned area alone is not indicative of emissions from wildland fire. In addition to fire size, fuel loading, fire intensity, and fuel consumption at a minimum need to be understood to accurately assess emissions. Emissions from a wildland fire are not indicative of ground level concentrations of Particulate Matter less than 2.5 microns ($PM_{2.5}$) and are difficult to predict (Yao and Henderson, 2013) even during large fires. Ground level concentrations and the subsequent impacts to air quality and human health are a product of emissions, plume height, transport distance, dispersal, and a suite of meteorological parameters. Background levels of pollutants in a fire-adapted ecosystem must also account for the reduction of smoke during an era of suppression where background levels are artificially reduced or in essence delayed until fuels are consumed. Lack of fire as has been typical during the era of fire suppression can be taken as artificially reducing smoke impacts to local air quality while providing a backlog of future emissions. Fire suppression in the Sierra Nevada which predates the beginnings of air quality monitoring for public health has likely led to an unsustainable expectation that background air quality in the Sierra Nevada is primarily smoke free. Burned areas during times now considered extreme fire years are potentially more indicative of the areas burned before fire suppression was implemented in California, although the overall impacts to ground level air quality may be higher than a natural background from the increased fuel loading and climate change (Hurteau et al., 2014). The backlog of fuels created through years of fire suppression has likely created an emissions deficit that will be confronted in the near future by an increasing population. Thus it is imperative to address the issue of smoke and public health with proactive policy that considers the dilemma of withholding smoke emissions that are to be saddled on the public in the future.

Mortality impacts attributed to smoke from wildland fire are only recently beginning to be understood. Sastry (2002) only consistently found a mortality impact at high particulate matter levels (PM_{10} above 210 μgm^{-3}). Lower levels ($PM_{2.5}$ level below 48 μgm^{-3}) were not found to create a significant mortality impact (Vedal and Dutton, 2006). Kochi et al., (2012) found a threshold effect for mortality in densely populated areas of San Bernardino County during the large wildfires of 2003 in southern California (PM_{10} levels over 360 μgm^{-3} and $PM_{2.5}$ levels over 100 μgm^{-3}) but did not find significant mortality impacts in less densely populated areas with similar levels or in densely populated areas with milder levels. Johnston et al., (2013) reported that decreased air pollution from biomass smoke was associated with reductions in mortality. Smoke management in a fire adapted ecosystem must incorporate both the immediate and long term smoke impacts to public health including the spatial scale of smoke impacts under different fire management scenarios.

Public Perception of Smoke

Without an understanding of smoke impacts from altering the fire regime, a strong incentive exists to suppress and delay emissions to the future. This is in part due to public perception of fire and fire management being complex and belief based (Bright et al., 2007) likely reflecting the complexity of fire, evolving fire management techniques (Brown et al., 2004; Dellasala et al., 2004), and public perceptions (Gauchat, 2012) and awareness (Murphy et al., 2007) in particular when attempting to understand the role of fire for ecologic and human health. Public health officials and the general public are biased to offsetting smoke emissions to some future date to limit the pollutants today. This is especially true in areas of high anthropogenic pollution and where smoke impacts directly impact the local economy. This creates a dilemma for fire managers where it is easier to suppress fire for the immediate benefits than to actively manage for forest health. Healthy relationships between stakeholders are integral to healthy forests and public understanding of smoke from wildland fires (Champ et al., 2012).

Compounding public perception of risk from smoke is the high visibility of smoke from wildland fires in the Sierra Nevada. The Sierra Nevada rise from the California Central Valley, perching Sierra Nevada wildland fires above major urban areas. Plumes from wildland fires in the Sierra Nevada are almost always visible from at least one urban area in the Central Valley. Additionally, prevailing winds typically move west to east dispersing smoke over the Owens Valley frequently reducing visibility in this tourist based economy. Even though smoke from wildland fire is innately a part of the Sierra Nevada and as essential as floods, blizzards, wind storms and other natural processes, agency and public perceptions, beliefs, and attitudes are variable (Steelman and McCaffrey, 2011) and include a belief that smoke free air in the mountains of the Sierra Nevada is a standard condition instead of an anthropogenic bi-product of fire suppression that is not sustainable.

Fire Policy as Perceived by Land Management Agencies in the Sierra Nevada

Large areas of the Sierra Nevada are managed by federal agencies. The U.S. Department of the Interior National Park Service (NPS) and the U.S. Department of Agriculture Forest Service (USFS) are the primary land management agencies in the Sierra Nevada west of the crest. Although these two agencies are quite often confused by the general public, their founding missions are fundamentally different. The NPS is a conservation organization where anthropogenic impacts are eliminated to the greatest extent possible while the USFS allows for regulated sustainable use of the forest ecosystem. On USFS managed land, human activities such as logging, hunting, mountain biking, motorized vehicle use, etc. are typically less restricted than on NPS managed land.

Both the USFS and the NPS manage designated Wilderness Areas. The Wilderness Act is the legislation intended to guide management of these areas for federal land managers (FLMs). The Wilderness Act restricts the development of these areas and requires FLMs to protect and preserve the natural conditions. FLMs are largely restricted from using invasive techniques such as mechanical thinning in Wilderness Areas. The goal is to have a naturally functioning ecosystem to provide the public with a place of solitude connected to the natural world, a connection with the historic value of the American wilderness, and an area set aside for the conservation of plant and animal species for all to enjoy. Natural areas also have the capacity to enhance air and water quality though restricting development and allowing the natural system to act as a pollution sink and natural buffer to anthropogenic impacts (Fulé, 2008; Hurteau and North, 2009, 2010; Hurteau et al., 2008).

Nowhere do these lofty goals come in more conflict than smoke management from a natural ignition wildland fire in a Wilderness Area. Not only is it well understood that fire plays an essential role in the health of the Sierra Nevada (Beaty and Taylor, 2008; Nesmith et al., 2011) but it is also understood that wildland fire is necessary to reduce fuels that have accumulated from past fire suppression policies to a degree which threaten the ecological integrity of these forests (Reinhardt et al., 2008) while attempting to mitigate potential increases in fire activity from climate change (Liu et al., 2010; Spracklen et al., 2009). But, FLMs are not only regulated by the Wilderness Act, they are also regulated by a myriad of other laws, acts, and policies including the Clean Air Act (CAA, 2004) and the Regional Haze rule (Icr, 2007). The public, public health officials, representatives of tourist based economies, and smoke sensitive residents pressure LMAs to fully suppress these fires.

Land management policy objectives and the current understanding of ecological benefits provide clear direction to manage these fires on the landscape while air regulation policy through the Clean Air Act (CAA, 2004), Regional Haze Rule (Icr, 2007), etc. attempt to restrict anthropogenic emissions. Economic interests compete for these emission thresholds and the overall capacity of an air shed to disperse and buffer pollutants. Wildland fire emissions to the air shed, even though a natural process emission, are typically managed as an anthropogenic emission. This provides conflicting policy and direction where regulator coordination has been difficult (Arbaugh et al., 2008) and public opinion is heavily weighted through the use of complaint programs. Additionally through policies such as the Exceptional Events Rule (Exceptional Events Rule, 2007), air regulators have more latitude for compliance with air quality standards when natural events can be determined to have caused air quality violations. Although in practice this has been complicated by how to document the contribution from a wildfire and the interpretation of what is a natural event wildfire (California Title 17 (CCR, 2016) considers a natural ignition fire as a PB). Land managers are thus in the conundrum of mitigating smoke impacts during a natural ignition fire with little public support or suppressing all fire which withholds emissions until a MF with full suppression occurs where air quality impacts are marginalized by the more immediate concerns of loss of life and property from the fire itself.

National and California Ambient Air Quality Standards and Smoke Management

Air quality in the U.S.A. is regulated through a multifaceted approach using Federal, State, and local laws to assure compliance with clean air requirements established by the CAA.

The CAA was passed in 1963 and establishes the regulatory framework for air pollution prevention and control in the United States of America. The primary goal of the CAA is pollution prevention through "…reasonable Federal, State, and local governmental actions" (CAA, 2004). The CAA (CAA, 2004) was established to "protect and enhance" the air resources of the United States, expand and improve research of air pollution, provide national assistance to State and local governments for air pollution control and prevention, and to foster regional air pollution control and prevention programs. The U.S. Congress, recognizing that the U.S. population was increasingly located in expanding urban areas and "that the growth in the amount and complexity of air pollution brought about by urbanization, industrial development, and the increasing use of motor vehicles, has resulted in mounting dangers to public health and welfare" (CAA, 2004), found "…Federal financial assistance and leadership is essential for the development of cooperative Federal, State, regional, and local programs to prevent and control air pollution." (CAA, 2004) The primary responsibility of prevention, reduction, and elimination of air pollution at its source lies with the States and local governments. Pollution prevention is expected with cooperation of all levels of government through Federal, State, and local laws. Federal standards provide basic requirements which State and local law can make more stringent. Cooperation has not fulfilled this standard. Because this cooperation is essentially based in regulating anthropogenic pollutants natural sources of pollutants are not willingly accepted into a polluted air district. When emissions from economically advantageous industries are by necessity being regulated, natural emissions are an easy target to eliminate. There is no incentive in the CAA to allow for natural source emissions providing a difficultly in allowing wildland fire.

The CAA establishes a framework of uniform laws with federal enforcement. In addition to regulating air pollutants, the CAA includes guidance on visibility, prevention of significant deterioration of air quality and visibility, in wilderness areas, emissions and fuel standards, noise pollution, and stratospheric ozone protection. Compliance to the CAA is primarily administered by the U.S. Environmental Protection Agency (EPA). The EPA establishes National Ambient Air Quality Standards (NAAQS) as benchmark levels of criteria pollutants and ensures compliance with these standards. Areas below the NAAQS are considered "attainment areas" while areas above are "nonattainment areas". Meeting the NAAQS is delegated to the state. Individual states are granted primary responsibility for assuring air quality by the establishment of a State Implementation Plan (SIP). The SIP specifies how the state will achieve the national primary and secondary ambient air quality standards.

A 1977 amendment to the CAA established regulatory protection for visibility in wilderness and other natural, scenic, and historic areas. Wilderness areas managed by federal land management agencies are now provided addition protection with a "prevention of significant deterioration of air quality" (CAA, 2004) from additional anthropogenic point source emissions (i.e. power plants).

For human health protection, the EPA has established primary standards for 6 criteria pollutants (Particulate Matter (PM_{10}, $PM_{2.5}$), Ozone (O_3), Sulfur Dioxide (SO_2), Nitrogen Dioxide (NO_2), Carbon Monoxide (CO), and Lead (Pb)). The California Clean Air Act (1988) established additional more stringent standards for these criteria pollutants and also established standards for sulfates, hydrogen sulfide, vinyl chloride, and visibility reducing particles with the California Ambient Air Quality Standards (CAAQS). Areas in compliance with the standards are considered "attainment" areas, while those areas not in compliance are considered "non-attainment" areas.

In California, the California Air Resources Board (CARB) is the responsible state agency. CARB further delegates this responsibility to local Air Pollution Control Districts or Air Quality Management Districts. The Districts have the primary responsibility for meeting the requirements of the CAA (Figure 3.2). Emissions from both MF and prescribed fire are considered air pollutants with regulatory oversight by CARB and the Districts. This causes a dilemma for a fire like the Lion Fire to be implemented as a MF as it is perceived as a PB and there is no incentive for an air district to accept additional emissions from fire. Additional emissions create disincentives including nuisance complaints and possible enhancement of standard violations in a non-attainment area to air regulators. Additionally, public health officials necessarily will discourage smoke emissions as any smoke subjects the public to exposure which will inevitably lead to public health issues particularly for the young, elderly, and other sensitive groups. In an anthropogenic polluted air basin it is particularly difficult to confront some smoke today when it can be postponed to some undetermined future date.

Emissions from wildland fires impact air quality and contribute to air pollutant concentrations (Langmann et al., 2009).Smoke from wildland fires impact visibility locally and regionally (McMeeking et al., 2006). Anthropogenic emissions in the California create widespread air quality impacts with approximately 28% of the land area of California and 26 million (70%) of its residents living in areas designated as nonattainment of the federal standards for $PM_{2.5}$. This includes the California Central Valley which is one of the most polluted air sheds in the world. $PM_{2.5}$ is a significant problem for air quality in this area and is a component of smoke generated from fires. This has led the Air regulatory agencies to be even more stringent when it comes to putting fire back on the landscape.

Exposure to smoke has adverse impacts to human health (Kochi et al., 2010). Large uncontrolled wildland fires on public lands can have significant impacts on air quality in urban areas (Viswanathan et al., 2006) with an increased exposure to smoke causing an increase risk to human health (Künzli et al., 2006; Tham et al., 2009).Current health research also underlines the impacts of smoke at low doses and differing socio-economic factors (Rappold et al., 2012) which will undoubtedly yield a better understanding of smoke mitigation in the ensuing years that will provide air and fire managers better insight into smoke impacts on all communities from multiple fire scenarios using PB, MF, and full suppression. Understanding impacts from wildland smoke are further confounded by location in California where chronic $PM_{2.5}$ exposure is highly dependent on location. Exposure to high concentrations of $PM_{2.5}$is typical in the Central Valley while concentrations typically are much lower as elevation increases and population density decreases (Cisneros et al., 2014). Excess mortality from wildland smoke is similar to general estimates for urban PM (Hänninen et al., 2009) suggesting federally developed health standards would accurately represent relative risk for a given population.

Competing Priorities in Smoke Management: The Balancing Act

Federal land managers and air regulators attempt to balance competing priorities between fire and air quality. Advances in wildfire simulation and predictive models are helping to frame federal wildland fire policy and operations by implementing a risk management framework to support the decision making process (Calkin et al., 2011a, 2011b; Vadrevu et al., 2010) but there is an inherent unpredictability in wildfire behavior and uncertainty in prioritizing value to ecological and human health impacts (Brugnach et al., 2011; Thompson and Calkin, 2011).This again leads to air policy through the CAA being invoked to discourage MF because the risk of a larger than predicted smoke event can essentially stop an active fire management program for many years.

Smoke transport and the unpredictability of wildland fire and the subsequent emissions make MF very difficult to conduct because of smoke management. Smoke impacts to air quality are dependent on weather (i.e. wind speed and direction), seasonal timing, and emissions (i.e. fuel loads, fire intensity) (Tian et al., 2008). There is an opportunity to use MF to better control these emissions and potentially mitigate some smoke impacts to public health with better control of the timing and intensity of emissions. Large uncontrolled fires with burn intensities greater than the historic normal will produce major air quality impacts while fires burning within normal fire intensity in the Sierra Nevada have less impact over a smaller area. Increasing MF ideally would reduce the spatial extent of smoke impacts by controlling fire intensity and size but would also likely increase localized exposure to some smoke as more fires in a given area are allowed to burn.

This manuscript will test the hypothesis that the Lion Fire did not significantly impact air quality and provide insight and information about the conflicting policies connected to this case study. Thus data presented in this paper are assumed to be representative of a conservative estimate of ground level concentrations of $PM_{2.5}$ from a wildland fire in the southern Sierra Nevada and representative of other areas where fire suppression has left an unnatural fuel load. Smoke impacts to public health are evaluated during the event and analyzed to understand the significance of a fire this size on local and regional air quality. This study uses air quality concentrations of $PM_{2.5}$, one of the criteria pollutants under the National Ambient Air Quality Standards (NAAQS), collected during the fire to identify impacts. Current Federal fire policy with respect to public and ecological health is discussed to help balance the conflict between air and land management as it relates to wildland fire and ultimately to determine if smoke impacts to air quality are minimized successfully when allowing active use of wildland fire as a tool for land managers.

Material and methods

Fire Location and description

The Lion Fire started near Lion Meadow in the Golden Trout Wilderness on the Sequoia National Forest (36° 16' 5" N, 118° 30' 40" W) and primarily burned in the Golden Trout Wilderness of Sequoia National Forest. The fire burned a total of 8,370 hectares from 7/8/2011 to 9/7/2011. 7,920 hectares burned in the Sequoia National Forest and 450 hectares in Sequoia National Park (Figure 3.2). Fire information data for the Lion Fire and other wildland fire emission sources during this period were obtained from the National Interagency Fire Center (NIFC, 2016) additional local fire information was obtained from the Sierra Wildland Fire Reporting System (SWFRS, 2012) and Forest and National Park Service staff.

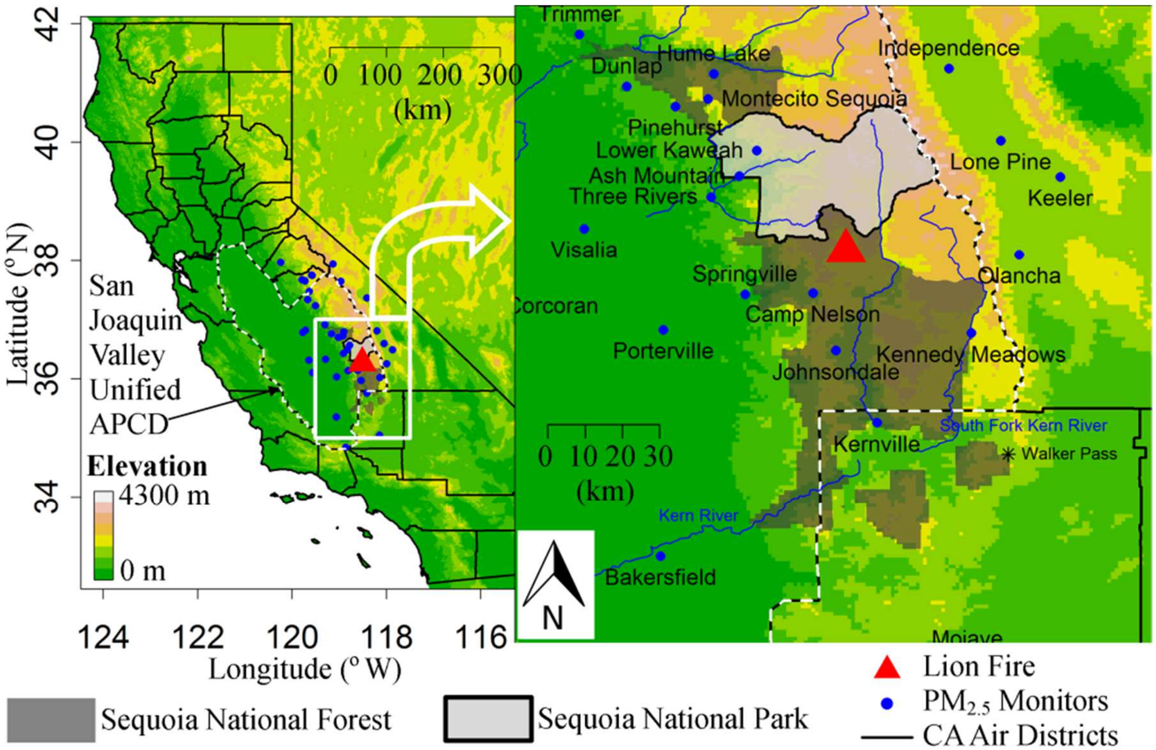


Figure 3.2 California air district boundaries and particulate matter less than 2.5 microns ($PM_{2.5}$) monitoring sites.

The ignition point was near Lion Meadow approximately three quarters of a mile north-northeast of the confluence of the Little Kern River and Lion Creek. The fire initially burned north and east toward the Great Western Divide which in this area separates the Little Kern and the Kern River drainages. The fire then progressed generally north and west; up the Little Kern River and Pecks Creek drainages. After ignition on July 8, 2011 the fire grew at less than 150 ha per day until July 17. Over the next three days the fire grew about 1,500 ha, followed by 2 days of slower growth. Reported fire size increased by over a 4,600 ha from July 23 to July 28, and then growth slowed to 206 and 334 ha. After July 30 the fire did not increase by more than 136 ha per day. Areas of the fire with high severity primarily occurred 7/19-20 and 7/25-28.

Data Collection

$PM_{2.5}$ data from 34 monitoring sites (Table 3.1) was compiled and used to assess smoke exposure to human health. Site locations ranged from approximately 16.6 km to 242.8 km from the ignition point of the Lion Fire.

Table 3.1 Site information arranged by distance from the fire.

Site	Latitude (N)	Longitude (W)	Regulatory	Temporary Site Dates	Distance (km) and direction from Lion Fire	
Camp Nelson	36.14105	118.60876	No	7/14-10/4/2011	16.6	SW
Springville	36.13625	118.81070	No	Permanent Site	30.7	WSW
Johnsondale	35.96970	118.54090	No	7/14-10/6/2011	33.2	S
Three Rivers	36.42792	118.91230	No	7/8-10/7/2011	40.2	WNW
Lower Kaweah	36.56580	118.77720	No	7/19-10/19/2011	40.6	NW
Ash Mountain	36.48940	118.82920	No	Permanent Site	40.8	NW
Kennedy Meadows	36.02135	118.13690	No	6/15-10/28/2011	43.4	SE
Olancha	36.25534	117.99390	No	7/29-9/7/2011	46.5	E
Lone Pine	36.59556	118.04917	Yes	Permanent Site	55.1	NE
Porterville	36.03183	119.05500	Yes	Permanent Site	55.5	WSW
Kernville	35.75506	118.41740	No	Permanent Site	57.5	S
Pinehurst	36.69731	119.01880	No	Permanent Site	62.1	NW
Keeler	36.48791	117.87111	Yes	Permanent Site	62.4	ENE
Montecito Sequoia	36.71900	118.92200	No	7/7-7/22/2011	65.9	NW
Independence	36.80994	118.20370	No	7/14-9/6/2011	66.1	NE
Hume Lake	36.79447	118.90490	No	7/7-8/11/2011	68.2	NW
Visalia	36.33250	119.29100	Yes	Permanent Site	70.4	W
Dunlap	36.75695	119.16531	No	7/8-7/20/2011	79.9	NW
Corcoran	36.10222	119.56583	Yes	Permanent Site	96.7	W
Trimmer	36.91119	119.30600	No	6/16-10/6/2011	100.8	NW
Hanford	36.31472	119.64333	Yes	Permanent Site	101.8	W
Bakersfield	35.35667	119.06278	Yes	Permanent Site	112.7	SW
Bishop	37.36667	118.41667	Yes	Permanent Site	122.2	N
Clovis	36.81944	119.71639	Yes	Permanent Site	124.1	WNW
Fresno	36.78194	119.77306	Yes	Permanent Site	126.6	WNW
Mojave	35.05035	118.14811	Yes	Permanent Site	139	SSW
North Fork	37.23300	119.50600	No	6/16-10/29/2011	139.1	NW
Oakhurst	37.33989	119.66700	No	8/12-9/7/2011	157.5	NW
Mammoth Lakes	37.64729	118.96430	No	8/14-9/6/2011	158.3	NNW
Lebec	34.84167	118.86056	Yes	Permanent Site	161.4	SSW
Lebec2	34.84150	118.86050	Yes	Permanent Site	161.4	SSW
Yosemite Valley	37.74861	119.58694	No	Permanent Site	190.2	NW
Lee Vining	37.93979	119.12840	No	8/14-9/6/2011	193.5	NNW
Tuolumne	37.96199	120.23920	No	8/13-9/6/2011	242.8	NW

Data for regulatory sites in California was obtained from the California Air Resources Board (CARB) Air Quality and Meteorological Information System (AQMIS, 2012). Data from tribal lands (Lone Pine and Bishop) is from the Tribal Environmental Exchange Network (TREX, 2014). Data from Nevada is from Clark County Department of Air Quality (CCDAQ, 2014). Smoke emissions from wildland fire in the Sierra Nevada 7/8-9/7/2012 (during the Lion Fire) were almost entirely from the Lion Fire with no other fire burning over 40 hectares. Wildland fires that occurred during this time (excluding the Lion Fire) exhibited short term (typically less than 1 day) and localized smoke impacts. Newly ignited wildland fires throughout the Sierra Nevada during this time were actively suppressed reducing or eliminating smoke impacts from other wildland fires at all monitoring sites. Levels of $PM_{2.5}$ at sites where smoke from the Lion Fire was not present were within normal variations typical in the Sierra Nevada (Cisneros et al., 2014).

Met One Instruments, Inc. (Oregon, U.S.A.) Beta Attenuation Monitors BAM-1020 (BAM) were used at permanent monitoring sites where data was collected year round, and Met One Instruments, Inc. Environmental Beta Attenuation Monitors (EBAM) were used at the non-regulatory temporary monitoring sites (see Table 3.1 for dates of operation). The BAM can be used as a Federal Equivalent Method (FEM) for measuring $PM_{2.5}$. FLMs BAMs do not adhere to the EPA FEM requirements and therefore the $PM_{2.5}$ data is not appropriate for compliance determination. The EBAM is designed for temporary and quick deployment. The EBAM has not been designated by the EPA as an FEM. BAM hourly measurements have a resolution of ±0.1 $\mu g\ m^{-3}$. EBAM accuracy is ±10% of the indicated value for hourly measurements with data resolution of 1.0 $\mu g\ m^{-3}$. The BAM (EBAM) hourly lower detection limit, set by twice the standard deviation of the hourly zero noise, is less than 4.8 $\mu g\ m^{-3}$ (6.0 $\mu g\ m^{-3}$). The BAM (EBAM) 24-hour average lower detection limit is less than 1.0 $\mu g\ m^{-3}$ (1.2 $\mu g\ m^{-3}$) (Met One Instruments, 2008a, 2008b).

Data calculations

Annual $PM_{2.5}$ calculations are based on the *Guideline on Data Handling Conventions for the PM NAAQS* (U.S. Environmental Protection Agency, 1999). Calculations and graphics were made using the R statistical environment (Carslaw and Ropkins, 2012; R Core Team, 2015). FLM air monitoring sites are typically not regulatory monitors and the data presented here is for comparative purposes to help better understand regulatory compliance and smoke impacts at more rural mountain communities in the southern Sierra Nevada. FOFEM (2015), a first order fire effects model, was used to calculate $PM_{2.5}$ emissions throughout the fire. Primary cover types used were Ponderosa Pine, Red Fir, and Sierra Nevada Mixed Conifer.

Air Quality Index (AQI) is a system of reporting daily air quality established by the U.S. Environmental Protection Agency (EPA). AQI has 6 categories (good, moderate, unhealthy for sensitive groups, unhealthy, very unhealthy, and hazardous) with thresholds depending on a given pollutant. EPA breakpoints (0-12, 12.1-35.4, 35.5-55.4, 55.5-150.4, 150.5-250.4, 250.5-500 μgm^{-3}) are used when determining the AQI for the daily or 24-hour $PM_{2.5}$ concentration. Daily human health impacts are also assessed by comparing one hour $PM_{2.5}$ concentrations to the standards set by the California Office of Environmental Health and Hazard Assessment for public health officials (Lipsett et al., 2013). The 1-3 hour average breakpoints from good to hazardous are 0-38, 39-88, 89-138, 139-351, 352-526, >526 μgm^{-3} respectively. These standards have not been implemented as regulatory standards but are routinely used by public health officials and land managers in California to assess smoke exposure and issue appropriate smoke advisories during a wildland fire.

Established regulatory NAAQS thresholds are used to determine long term health impacts from smoke exposure during the Lion Fire. Annual $PM_{2.5}$ statistics for permanent non-regulatory sites (Table 3.1) are calculated using the NAAQS data handling conventions and are compared to both regulatory standards and the urban regulatory monitoring sites.

NAAQS data handling convention protocol require the 98th percentile to be calculated as a rank value. All daily concentrations over a given year are ranked from highest to lowest. The 98th percentile is then determined dependent on the number of daily samples obtained. Daily samples are required to be representative of the entire year so that high concentrations are not missed due to sample timing. If less than 50 days are sampled equally throughout the year, the highest (1st) daily concentration is the 98th percentile for the year. With more samples, the rank increases until the 8th highest day of a given year is used when 351 or more days are recorded. Because temporary sites used in the Lion Fire did not collect full year data and were sampling every day, we used the 5th highest concentration as a conservative estimate of the annual 98th percentile. Using the 5th highest concentration by NAAQS data handling conventions would mean 201-250 of the daily mean concentrations for the year were recorded or 3 of the highest daily concentrations occurred when data was not being collected. Temporary monitoring site locations were located at elevations where $PM_{2.5}$ concentrations are highest in the summer and with maximum concentrations typically occurring during a smoke event (Cisneros et al., 2014). We believe using the 5th highest concentration, rather than the 8th highest concentration, is a conservative estimate of the 98th percentile for these temporary sites because the highest $PM_{2.5}$ concentrations for the year were likely during the Lion Fire. Use of the 5th highest concentration allows for 3 daily high concentrations to be missed during the segment of the year when $PM_{2.5}$ was not monitored and concentrations were likely lower. This ensures at least 55% of the highest daily readings for the year were captured at temporary sites during the Lion Fire thus leading to a conservative (low) valuation of the 98th percentile.

Annual federal standards are 3-year mean concentrations. Exceeding the federal threshold on a given day or for a year does not inevitably result in exceeding the federal standard.

Results

Smoke transport

Emissions estimates of $PM_{2.5}$ were approximately 24,000 Mg for the entire fire with the 3 highest emissions days being 7/26 (3,300 Mg), 7/25 (2,980 Mg), and 7/27 (2,600 Mg).

Upper air winds were typically from the west and generally moved smoke that was aloft to the east. HYbrid Single-Particle Lagrangian Integrated Trajectory (HYSPLIT) forward trajectories (NOAA Air Resources Laboratory, http://ready.arl.noaa.gov/HYSPLIT.php) frequently predicted air transport to the south or southeast and north through the Owens Valley in the mornings and northeast in the afternoons (Figure 3.3). Satellite imagery including smoke density data from the Hazard Mapping System (HMS) Fire and Smoke Product (NOAA National Environmental Satellite, Data, and Information Service http://satepsanone.nesdis.noaa.gov/FIRE/fire.html) and web cameras show the typical transport pattern from the fire was east and northeast across the Owens Valley into Nevada during the day (Figure 3.3) and would sink into local river drainages (primarily the Kern River drainage) at night.

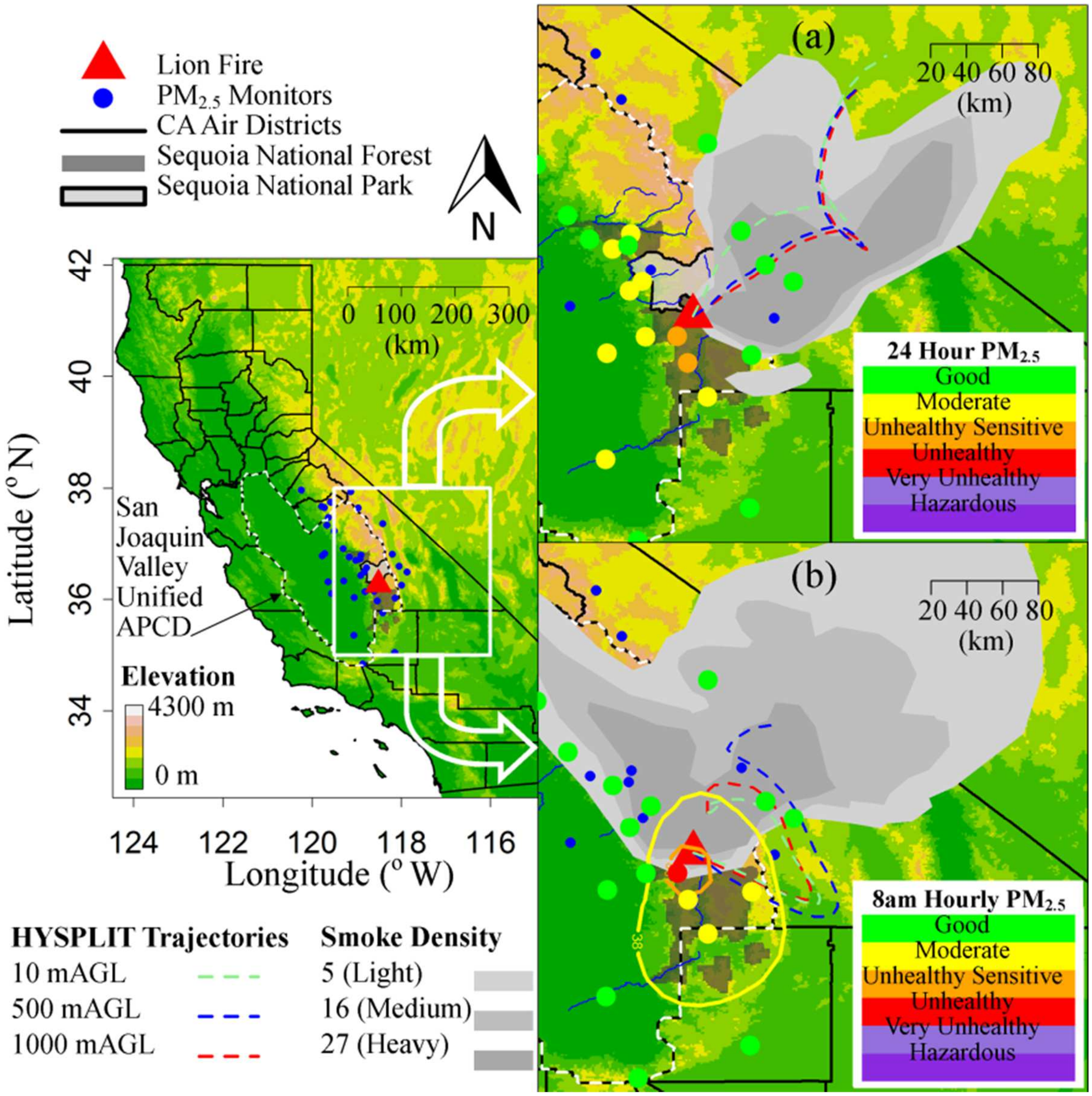


Figure 3.3 Smoke dispersal patterns using monitoring site data, Hazard Mapping System (HMS) data, fitted trend surface using hourly ground based data, and Hybrid Single-Particle Lagrangian Integrated Trajectory (HYSPLIT) forward trajectories for (a) afternoon 7/18/2011 1600 with daily Air Quality Index (AQI) for each site (fitted trend surface modeled using the hourly data was good for entire area) and (b) morning (7/26/2011 0800) with 1-hour AQI for each site.

Smoke was transported throughout the Sierra Nevada and east into and beyond the Owens Valley. The complex topography of the Sierra Nevada dictated timing and location of the largest $PM_{2.5}$ impacts as smoke frequently settled into and was transported through drainages. The Kern River drainage dominated the ground level transport of smoke from the fire. Night-time conditions drew smoke from the Lion Fire down the Little Kern River drainage into the Kern River drainage south to Lake Isabella and east towards Walker Pass. This pattern was typical of smoke patterns documented by ground observations and $PM_{2.5}$ monitoring during the fire.

$PM_{2.5}$ AQI during the Lion Fire

Air quality impacts to $PM_{2.5}$ from the Lion Fire as determined by AQI were localized, and extended furthest in the major transport corridor of the Kern River drainage. Effects from the Lion Fire could not be determined to impact the Central Valley. Sites nearer the fire typically saw increased concentrations into the good or moderate category.

Monitoring sites with the largest impacts from the Lion Fire $PM_{2.5}$ were Johnsondale, Camp Nelson, and Kernville. Camp Nelson experienced the highest levels of $PM_{2.5}$ (376 μgm^3 max hourly and 166.7 μgm^3max 24-hour) but had only 17 days (24-hour mean) above and AQI of good. Both Kernville (211 μgm^3 max hourly and 68.4 max 24-hour) and Johnsondale (333 μgm^3 max hourly and 95.8 μgm^3 max 24-hour) did not have such high single day concentrations, but both experienced more days above an AQI of good (38 and 35 days respectively) than Camp Nelson (Table 3.2).

Table 3.2 Estimated 98th percentile particulate matter less than 2.5 microns (PM2.5) for 2011 (3-year mean), mean $PM_{2.5}$ during and after the Lion Fire, and daily and hourly Air Quality Impacts (AQI) arranged by distance from the fire.

Site	98th per. $PM_{2.5}$ μgm^{-3}	Mean $PM_{2.5}$ μgm^{-3} during[a] fire (SD[c])	Mean $PM_{2.5}$ μgm^{-3} after[b] fire (SD[c])	AQI category Number of Days (Number of Hours) Good	Moderate	Unhealthy Sensitive	Unhealthy	Very Unhealthy	Hazardous
Camp Nelson	28.0[d]	16.0 (31.1)	9.0 (7.1)	39 (1226)	13 (41)	2 (7)	2 (14)	0 (2)	0 (0)
Springville	37.7 (30)	18.9 (8.1)	13.2 (7.5)	0 (1386)	57 (29)	0 (1)	1 (0)	0 (0)	0 (0)
Johnsondale	35.0[d]	17.8 (26.9)	7.4 (6.6)	22 (1227)	31 (113)	3 (19)	1 (15)	0 (0)	0 (0)
Kennedy Meadows	17.6[d]	8.3 (10.5)	4.1 (4.8)	53 (1431)	8 (19)	0 (4)	0 (0)	0 (0)	0 (0)
Olancha	7.4[d]	6.2 (6.4)	No Data	33 (849)	1 (7)	0 (0)	0 (0)	0 (0)	0 (0)
Kernville	34.0 (23)	20.1 (24.5)	8.6 (6.6)	22 (1230)	32 (118)	5 (33)	1 (10)	0 (0)	0 (0)
Pinehurst	19.8 (19)	11.6 (5.8)	11.8 (7.0)	40 (1436)	20 (5)	0 (0)	0 (0)	0 (0)	0 (0)
Independence	14.8[d]	6.9 (9.5)	No Data	41 (1111)	6 (24)	0 (0)	0 (0)	0 (0)	0 (0)
Trimmer	12.4[d]	8.6 (6.2)	9.7 (6.0)	56 (1451)	2 (1)	0 (1)	0 (0)	0 (0)	0 (0)
North Fork	12.2[d]	7.7 (6.4)	7.4 (6.6)	54 (1385)	2 (2)	0 (0)	0 (0)	0 (0)	0 (0)

[a] 7/9-9/7/2011
[b] 9/8-11/7/2011
[c] Standard Deviation
[d] 24-hour average estimates using 5th highest daily mean

Highest concentrations of $PM_{2.5}$ at Kernville typically come from the south and southeast (Figure 3.4a). During the Lion Fire the highest $PM_{2.5}$ concentrations at Kernville were from the north and northeast as smoke transport down the Kern River canyon dominated (Figure 3.4b). Smoke sank into the Kern River drainage and was transported down drainage to Kernville during the night and was reflected in the higher $PM_{2.5}$ concentrations from the north (Figure 3.4c). The

Kernville area then vented in the afternoon subjecting the Kernville area to smoke exposure primarily between 5 am and 11 am (Figure 3.5) with afternoon emissions from the fire lofted and generally transported east over the Owens Valley. Highest hourly concentrations at Kernville were in the unhealthy range. Hourly concentrations of $PM_{2.5}$ were normally lower at Kernville than Johnsondale (Figure 3.6) which was further up the Kern River drainage and nearer to the Lion Fire. At Kernville, AQI was unhealthy for 10 total hours and 1 day (Table 3.2) and unhealthy for sensitive for 33 total hours and 5 days.

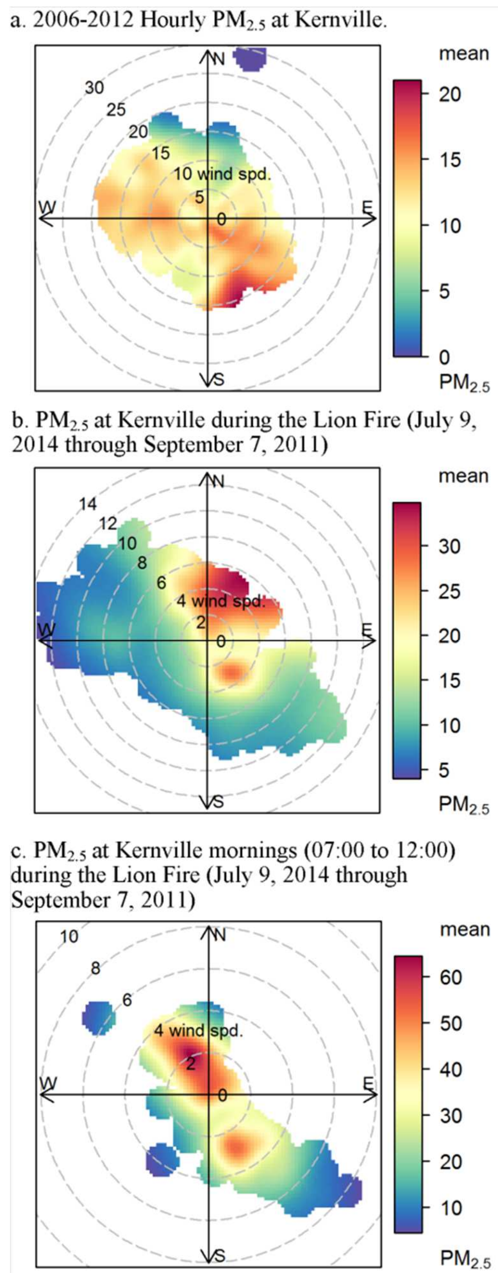


Figure 3.4 Particulate matter less than 2.5 microns ($PM_{2.5}$) showing concentration by wind speed and direction at the Kernville monitoring site for (a) all hourly data (2006-2012), (b) hourly $PM_{2.5}$ concentrations during the Lion Fire (7/9-9/7/2011), and (c) hourly $PM_{2.5}$ concentrations mornings (0700-1200) during the Lion Fire.

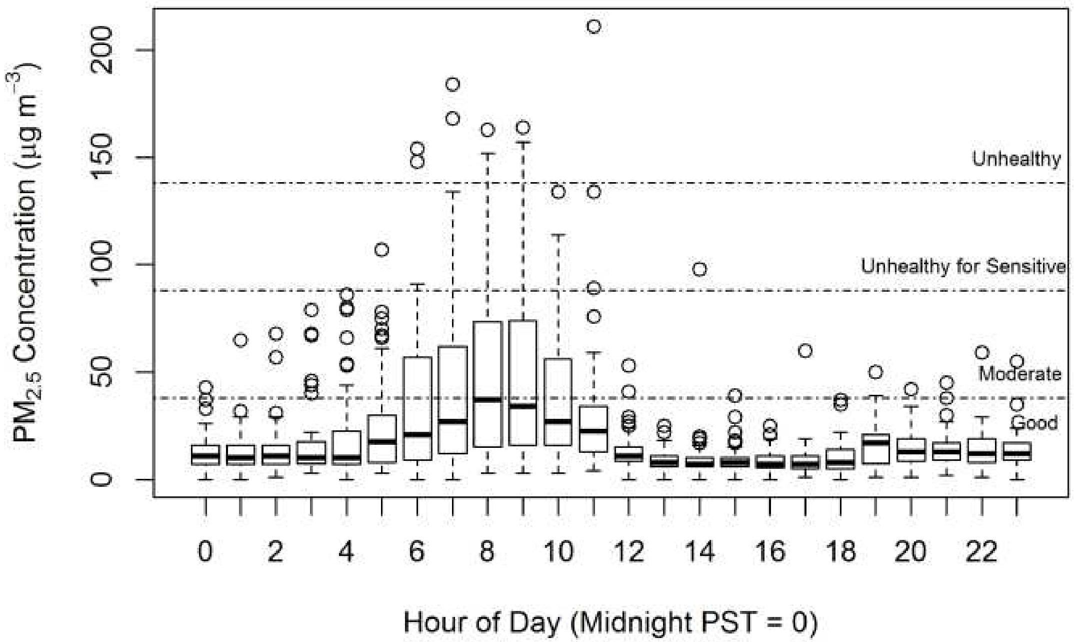


Figure 3.5 Modified boxplot (where the box is defined by the first quartile as the lower edge of the box, second quartile (median) as the black line, and 3rd quartile as the upper extent of the box, lower extent of data (minimum) are the horizontal lines below the box (lower whisker), the upper extent are the largest data point less than 1.5 times the interquartile distance (upper whisker), and outliers are represented by empty circles) showing diurnal pattern of particulate matter less than 2.5 microns ($PM_{2.5}$) concentrations at Kernville during the Lion Fire (7/9-9/7/2011).

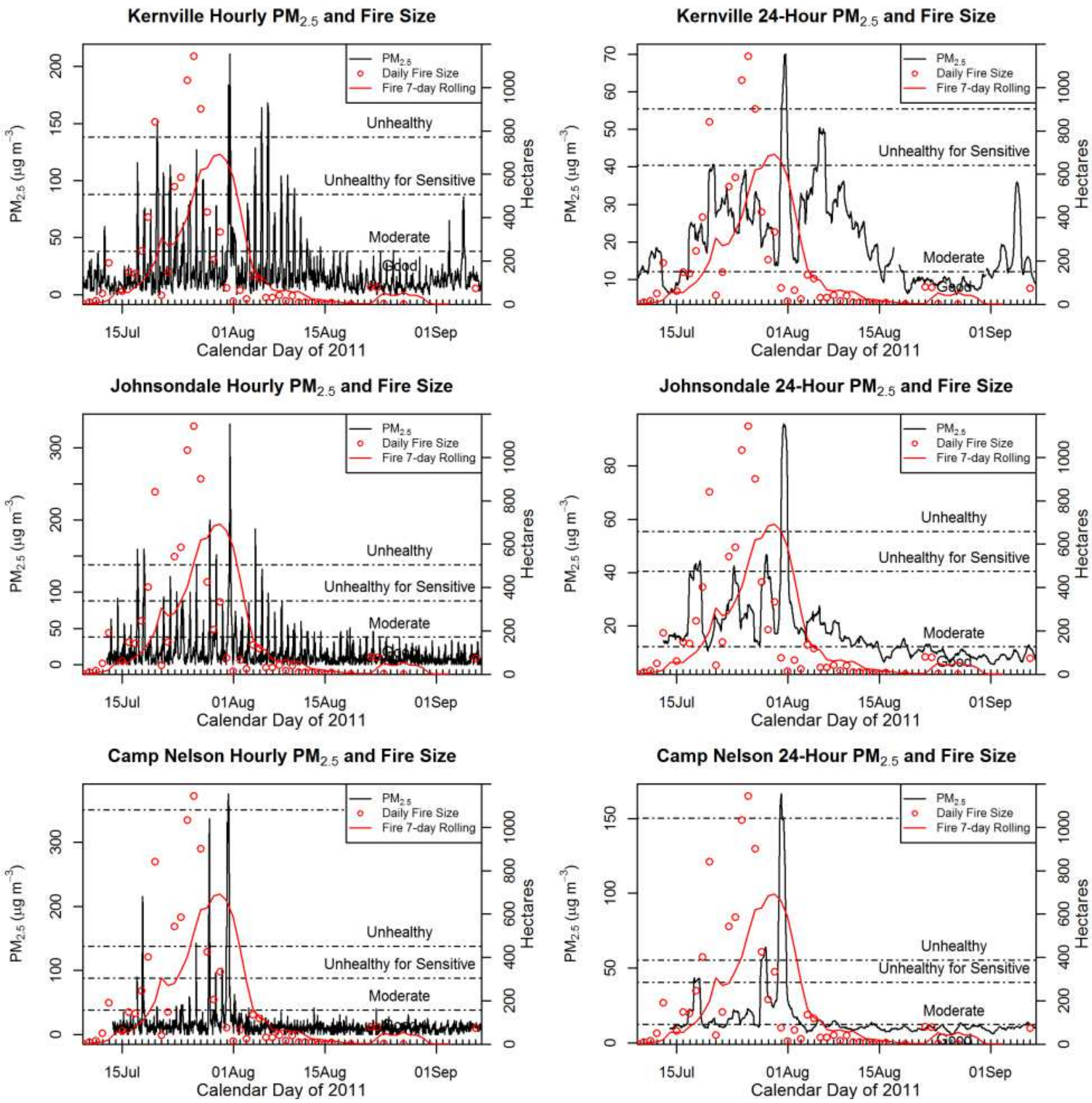


Figure 3.6 Hourly and 24-hour particulate matter less than 2.5 microns ($PM_{2.5}$) with Air Quality Index (AQI) breakpoints and reported daily and 7-day rolling fire size at Kernville, Johnsondale, and Camp Nelson during the Lion Fire (7/9-9/7/2011).

Smoke was transported from the Lion Fire to Johnsondale from the east through the South Creek drainage with highest concentrations in the morning. Hourly and 24-hour rolling concentrations at Johnsondale were typically highest in the morning with maximums in the unhealthy category. Fifteen hours on 6 different days with 1 daily AQI of unhealthy occurred at Johnsondale and 19 hours and 3 days were unhealthy for sensitive (Table 3.2).

The nearest monitor was approximately 16.6 km southwest at Camp Nelson. Camp Nelson was generally upwind of the fire as smoke typically was blown east during the day and settled into drainages east of Camp Nelson at night keeping concentrations generally low punctuated by a few hours of very high concentrations on 3 separate days. The Camp Nelson site recorded both the highest hourly (376 μgm^{-3}) and highest daily (167 μgm^{-3}) mean $PM_{2.5}$ concentrations during the Lion Fire (Figure 3.6) illustrating the impacts of nearness to the emissions source. Camp Nelson had hourly $PM_{2.5}$ in the good category for some time during every day monitored during the Lion Fire. For 2 hours on 7/31/2011 hourly $PM_{2.5}$ was very unhealthy. The 22 hours of unhealthy occurred over 7 days with 8 of those hours also occurring on 7/31/2011. On 7/31/2011 hourly $PM_{2.5}$ was good for 10 hours. AQI at Camp Nelson was very unhealthy for 2 hours, unhealthy for 14 hours and 2 days, and unhealthy for sensitive for 7 hours and 2 days (Table 3.2).

At the Kennedy Meadows site, $PM_{2.5}$ concentrations typically are highest with winds from the west but were highest from the south during the Lion Fire as smoke from the Kernville area was transported up the South Fork of the Kern River. At Kennedy Meadows, $PM_{2.5}$ concentrations were moderate or good with 4 hours unhealthy for sensitive.

East side monitors in the Owens Valley at Olancha and Independence reached maximums in the moderate category for $PM_{2.5}$.

Central Valley sites in Bakersfield, Visalia, and Fresno were moderate or good which is typical for this time of year in these urban sites during the fire. These sites were similar to other sites throughout the entire Central Valley.

$PM_{2.5}$ compliance with federal standards.

Although smoke impacts to mountain communities can be ascertained through temporary event drive monitoring, this is at best an incomplete assessment of public health impacts. To determine the impacts to public health from smoke in a fire and smoke adapted ecosystem, it is necessary to have some understanding of air quality over time. Federal air quality standards for $PM_{2.5}$ are a way to determine air quality in an area over time. Mountain communities throughout the Sierra are located in areas where prior to Euro-American settlement, smoke was present through most of the summer. These areas are not typically monitored for routine air quality because the populations are too small to warrant EPA monitoring sites. With the ability of land managers to suppress fire and fire becoming an inevitability that needs to be managed to mitigate impacts, background levels of pollutants generated from smoke must be incorporated into long term air management goals.

$PM_{2.5}$ monitoring in Kernville from 2006 to 2012 have highest concentrations typically coinciding with wind from the south and southeast from Lake Isabella with typical transport likely up the Kern River drainage from the Bakersfield area (Figure 3.4a). Annual three year mean (8.0-10.0 μgm^3) and 98th percentile (23-24 $\mu gm^{3)}$ concentrations of $PM_{2.5}$ at Kernville have been below federal standards since monitoring began in 2005 (Table 3.3). Monitors at Springville and Pinehurst, also ran year round, showing similar $PM_{2.5}$ readings. This illustrates that $PM_{2.5}$ in these areas is typically lower than the Central Valley. Although these areas are in federal and state nonattainment areas, the complications and limitations of air quality monitoring over an entire air basin is clear. Mountain communities of the Sierra Nevada have some capacity to allow for both some smoke impacts and still remain below federal and state thresholds for $PM_{2.5}$.The difficulties and challenges to administer air quality regulations where fire and urban pollution compete for use of the air basin are clear.

Table 3.3 National Ambient Air Quality Standards (NAAQS) calculated for the U.S. Forest Service year round monitoring sites.

$PM_{2.5}$ (μgm^{-3}) annual mean (3-year mean)							
Site	2012	2011	2010	2009	2008	2007	2006
Kernville	8.7(8.0)	8.1(7.4[a])	7.2(10.0[a])	7	15.9		
Pinehurst	7.9(8.1)	8.0(7.5)	7.9(7.7)	6.2(7.5)	9.1(8.0)	7.2	7.8
Springville	9.7(11.6)	14.1(11.7)	9.1(11.8)	10.0(13.7)	16.2(14.6)	14.8	12.8

$PM_{2.5}$ (μgm^{-3}) annual 98[th] percentile (3-year mean)							
Kernville	18.8(23)	34.0(23[a])	17.3(24[a])	17.7[a]	36.3	34.9[a]	37.3[a]
Pinehurst	16.8(18)	19.8(19)	18.5(24)	17.3(25)	34.6	22.1	22.9[a]
Springville	31.9(31)	37.7(30)	21.1(35)	28.6(43)	54.9(48)	44	44.1

[a] at least one quarter does not meet NAAQS requirement for number of valid daily averages

The concentrations of $PM_{2.5}$monitored at Kernville are representative of the higher elevation temporary sites set up to monitor air quality during the Lion Fire. Other monitoring throughout this area of the Sierra Nevada typically is similar to that at Kernville unless there are localized smoke impacts. Post fire $PM_{2.5}$ concentrations were similar between all sites once the smoke had subsided. Kernville and the other monitors run over multiple years at elevations above ~1,500 msl in the southern Sierra Nevada have shown that annual $PM_{2.5}$ concentrations are generally low (less than 12 μgm^{3})with smoke impacts from wildland fire typically producing the highest concentrations of $PM_{2.5}$ during the summer (Cisneros et al., 2014).

Discussion

Summary of $PM_{2.5}$ and smoke impacts from the Lion Fire

Hourly $PM_{2.5}$ concentrations during the Lion Fire (7/9-9/7) were typically good regionally with unhealthy for sensitive days going to near zero for sites in the Sierra Nevada out of the direct transport of the smoke (Table 3.2). AQI for daily exposure to $PM_{2.5}$ was generally good or moderate throughout the fire with 4-6 days unhealthy for sensitive or higher in the most impacted sites of Kernville, Johnsondale, and Camp Nelson (Table 3.2). Four unhealthy 24-hour averages were measured (1 at both Johnsondale and Kernville and 2 at Camp Nelson). Three of these readings occurred on 7/31/2011 at the three sites of Kernville (68μgm^{-3}), Johnsondale (95μgm^{-3}), and Camp Nelson (138μgm^{-3}) with the additional day at Camp Nelson (63 μgm^{-3} on 7/28). This was at the end of the most active period for the fire where the average reported acres burned was over 660 ha per day for the preceding week. Exceeding the unhealthy threshold helps to illustrate the importance of transport and distance which is particularly important when managing the lower emissions of a MF. Camp Nelson, the closest site to the fire, was typically upwind of the fire, but the nearness reduced the transport distance and the smoke would be more concentrated. Therefore when smoke was present, ground level concentrations of $PM_{2.5}$ were higher than other monitoring sites due to less mixing and dispersion during transport. Johnsondale was in the predominant direction of smoke transport as seen from the increased moderate and higher daily and hourly impacts. Typical nighttime transport of smoke into and down the Kern River drainage exposed the Kernville area to more smoke even though this location was further away than many other sites thus exposing the Kernville area to lower hourly concentrations but over more hours.

Kernville, the only year round site that recorded smoke impacts saw an increase in annual $PM_{2.5}$in 2011 of approximately 1 μgm^{-3} from 2010 and 2009 but was .6 μgm^{-3} lower than 2012. At Kernville, the 2011 annual mean was 8.1 $\mu gm^{-3}$with a 3-year annual mean of 7.4 μgm^{-3}. Annual mean has been below the federal standard since 2008 with 3-year mean consistently below the standard. This is similar to the other $PM_{2.5}$ monitoring sites operated by the USFS in the wildland urban interface of the western slopes of the Sierra Nevada (Table 3.3). Annual mean $PM_{2.5}$ at Kernville below the EPA annual mean standard (12 μgm^{-3}) in 2011 when the Lion Fire occurred and again in 2012 indicates the potential for a MF of this size in this area to be conducted without causing violations to the federal standards.

The federal standard (98^{th} percentile) was not exceeded for 2011 (Lion Fire year) at any site. Springville, the only site where the 98^{th} percentile estimate is above the 3-year standard, had only 1 of the ten highest days occurring during the Lion Fire(Table 3.2) showing the influence of anthropogenic impacts on this lower elevation site nearer to the populated Central Valley. Most of the sites monitored for the Lion Fire did not go above the federal standard for even a single day. Kernville, Johnsondale, and Camp Nelson all had days (7, 6, and 3 respectively) above the 35 μgm^{-3} annual standard from smoke. The federal annual 98^{th} percentile at Kernville for both the Lion Fire year of 2011 (34.0 μgm^{-3}) and the 3-year mean (23 μgm^{-3}) were below the federal standard (Table 3.3). $PM_{2.5}$ exposure including sites with estimated 98^{th} percentile all remained below the federal standard for 2011.

Implications of Wildland Fire Smoke

Wildland fire and subsequent smoke impacts have been an evolutionary pressure in the Sierra Nevada that is integral to this forest ecosystem. As people have moved into this area, smoke impacts to human health have become a pressing political issue. Historic wildfire suppression has erased the cultural memory of fire and smoke in this area leading regulators and the general public to have little tolerance to smoke and unrealistic expectations for continued suppression of fire in the face of climate change and unnatural fuel loading. Additionally, because smoke can be sensed at low levels, any smoke in this area leads to the assumption that air quality has been hazardously compromised. Although emissions from wildland fire undoubtedly have adverse impacts to human health during a fire, there has been little interest in future impacts to air quality and public health through fire suppression. Because weighing beneficial impacts to air quality from a healthy forest ecosystem is difficult, it is typically ignored for event driven reactionary land and air management. This strategy is likely leading to a less healthy and resilient Sierra Nevada ecosystem (Miller, 2012) which is more susceptible to other stressors such as climate change which in turn may lead to larger more intense fires. Additionally, the current excessive fuel loadings will produce increased emissions and smoke impacts when burned. Fuel loads can be expected to increase unless the use of fire is increased leading to the possibility that smoke in the future will significantly impact large portions of the region including the heavily populated Central Valley when this area inevitably burns.

The Sierra Nevada is further restricted by current air regulatory policy and alignment. While areas throughout the Sierra Nevada (similar to sites in this study) are likely in attainment of federal $PM_{2.5}$standards (Cisneros et al., 2014), these areas, in the regulatory environment, are in a non-attainment area. Fire in any portion of a non-attainment area will necessarily be under greater scrutiny for any emissions with current air quality, even at distant sites with little to no impacts from the fire, anywhere within the air basin being the primary factor for MF decisions. Regulators will find it difficult or impossible to consider long term consequences and impacts to air quality when air quality standard is being exceeded in the Central Valley urban areas of the air district.

Smoke from the Lion Fire impacted air quality both visibility and public health as was apparent in the high hourly concentrations and over a month of some levels of smoke in many areas, but in regard to current federal standards for annual (12 μgm^{-3}) and 98th percentile (35 μgm^{-3}), Kernville (and likely much of the surrounding area) were in compliance. Without an increase in anthropogenic emissions in this area, a fire of the size and intensity of the Lion Fire could be allowed to progress naturally every three years with little chance of exceedance of federal regulatory standards for $PM_{2.5}$ in the Kernville area. The relatively few days and hours of unhealthy AQI during the Lion Fire can be contrasted to the McNally Fire. The McNally Fire was a full suppression high intensity fire in this area of the Sierra Nevada in 2002. Smoke from the McNally Fire impacted the Owens Valley (Cisneros et al., 2012). Fire of this magnitude and intensity and the subsequent increased emissions can be expected to increasingly impact large urban centers if smoke emissions are not managed more efficiently. Timing and quantity of smoke emissions have no opportunity to be managed during a high intensity wildfire in the Sierra Nevada. Options to control or mitigate smoke impacts from the large high-intensity Rim Fire (2013 - Stanislaus National Forest and Yosemite National Park) were virtually non-existent. High intensity fires with full suppression send large amounts of smoke long distances impacting a much more extensive and populated area.

Smoke causes a myriad of impacts to human health. Suppression policies appear to not only be moving these impacts to future generations but with increased fire size and intensity smoke impacts can be expected to have increasing mortality impacts and the associated social cost (Kochi et al., 2012). $PM_{2.5}$ air quality impacts from the Lion Fire were primarily moderate or good using current thresholds, showing the potential for mitigating future impacts from a larger more intense fire using MF. Additionally, with a return to the natural fire regime in the Sierra Nevada more typical of the size and intensity of the Lion Fire there is the potential to use MF to control both the timing and amount of smoke to adhere to present federal air quality standards.

Smoke from fire will be experienced by people living or recreating in the Sierra Nevada. Only the timing, extent, and intensity of smoke exposure can reasonably be managed. Air quality impacts from emissions from the Lion Fire were under current NAAQS for $PM_{2.5}$.

Policy Recommendations

More case studies are necessary to understand the complex interaction between fire emissions and public health in the Sierra Nevada. The Lion Fire illustrates that while there were no violations to the annual federal $PM_{2.5}$ standards, smoke exposure to a small proportion of the public did occur which likely impacted public health (Delfino et al., 2009), particularly for sensitive groups (Elliott et al., 2013). Thus, managing a fire of this magnitude and intensity in the

Sierra Nevada is possible without extreme smoke impacts typical of large high intensity suppression fires.

Current policy and regulatory enforcement is designed to concentrate protection on immediate impacts. This works well with anthropogenic emissions but is in effect pushing the onerous impacts of smoke exposure to subsequent years. Fuel loading, increases in wildland urban interface, and climate change are coalescing to limit the proactive use of fire for ecological and thus public health benefit. Fire managers are near to having no alternative but to be reactive to fire which will limit their effectiveness to control emissions.

Although satellite imagery, dispersion models, other products designed to evaluate potential smoke impacts are useful tools (Price et al., 2012), they typically over-predicted the ground level concentrations and extent during this event leading to an increased perceived risk of smoke exposure (Figure 3.3). These tools should be used with caution for projecting real ground level concerns about air quality for smaller fires in complex terrain such as the Sierra Nevada and should not be substituted for ground based measurements if at all possible.

A large high intensity fire in the Sierra Nevada, the McNally Fire in 2002, impacted a much larger area and caused federal standards to be exceeded (Cisneros et al., 2012)). During the McNally Fire, monitoring for coarse particulate matter (particulate matter less than 10 microns in diameter) recorded concentrations that were in the hazardous range. A large high intensity fire in this area has the potential to increase public health impacts to local communities, the Owens Valley, and the Central Valley. Current policy should be altered to encourage fire of historic size and intensity to be managed in the Sierra Nevada even during times of poor air quality in the Central Valley. Future emissions potential including considering emissions from large high intensity fires and the increased impact to public health should not be ignored. Failure to confront this difficult air quality conundrum necessarily leaves the impacts to future generations. Management of naturally occurring fires during advantageous meteorological and ecological conditions should be prioritized to limit future air quality impacts.

Managing smoke impacts from fire on a landscape level in the Sierra Nevada is a complex policy problem. A possible scenario under current air quality conditions would be to manage $PM_{2.5}$to remain below current federal thresholds. This would provide policy managers the opportunity for managing smoke emissions and consider current and future levels of exposure possibly cycling MFs into and out of an area over multiple years to provide landscape level forest restoration while exposing the public to years with and without MF. Fire size can also be managed more easily during a MF possibly managing total area burned where feasible to take advantage of good dispersal days. Smoke impacts were difficult to determine from the reported burn area. This is possibly due to the difficulty of timely estimating the area during a fire. Reported hectares were a poor indicator of localized smoke impacts but impacts typically went over unhealthy for sensitive at the monitoring sites when more than 300 hectares were reported for the day. The 7-day rolling average of hectares burned (Figure 3.6) was a better predictor of $PM_{2.5}$ impacts ($r^2 = .34$). This likely was because the longer average helped to reduce the error in reported fire size and also included some matrix for areas still burning and is an area that needs additional research. Considering fire size and duration has the potential to be a simple way to create trigger points where a progressive program of outreach could protect the public from smoke exposure during a fire.

Conclusions

Managing smoke from wildland fire is complex with no simple way to approach the myriad of decisions required. Quantifying impacts to air quality from a smoke event help to inform these decisions. The smoke from large high intensity fires, because of their widespread and highly detrimental impacts to air quality and public health are the focus of much of the current research. These fires were not typical and are a product of anthropogenic activities. Smoke impacts from wildland fire the size and intensity typical of the ecosystem such as the Lion Fire are not well documented. Without this understanding, smoke regulatory agencies necessarily must take a conservative approach where impacts must be predicted and assumed to be as widespread and intense as a large high intensity fire. Measured smoke impacts from the Lion Fire suggest this should not be the assumption.

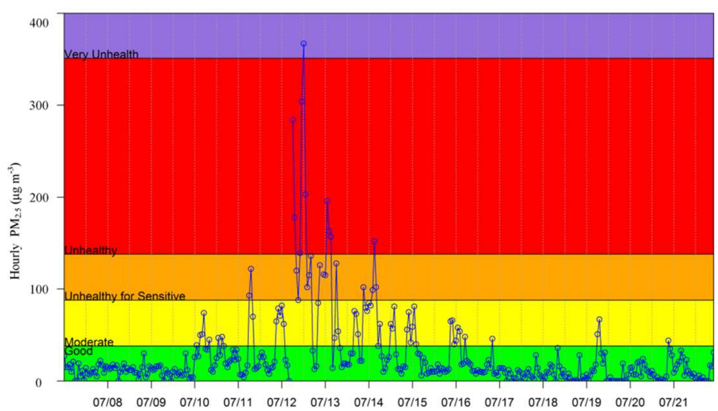
Smoke from the Lion Fire was present for 1-2 months in small rural and mountain communities close to the fire in the Sierra Nevada. There were impacts to air quality, particularly to sensitive groups that must be mitigated if future fires of this size are allowed to burn. Impacts for the Lion Fire were localized and below federal thresholds for $PM_{2.5}$ at sites closest to the fire. $PM_{2.5}$ concentrations at urban sites in the Central Valley remained low and typical of other years without fire throughout the fire duration. Smoke from the Lion Fire did not appear to impact $PM_{2.5}$ at Central Valley sites. $PM_{2.5}$ concentrations were found to be below federal and state standards for air quality at all monitoring sites during the Lion Fire.

The potential exists to manage fire at the intensity and extent historically seen in the Sierra Nevada while risks to public health from smoke are minimized with air quality impacts held below regulatory standards. This opportunity to manage smoke by mitigating public health impacts to federal regulatory standards also provides a framework for assessing the tradeoff between short- and long-term impacts of smoke to public health through assessing smoke impacts over 1 to 3 years instead of only considering hours or days. Recognizing and mitigating short term exposure, particularly to sensitive populations is obviously essential but should not be the only factor considered in this complex decision. Wildland fires should not all be treated the same where the avoidance of short term smoke impacts are the only consideration. Avoidance of major air quality impacts from large high intensity fire and future public health benefits of allowing some emissions must be considered. Policy needs to allow flexibility to manage air quality impacts from a fire the size and intensity of the Lion Fire historically experienced in the Sierra Nevada including allowing wildland fire at times of poor air quality in the Central Valley. Allowing smoke events of this magnitude has potential to reduce regional smoke impacts to air quality.

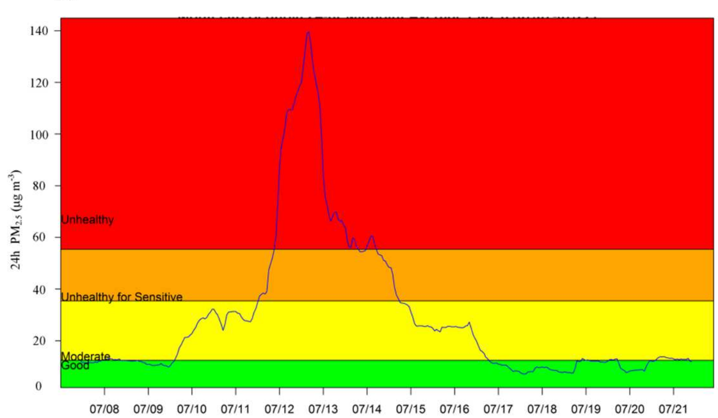
Chapter 4.0 Smoke impacts from suppression versus ecologically beneficial wildland fires on public health over multiple years at a single site

The Lion Fire case study quantified the impacts from a single managed fire. Concentrations of $PM_{2.5}$ during the Lion Fire impacted air quality especially for sensitive groups where for hours or days at a time the levels of $PM_{2.5}$ were a concern. Large concentration gradients allow wildland firefighters to move small distances to avoid exceeding harmful exposure (Viner et al., 2015). Controlling emission timing and quantity through prescribed and ecologically beneficial fire can capitalize on these large gradients in $PM_{2.5}$ concentrations to reduce public health exposure. Many locations in the southern Sierra could see and smell smoke for over a month during the Lion Fire. Short term air quality impacts were observed while federal health standards for population level exposure were still in compliance.

Full suppression has left an abundance of fuels in many areas. Landscape management of a fire prone area must compensate for additional fuels. Prescribed and managed fire that is returned to an area after missing numerous fire cycles (reentry burn) typically produces much more smoke. The Redwood Mountain prescribed fire in Kings Canyon National Park is an example where a reentry burn blowing into a recreation area (Montecito Sequoia) created high concentration of $PM_{2.5}$ (Figure 4.1).



(a)



(b)

Figure 4.1 Fine particulate matter ($PM_{2.5}$) (a) hourly and (b) 24 hour mean concentrations at the Montecito Sequoia during the Redwood Mountain prescribed fire in 2011.

The Redwood Mountain Prescribed Fire burned 250 ha over 4 days in 2011. However, long term smoldering continued for about a month but impacts to air quality at Montecito Sequoia were minimal. The burn was approximately 4 km directly west from the site. Hourly and daily AQI was unhealthy during much of the burn and interior burn down days. This single event illustrates the obvious importance of distance and fuel loading on smoke. Additionally, fire size and intensity also have a role in emissions.

Regulating to one monitor can be problematic. Essentially, placement of an individual monitor at or near any smoke emission source can capture high concentrations of $PM_{2.5}$. Even the smallest fire will create an area of extremely high $PM_{2.5}$ not representative of the larger area. The Lion Fire case study illustrated that a fire burning ~200-400 ha per day did produce impacts but they were below federal health standards over a wide area and may provide insight into controlled release.

Single events while useful in determining the possibility of managing wildland fire while still largely remaining in federal compliance. The consequences of fire suppression policy are difficult to quantify. This chapter attempts to explore the relative effects of prescribed, managed, and full suppression fire over time. A single site in the Sierra Nevada between federally protected forests and the urban Central Valley was selected to represent landscape level $PM_{2.5}$ in an area that experienced these types of fires.

Chapter 4.1 Managed wildland fire as an option for smoke mitigation to air quality in the California Sierra Nevada by the reduction of landscape level megafire impacts to federal health standards for fine particulate matter.

Abstract

Wildland fire is an important ecological process that has shaped the ecosystem of the California Sierra Nevada. Personal accounts from pre-20th century describe a much smokier environment than present day. The policy of suppression beginning in the early 20th century and climate change are contributing to increased megafires and emissions. We use a single particulate monitoring site representative of the local wildland urban interface to explore impacts from prescribed, managed, and full suppression wildland fires from 2006-2015 to produce a contextual assessment of smoke impacts over time at a landscape level. Prescribed fire had little effect on local particulate matter air quality exclusively in the short term (hours). Managed fire and increased emissions from the largest burn areas increased short term air quality impacts while the site remained below federal health standards. The single megafire in this area created the highest short term impacts and produced the only annual value over a federal standard. Wildland fire could be managed annually at 8-22% of the historic area and still likely remain below federal standards. Considering air quality impacts from smoke at a landscape level over time can give land and air managers a metric for broader evaluation of smoke impacts particularly when assessing ecologically beneficial fire and help to lessen large smoke impacts to public health from megafire.

Background

Wildland fire in the Sierra Nevada

Wildland fire has been an important component to the fire prone ecosystem of the Sierra Nevada of California (Kilgore et al., 1979). Smoke from these wildland fires was present throughout the Sierra Nevada. Pre-European settlement, the Sierra Nevada had much more wildland fire and smoke (Stephens et al., 2007). Wildland fire is increasing in frequency and size with longer fire durations since pre-European settlement and the era of suppression (Westerling et al., 2006) and increasing at higher elevations (Schwartz et al., 2015). Changing climate and fuel build up through policies of suppression contribute to the increase in megafires. Large scale fuel reduction programs can help reduce the chance and help in suppression of high intensity megafires (Williams, 2013). Fire has long been an integral part of the evolution of the Sierra Nevada and will inevitably remain a component of wilderness and other protected natural areas. Whether fire comes as a megafire or where prescribed or natural ignitions are managed for forest health is largely a matter of policy and land and air managers' willingness to allow wildland fire.

The link between human and ecological health

Global anthropogenic driven biodiversity loss and change are a risk to ecological function with 322 of terrestrial vertebrates having become extinct since 1500 and remaining species showing a 25% decline in abundance with invertebrates declining by 45% (Dirzo et al., 2014). While human activity is understood to be rapidly transforming environmental systems globally and affecting human health, there are limitations and gaps in research into this important topic (Myers et al., 2013). Understanding the link between fire adapted ecosystem health and smoke impacts on public health from naturally occurring wildland fire is one of these gaps. Research on smoke impacts from wildland fire has focused on large fires with pronounced impacts to air quality. While this is an important topic to understand, it is impossible to accurately assess tradeoffs between full suppression and using managed fire to control emissions when only considering large high intensity fires such as the Rim Fire in 2013.

Management actions can help minimize loss of ecosystem services (Millar and Stephenson, 2015). Fire is a component necessary for ecosystem health that is particularly evident in fire prone areas such as the California Sierra Nevada. While urbanization has reduced contact for much of the population and the overall beneficial effects from contact to natural areas (Hartig et al., 2014), natural processes are vital to keeping the integrity of remaining natural systems for future generations. Human and ecosystem health are interrelated with ecosystem services broadly providing cleaner air and water, natural hazard mitigation, and climate stabilization (Jackson et al., 2013). The Sierra Nevada of California contains large tracts of federally protected land where conservation of this fire prone ecosystem is prioritized.

Fine particulate matter, health, and wildland fire smoke

Ambient particulate matter pollution is important to human health (Lim et al., 2012). Particulate matter less than 2.5 µm ($PM_{2.5}$) is a leading environmental risk factor (Hänninen et al., 2009) and improvements in the analysis of health impacts from exposure (Fantke et al., 2015) are helping advance understanding.

Wildland fire is largely found to be associated with risk of respiratory and cardiovascular disease (Liu et al., 2015). Research on monitoring impacts of wildland smoke is attempting to provide tools for early warning for evacuation during fire suppression (Huff et al., 2015) where impacts can be long distances from the fires (Le et al., 2014). Prescribed burning and more localized smoke impacts are also a concern for public health (Haikerwal et al., 2015). It is easy to understand that any increased pollutant emission leads to additional exposure and will have negative impacts, but smoke exposure in a fire prone community cannot be completely eliminated by suppression.

Scientific publication is likely easier when proving smoke impacts to human health. Similar to other areas of research, relative impacts from smoke emitted under different fire management scenarios or findings of no significant impacts from a lesser smoke event are likely more difficult to publish in a system that disfavors negative result publication (Fanelli, 2012). There has been little research parsing out tradeoffs and differences between suppression management and megafire impacts and managed naturally occurring ecologically beneficial fire smoke impacts.

Policy and conflicting consensus when managing wildland fire

Restoring forest heterogeneity is complex and not simply dependent on reducing fuels and fire severity (Baker, 2014). Naturally occurring managed Sierra Nevada fires can help shape this fire prone landscape (Collins et al., 2007). Fire management is a complex undertaking (Thompson and Calkin, 2011; Vining and Merrick, 2008) with entrenched agency disincentives (North et al., 2015a; Topik, 2015) and robust scientific debate (Boer et al., 2015; North et al., 2015b; Thompson and Calkin, 2011). Policy and public support are needed for large scale use of managed fire. The fire prone Sierra Nevada contains many widely studied species with exaptation to fire from the evolutionary pressure of historic fire regimes (Keeley et al., 2011). Human health impacts must include suppression altering of this ecosystem.

Smoke is an important component to managing federal wildland fire (Stephens and Ruth, 2005). Managing smoke from wildland fire in the Sierra Nevada is made more complex by being adjacent to the Central Valley of California in an already compromised airshed due to anthropogenic uses such as vehicle emissions and agriculture. The Central Valley is in non-attainment to a number of air pollutants including $PM_{2.5}$. $PM_{2.5}$ monitoring sites throughout the Sierra Nevada are below these standards even while experiencing wildland fire smoke events (Cisneros et al., 2014). This complexity is likely in part because of the visibility of smoke and the assumption that seeing or smelling any smoke equates to bad air quality. While any wildland fire will inevitably lead to some impacts the assumption that simply seeing smoke associates to ground level impacts is misleading (Preisler et al., 2015). Ecologically beneficial fire the size and intensity historically seen in the Sierra Nevada can be managed to meet federal standards (Schweizer & Cisneros, 2014).

Fire and air quality policy has created an atmosphere that is not conducive to mitigating the risks to public health in a fire adapted wilderness (North et al., 2015). This may be in large part due to intolerance of regulators to any smoke and where broader acceptability needs public deliberation (Weisshaupt et al., 2005). Suppression is the default for immediate political pressures being felt by air and land managers. Policy is intended to provide long term benefits to society. The policy of suppression of wildland fire has routinely been discredited for ecological health

(Backer et al., 2004) while ecosystem function and health are integral to human health (Jackson et al., 2013). Smoke impacts from wildland fire can easily be used as a community rallying point to entrench a desire to suppress all fires for a smoke averse population (Shindler and Toman, 2003). A single day of smoke can be used to advance a suppression agenda by relying on perceptions derived from an era where suppression can virtually eliminate smoke. But, these emissions are not gone they are simply delayed until suppression is no longer possible (Steel et al., 2015) and additional fuel loading creates smoke events beyond the normal (Gonzalez et al., 2015; Schoennagel et al., 2004). Smoke impacts are inevitable in a fire adapted ecosystem. But, increased population may make a return to a natural fire regime and the smoke emissions problematic (Dombeck et al., 2004; Hurteau et al., 2014). We assess wildland fire smoke impacts on air quality to a segment of the Sierra Nevada by using short term (hourly and daily) along with long term (regulatory thresholds) impacts on $PM_{2.5}$ in an effort to give a more comprehensive understanding of prescribed, managed, and full suppression fire emissions and public health.

Objectives

This study investigates whether continued wildland fire suppression policy and the subsequent large high intensity wildland fires on federally protected forests in the Sierra Nevada produce a greater impact on public health than managed wildland fire used for ecological benefit.

To assess this, we attempt to answer the following questions: 1) Does $PM_{2.5}$ at a site located mid-elevation on the west side of the Sierra Nevada have a consistent annual pattern where the highest measurements are associated with wildland fire? 2) Are wildland fire location, emissions (size and intensity), and distance the primary drivers for impacts to $PM_{2.5}$ at this site? 3) Are the highest measurements at this site caused by large full suppression fires and what are the quantifiable impacts to public health? 4) Do managed fires in this area minimize these impacts to public health? 5) Can any smoke impact mitigation be accomplished by using ecologically beneficial wildland fire on the landscape?

Answering these questions is intended to provide a more nuanced understanding of smoke impacts along with furthering the discussion of smoke, public health, and air quality in a fire prone area. The implications of landscape level use of ecologically beneficial wildland fire and smoke management are explored. The objective of this analysis is to provide a context for linking air quality impacts from wildland fire smoke to fire management policy and is intended to provide insight into the tradeoffs in air quality and human health protection from full suppression and managed fire.

Methods

Site description

The Pinehurst site (Latitude 36.69731; Longitude -119.01880; elevation 1246 masl) is located in the western slopes of the Sierra Nevada in Fresno County, California. Pinehurst is located approximately 0.5 km east of the Sequoia National Forest boundary and approximately 2.5 km west of the boundary to Kings Canyon National Park on the ridge between the Mill Creek and Dry Creek drainages (Figure 4.2). Mill Creek flows west and eventually merges with the Kings River below Pine Flat reservoir. Dry Creek drainage runs south joining the Kaweah River just west of the Lake Kaweah reservoir. Pinehurst is located in the San Joaquin Valley Air

Pollution Control District (SJVAPCD) and is considered for regulatory purposes to be in non-attainment of $PM_{2.5}$ while most of the Sierra Nevada is actually below the federal standard (Cisneros et al., 2014).

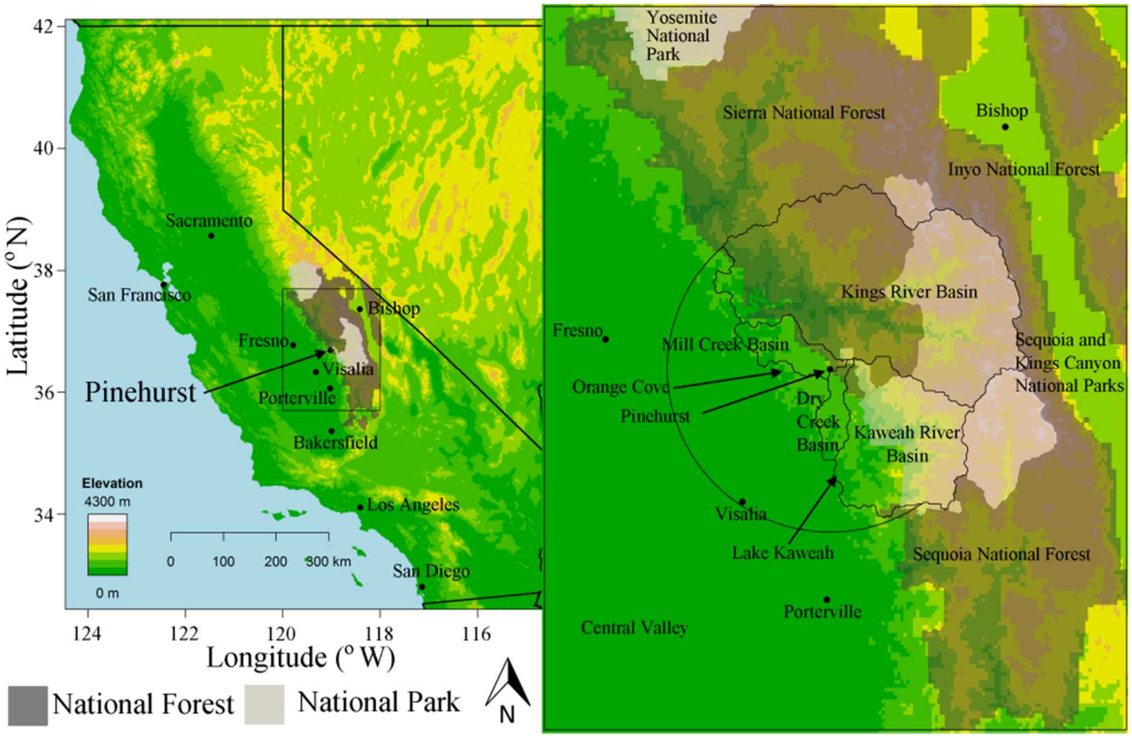


Figure 4.2 Site location overview with federally protected National Forest and Park Service land in the Sierra Nevada along with Pinehurst region enlarged and displaying local area drainages and foothill bounding area (semi-circle to the west) into the Central Valley of California.

Pinehurst is situated east of the Central Valley of California (the lower elevation (dark green) in Figure 4.2) near the middle of the north-south orientation and is mid-elevation on the western slope of the Sierra Nevada with the nearest point on the crest (>3,000 masl) 55 km to the east. Immediately east of Pinehurst is almost exclusively federally protected land of Sequoia and Kings Canyon National Parks, Sierra National Forest, and Sequoia National Forest with Inyo National Forest east of the crest of the Sierra Nevada. Much of this land is wilderness with the primary source of pollutants from wildland fire emissions due to limited anthropogenic development and the subsequent emission sources.

The Central Valley is largely developed urban and agricultural area. Emissions from the Central Valley are largely anthropogenic with urban and agricultural sources. Large urban areas in the Central Valley are located from Sacramento to the north and Bakersfield to the south (Figure 1). The city of Fresno, California is located approximately 60 km west of Pinehurst while Visalia is 45 km southwest. The closest point on the Central Valley floor (<150 masl) is about 25 km WSW near Orange Cove. Atmospheric transport is generally from west to east although ground level transport at Pinehurst is influenced by the major drainages of the Kings and Kaweah Rivers.

Meteorological data is collected by a Remote Automatic Weather Station (RAWS) site maintained by the Sequoia National Forest and Met One Instruments, Inc. Beta-attenuation monitor (BAM) was used to collect hourly $PM_{2.5}$ from 2006-2015.

A BAM, designed to be used as a permanent $PM_{2.5}$ monitor, can be used as a Federal Equivalent Method (FEM) for regulatory compliance. The BAM at Pinehurst does not adhere to EPA FEM requirements. The BAM, having a standard range of 0-1,000 µg m^{-3}, resolution of ±0.1 µg m^{-3},with an hourly lower detection limit, set by twice the standard deviation of the hourly zero noise, less than 4.8 µg m^{-3}, and a 24-hour average lower detection limit less than 1.0 µg m^{-3} (Met One Instruments, 2008a) is widely used to determine air quality and public health throughout California.

Wildland fire data and emission calculations

Wildland fire data were obtained from the National Interagency Fire Center (NIFC, 2016) and the National Fire and Aviation Management web applications data warehouse (FAMWEB, 2016). These data include descriptive information on wildland fires from federal and state agencies such as size, start and end dates, location, etc. Fire start date was used to compile monthly statistics. Start date was used because it was available for all fires and provided a consistent starting point for analysis. Start date was used in analysis as a general indicator of broadly when smoke was impacting the site at Pinehurst. Using start date for all fire sizes has the potential to bin larger fires into a single month where emissions often extended to subsequent months or at times even primarily in the next month. Therefore, to avoid having all smoke impacts from larger fires in a single month, when assessing smoke impacts on a more specific time frame at Pinehurst we used the individual fire size and rate of growth as reported in daily reports. We used all available data (1970-2015) to assess seasonality and timing of emissions but relied on the years after 2006 for direct comparison to air quality data from Pinehurst.

Individual fires were categorized for fire management action. Fire management actions were prescribed, managed, and suppression. Prescribed were planned fires ignited by the managing agency. Managed fire were unplanned natural ignition (lightning) fires where land managers had determined conditions were advantageous for ecological benefit and the fires were not fully suppressed. Suppression fire was any fire managed for full and immediate stop of the fire. Wildland fire was assessed at statewide for California (42 million ha), regional (3.97 million ha), and local level (862,400 ha). Local wildland fires were defined to include subsets of the Mill Creek (32,400 ha), Dry Creek (19,900 ha), Kaweah River (147,100 ha), and Kings River (396,800 ha) drainages and an area west of the drainages (foothills) defined as those within 50 km (266,300 ha). Regional fires included all fires within a bounding box (Latitude -120, -118; Longitude 35.7, 37.7) as seen as the area of enlargement in Figure 4.2. Pre-historic area burned was calculated using high and median fire return intervals for California vegetation types established by Stephens et al. (2007) for vegetation types in all local areas and regionally.

Data handling

BAM protocol for data quality were performed usually once every two weeks and included integrity of the flow (leak check), temperature, and flow checks and calibration as necessary. Hourly data was considered valid if there were no errors logged by the instrument and all audits were passed. Hourly data was validated by ensuring internal relative humidity (RH) was at or below the internal threshold set at 40% with and flow was 16.7 +/- 5% lpm. To correct for the noise band of several micrograms on the BAM (Met One Instruments, 2008a), the occasional negative values for the BAM were set as zero for all calculations. Daily (24 hour) averages required a minimum of 16 valid hours.

Air quality in this paper is determined by using hourly and 24 hour impacts as defined by the Air Quality Index (AQI) for $PM_{2.5}$. The 6 AQI categories are good, moderate, unhealthy for sensitive groups, unhealthy, very unhealthy, and hazardous are used to indicate the level of air quality for $PM_{2.5}$. EPA 24-hour breakpoints for AQI are: 0-12, 12.1-35.4, 35.5-55.4, 55.5-150.4, 150.5-250.4, 250.5-500 $\mu g\ m^{-3}$. The 24-hour AQI is used with daily mean concentrations from the Pinehurst BAM. California Office of Environmental Health and Hazard Assessment for public health officials (Lipsett et al., 2013) 1-3 hour average breakpoints for AQI (0-38, 39-88, 89-138, 139-351, 352-526, >526 $\mu g\ m^{-3}$) are used for hourly measurements.

National Ambient Air Quality Standards (NAAQS) for $PM_{2.5}$ established by the U.S. Environmental Protection Agency (EPA) are used determine public health impacts. NAAQS are threshold levels of $PM_{2.5}$ designed to protect public health where a location is in “attainment” if below the threshold and “non-attainment” if above. NAAQS thresholds for $PM_{2.5}$ (3 year average annual mean of 12.0 $\mu g/m^3$; 3 year mean 98th percentile 35 $\mu g/m^3$) were calculated using the *Guideline on Data Handling Conventions for the PM NAAQS* (U.S. Environmental Protection Agency, 1999).

NAAQS data handling for $PM_{2.5}$ require a minimum daily samples each quarter of the year for accurate representation for the entire year for a rolling 3 year mean being the standard. Annual mean is a mean of all days while the 98th percentile is calculated as a rank value where daily concentrations over a given year are ranked from highest to lowest. The 98th percentile is then determined dependent on the number of daily samples obtained. For clarity, the NAAQS annual (3 year annual mean) and the NAAQS 24 hour (3 year mean of the 98th percentile) are defined as the calculated federal standards while the annual mean and annual 24 hour are the individual year average and 98th percentile respectively.

Air quality and public health impacts are often confounded when determining wildland fire smoke impacts. Visibility and the ability to smell smoke at low levels of exposure can easily bias public health concerns. While other pollutants are of concern (e.g. carbon monoxide in the immediate areas of burn and ozone formation through plume chemistry), we use $PM_{2.5}$ because of its dominance as an air quality pollutant in wildland fire smoke. In this paper we define air quality as short term impacts to health. We use $PM_{2.5}$ AQI to determine these impacts or nuisance levels (e.g. a given hour of the day may be impacted by smoke and be unhealthy over a week and change exercise patterns) where the AQI reporting system can help protect human health. Public health is defined as impacts to NAAQS $PM_{2.5}$ thresholds. These thresholds are used to define exposure limits to quantify impacts to human health where levels are sustained and consistent enough to adversely impact public health.

Smoke impacts to $PM_{2.5}$ are evaluated using impacts to air quality and public health from local, regional, and statewide wildland fires at Pinehurst. Relative importance of wildland fire is determined by comparing AQI and NAAQS values to fire size and location. Smoke impacts are determined using fire start and end dates, satellite imagery for smoke (MODIS, 2016), National

Oceanic and Atmospheric Administration (NOAA) Hazard Mapping System Fire and Smoke Product (HMS) data (NOAA, 2016), Hybrid Single Particle Lagrangian Integrated Trajectory Model (HYSPLIT) forward and back trajectories (HYSPLIT, 2016), U.S. Forest Service BlueSky modeling framework (BlueSky, 2016), and personal observations. All date sources are used when available to determine the presence or absence of smoke during the sampling period.

Results

Typical patterns of $PM_{2.5}$, meteorology, and transport

Prevailing winds for California are generally from the W or NW for most of the year. Upper air winds at Pinehurst are generally from the west or northwest (Air Resources Lab, 2016). Mountain ranges throughout the state modify upper air patterns of wind direction nearer to the ground level where local circulation patters are often a product of the local terrain. The complex terrain includes river drainages that can be over 1,000 m deep. Terrain generally controls ground level transport with the prevailing wind direction in Fresno from the NW March through November and the E/ESE December through February (WRCC, 2016). Localized transport includes nighttime subsidence of air masses from higher elevation and upslope afternoon winds. HYSPLIT back trajectories from Pinehurst showed a seasonality pattern with transport driven by the upper air patterns. Although there was daily variance, general transport for the Pinehurst area consistently followed these general patterns between the years.

Pinehurst has low concentration of $PM_{2.5}$ throughout the year with maximums typically July through September (Figure 4.3). This pattern of summer high concentrations is in contrast to the higher overall concentrations and winter high pattern of $PM_{2.5}$ in the Central Valley and shows the influence of anthropogenic emissions and relatively stagnant air in the Central Valley that is largely disconnected to $PM_{2.5}$ pollutant transport to Pinehurst (Cisneros et al., 2014). Daily mean concentrations have been typically lowest on Mondays (Figure 4.4) with hourly mean maximum of 10.8 μg m^{-3} at 5 PM (Figure 4.5).

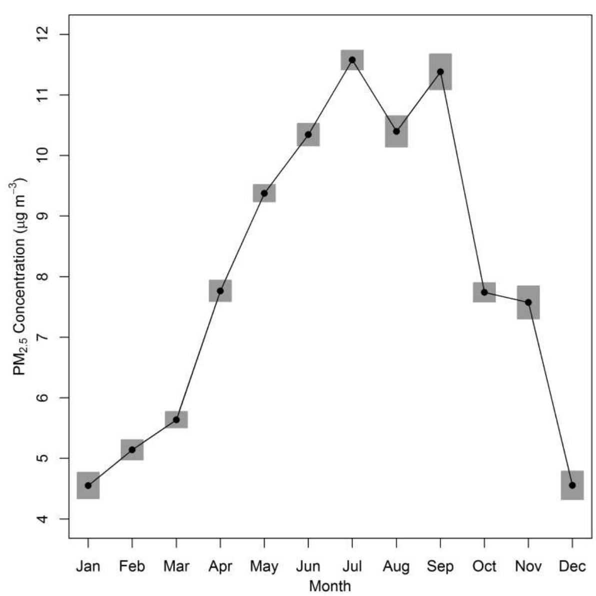


Figure 4.3 Monthly mean concentration of fine particulate matter ($PM_{2.5}$) with 95% confidence intervals of the mean (grey boxes) at Pinehurst, California from 2006 to 2015.

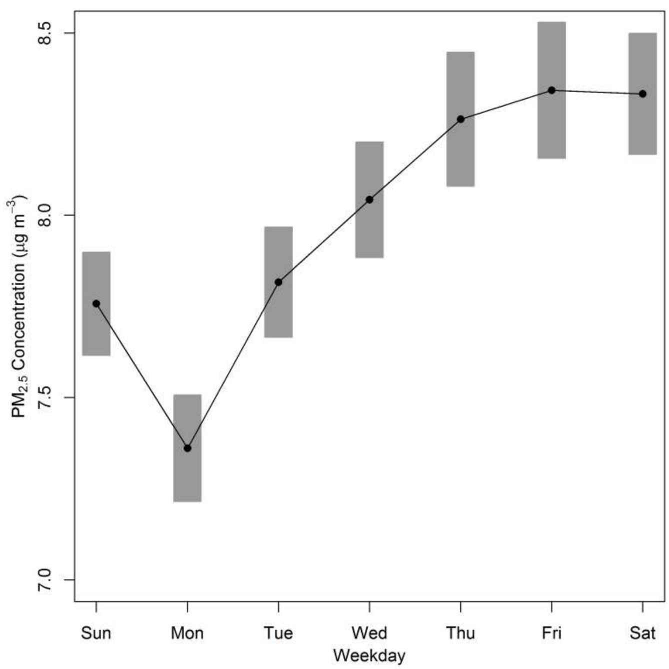


Figure 4.4 Daily mean concentration of fine particulate matter ($PM_{2.5}$) with 95% confidence intervals of the mean (grey boxes) at Pinehurst, California from 2006 to 2015.

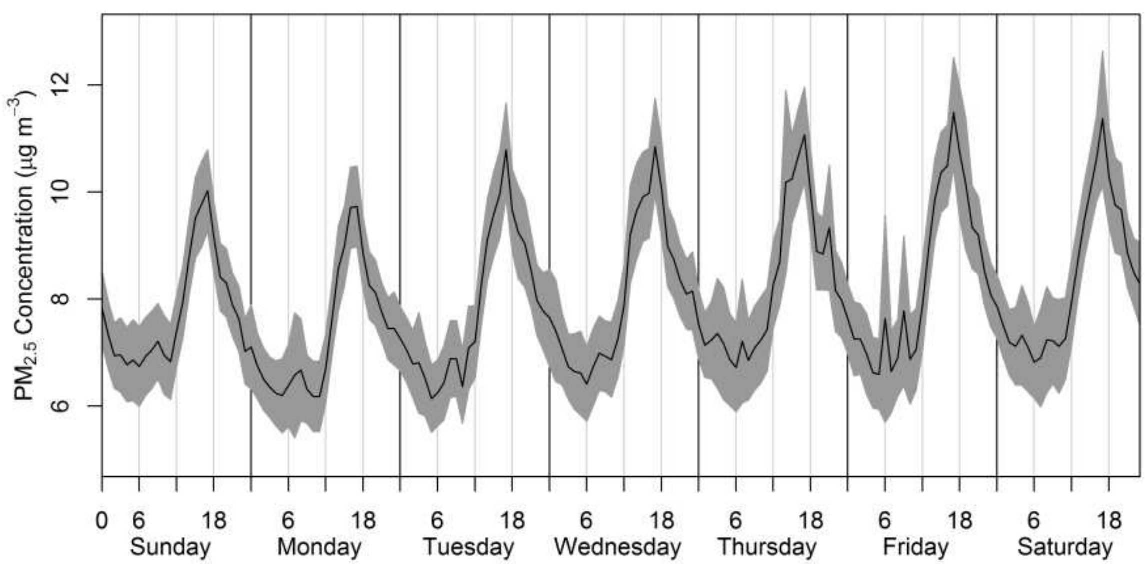


Figure 4.5 Weekday hourly mean concentration of fine particulate matter ($PM_{2.5}$) with 95% confidence intervals of the mean in grey at Pinehurst, California from 2006 to 2015.

These concentrations typical of trends at the Pinehurst site show a seasonal variation where summer air quality is most impacted by $PM_{2.5}$. There is an abundance of wildland fire in and around the Pinehurst area. Fire season in the Sierra Nevada normally begins about mid-summer and ends with the first significant rain or snow event in the late fall or early winter. Wildland fire can occur any month of the year but the typical wet winters and warm dry summers of the Sierra Nevada usually preclude much fire activity in the Pinehurst area until May with fires starting in July (Figure 4.6) and the subsequent area burned emitting smoke through October. In October, with the shorter days, cooler temperatures usually slow fire activity and emissions until winter storms bring increased precipitation and end the fire season (Figure 4.7). Wildland fire smoke was an important contributing source to $PM_{2.5}$ at Pinehurst.

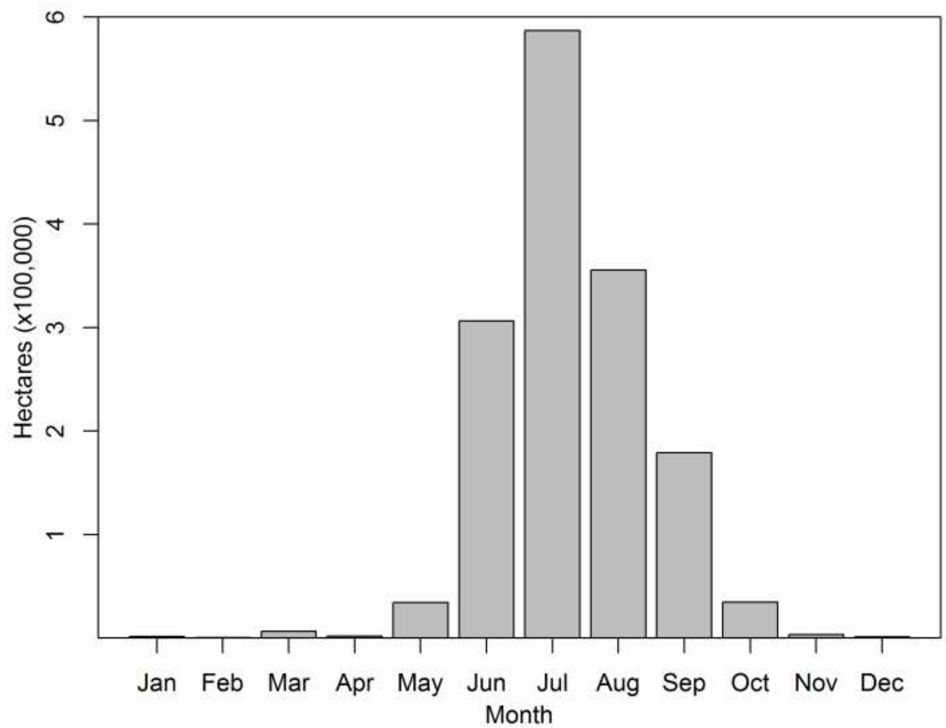


Figure 4.6 Hectares of wildland fire burned in the Pinehurst region from 1970-2014 using the month of the start date.

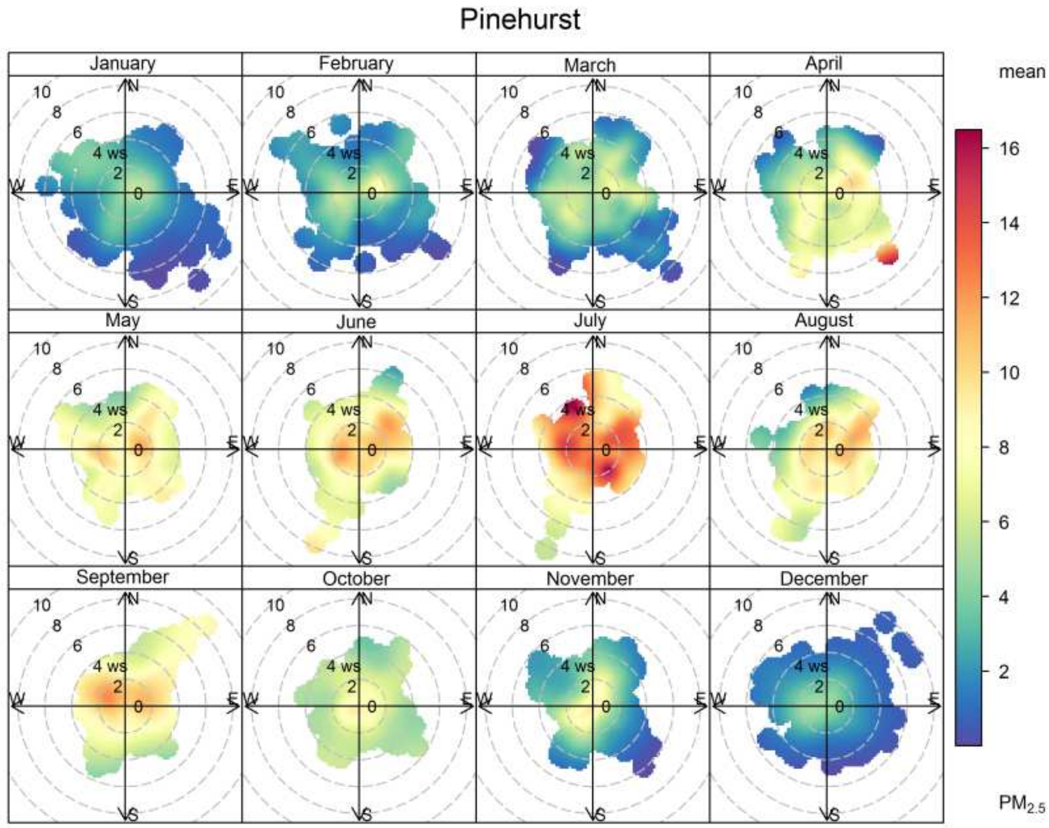


Figure 4.7 Monthly polar plots of mean fine particulate matter ($PM_{2.5}$) using ground level wind speed and wind direction at Pinehurst.

Wildland fire

Wildland fire burns regularly throughout California with a median of 8,095 wildland and 657 prescribed fires from 2006 to 2015. Many of the large federally protected land areas are fire prone and burn regularly. The largest fires in the Pinehurst region were on federal US Forest Service and National Park Service land (Figure 4.8). The years of 2006 to 2015 in California burned from 73,691 ha in 2010 to 604,074 in 2008 (Figure 4.9(a)). While 2008 had the largest total area burned, this overall burn area may be much more typical of the historic cycle (Stephens et al., 2007). 2015 was the highest local burn year for Pinehurst (Figure 4.9(b)) due to the full-suppression Rough Fire (61,360 ha). The local Pinehurst burn area was an order of magnitude below the 2015 year for 2006 to 2014. The 2008 year, in addition to the large overall area burned in California, was the next largest in total area burned (Figure 4.9(c)).

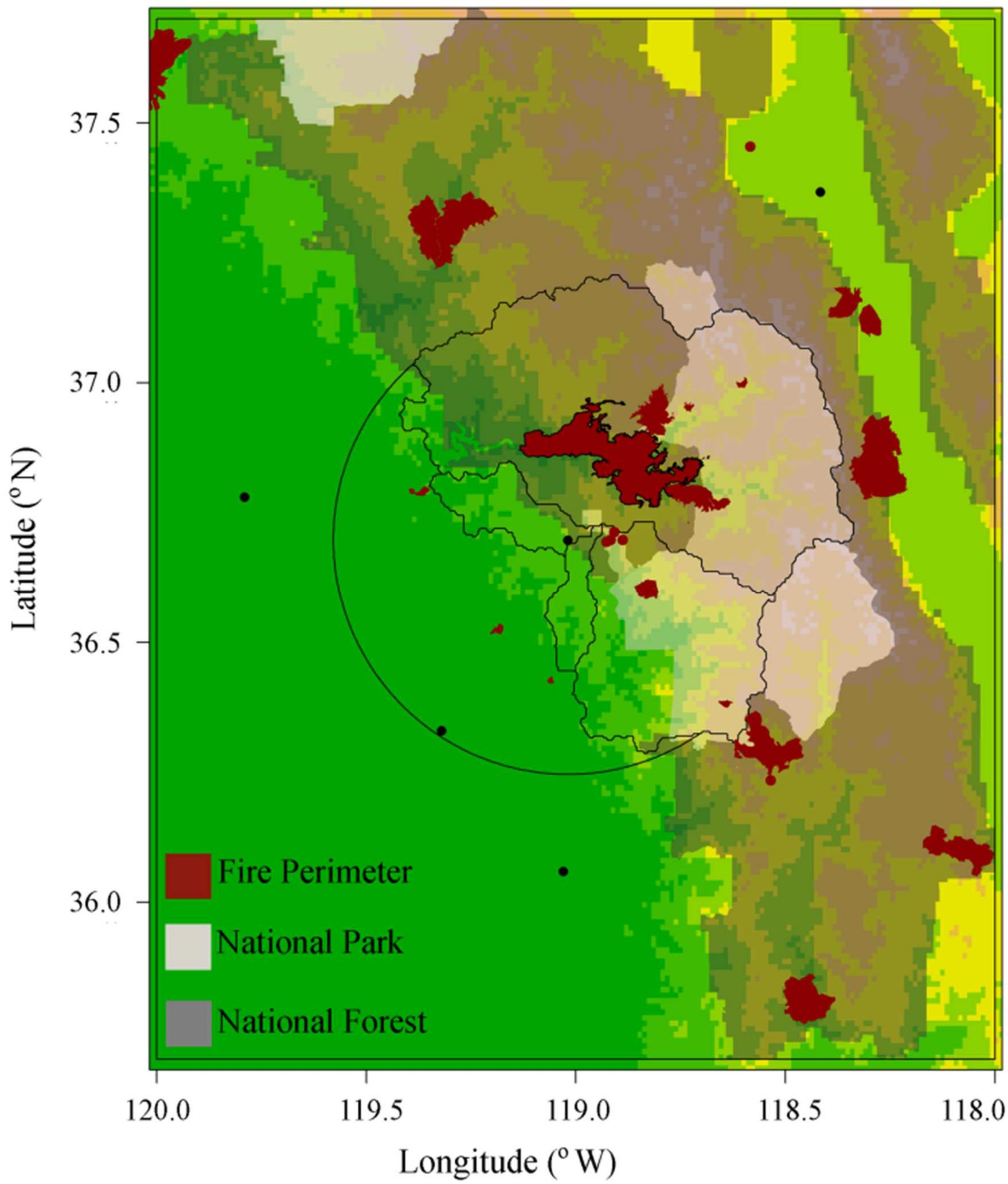


Figure 4.8 Large fires (larger than 200 ha local and the 10 largest regional) in the Pinehurst area.

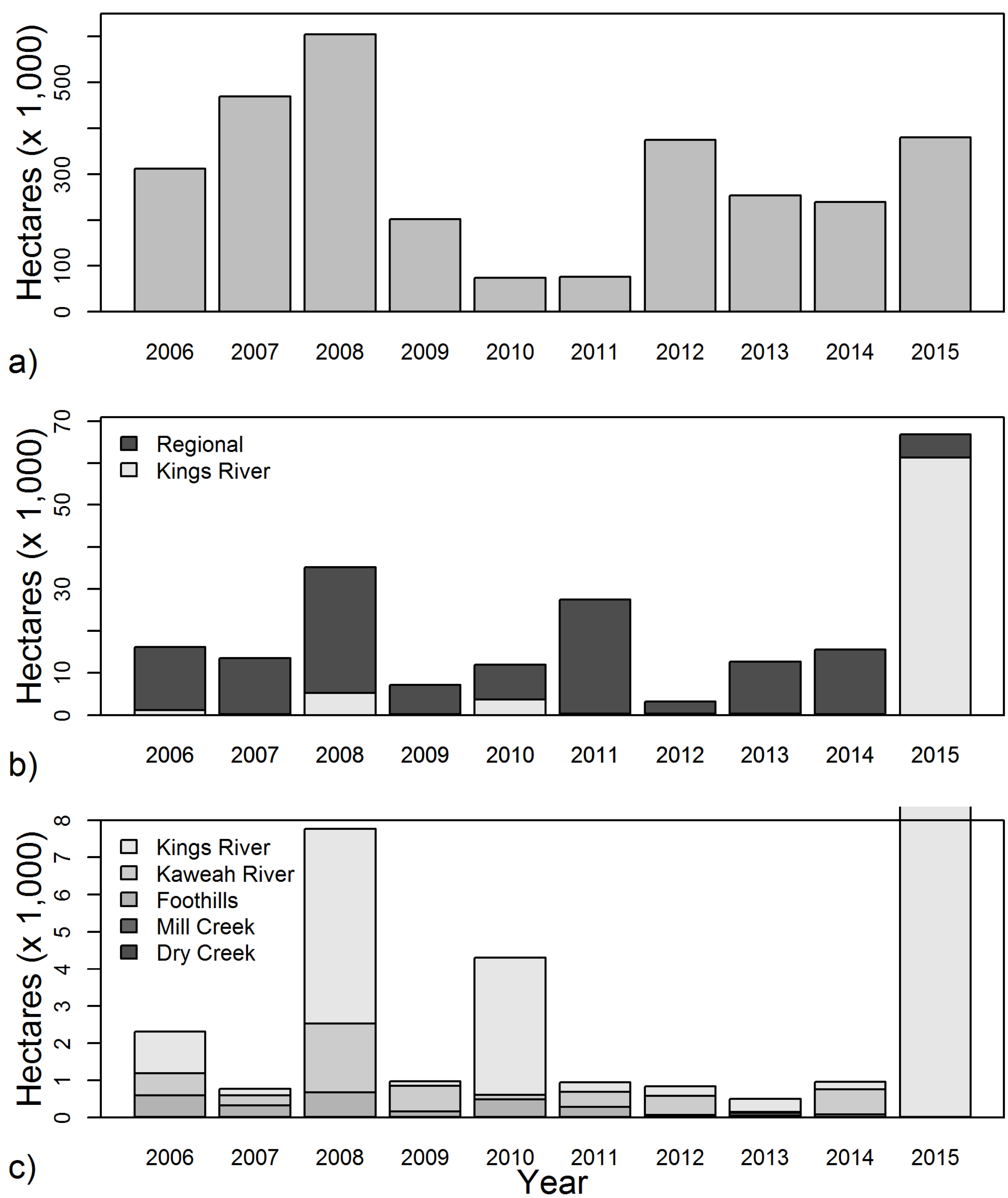


Figure 4.9 Wildland fire burned area (2006 to 2015) for (a) statewide (California), (b) regional to the Pinehurst monitor and also showing the total in the Kings River drainage, and (c) individual local areas; note the burn area in 2015 was primarily in the Kings River drainage and the total amount is shown in (b).

The Pinehurst regional and local area is an active area of wildland fire ignitions (Figure 4.10(a)) where the number of regional fire starts and ignitions in the foothills exceeded 1,000 in both 2006 and 2007. The local federally protected land primarily east of Pinehurst usually had 50-70 annual wildland fires starts (Figure 4.10(b)).

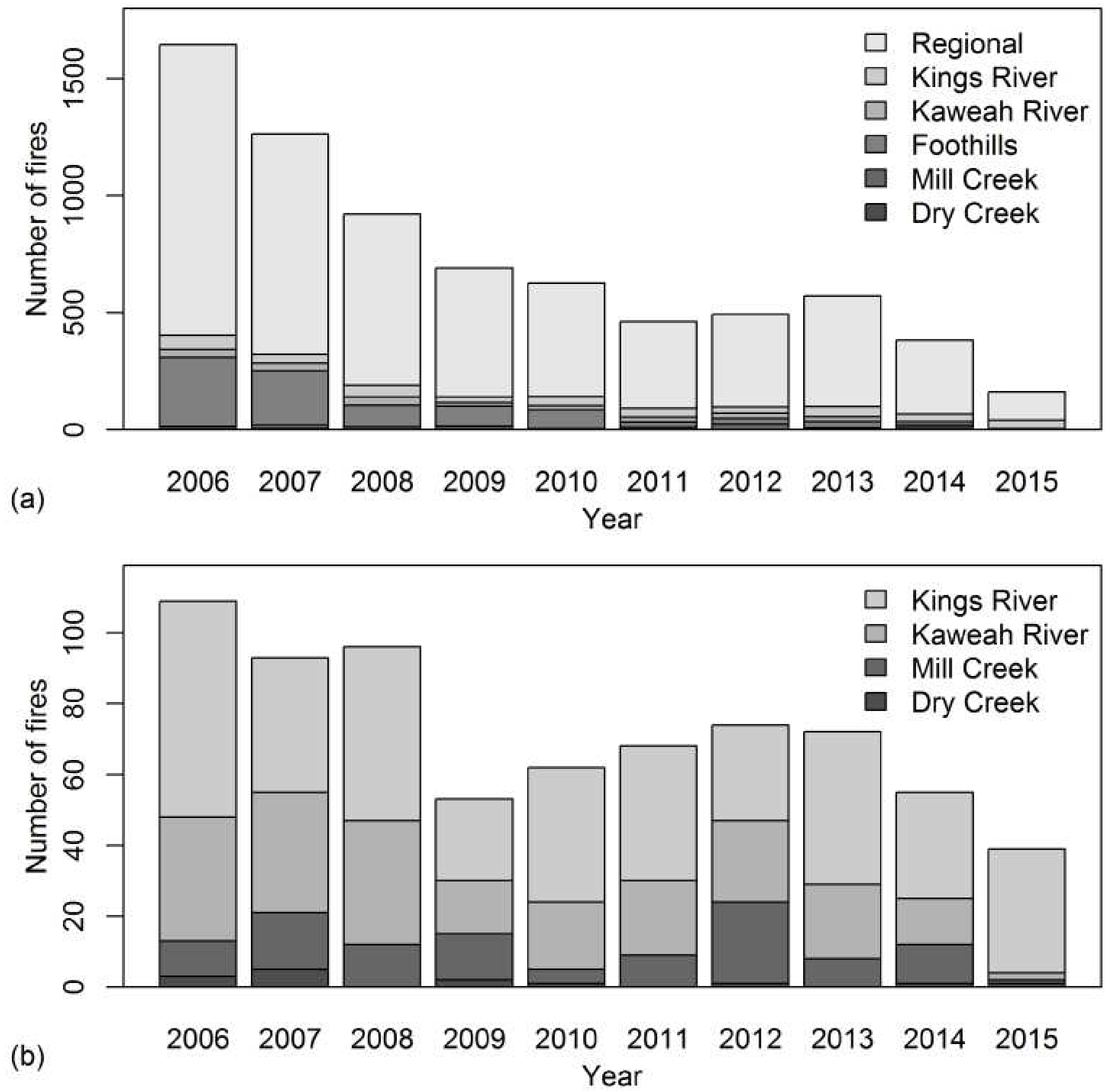


Figure 4.10 Number of wildland fire ignitions from 2006 to 2015 for the (a) region and (b) local areas of primarily forested federal lands.

Both Dry Creek (14 fires) and Mill Creek (107 fires) had no fire over 200 ha and much of the fire in the foothills area (871 fires total) was small with only 3 fires larger than 200 ha (Table 4.1). The Kaweah River drainage had 218 fires between 2006 and 2015 with 8 fires over 200 ha. The Kings drainage had 382 total fires and the largest burned area during the Hidden Fire (Table 4.1). Regionally, there were an additional 61 fires over 200 ha. Four prescribed fires in the Kaweah drainage were the closest to Pinehurst. The Rough and the Hidden Fires suppression fires were similar in closeness. The Sheep Fire was the closest managed fire.

Table 4.1 Ten largest regional and all local larger than 200 ha wildland fires and burned area midpoint location with distance (minimum/maximum) to Pinehurst.

Year	Fire Name	Hectares	Distance (km) to Pinehurst (min/max)	Longitude (W)	Latitude (N)	Local area	Management Action
2012	Whitaker	207	8 (7/9)	118.927	36.694	Kaweah	Prescribed
2011	Redwood Mountain	254	9 (8/10)	118.917	36.698	Kaweah	Prescribed
2006	Upper Redwood	251	10 (9/11)	118.908	36.713	Kaweah	Prescribed
2009	Hart	329	12 (11/13)	118.888	36.698	Kaweah	Prescribed
2015	Rough	61,360	20 (14/35)	118.911	36.859	Kings	Suppression
2008	Hidden	1,491	20 (17/23)	118.829	36.603	Kaweah	Suppression
2010	Stokes	356	24 (23/26)	119.186	36.525	Foothills	Suppression
2010	Sheep	3,650	28 (23/32)	118.722	36.783	Kings	Managed
2006	W	243	30 (29/31)	119.060	36.427	Foothills	Suppression
2008	Tehipite	4,693	33 (28/39)	118.812	36.942	Kings	Managed
2008	Avocado	445	33 (31/36)	119.371	36.791	Foothills	Suppression
2006	Roaring	665	33 (31/36)	118.656	36.769	Kings	Managed
2008	Cedar Bluffs	407	34 (33/36)	118.651	36.780	Kings	Managed
2006	Burnt	254	38 (37/40)	118.731	36.954	Kings	Managed
2008	Davenport	316	40 (38/41)	118.687	36.450	Kaweah	Prescribed
2014	Mosquito	601	45 (43/47)	118.630	36.445	Kaweah	Prescribed
2009	Horse	269	49 (47/50)	118.643	36.383	Kaweah	Managed
2013	Windy Peak	276	50 (49/51)	118.602	36.999	Kings	Suppression
2011	Lion	8,296	61 (54/68)	118.543	36.300	Regional	Managed
2015	Cabin	2,817	67 (65/69)	118.536	36.235	Regional	Managed
2007	Inyo Complex	2,614	68 (61/73)	118.280	36.848	Regional	Suppression
2013	Aspen	9,305	72 (64/77)	119.277	37.308	Regional	Suppression
2014	French	5,600	73 (65/80)	119.355	37.298	Regional	Suppression
2011	John	2,166	79 (77/82)	118.298	37.118	Regional	Suppression
2015	Round	2,644	92 (90/95)	118.584	37.454	Regional	Suppression
2008	Clover	6,192	106 (99/114)	118.088	36.104	Regional	Suppression
2010	Bull	6,654	111 (105/116)	118.449	35.812	Regional	Suppression
2008	Telegraph	13,796	134 (127/141)	120.016	37.606	Regional	Suppression

Smoke Transport

In addition to general smoke transport patterns, smoke in the Sierra Nevada is dependent on drainage transport. Drainage transport was important to smoke impacts at Pinehurst. Smoke from wildland fires in the Pinehurst local area often settled into drainages during the evening and night as fire intensity reduced. Satellite imagery, HYPSLIT trajectories, and HMS polygon data often helped in the understanding of general smoke transport from local fires. Wildland fire in the Sierra Nevada often moved north, south, and west of the fire as smoke followed generally transport east as illustrated by the managed 2011 Lion Fire south of Pinehurst (Figure 4.11). Similarly, the 2010 Sheep Fire was widespread across the Sierra Nevada (Figure 4.12). HMS data also indicated some of the drainage transport with this fire (Figure 4.12(a)) and the complexity of satellite imagery being used to determine source contribution to ground impacts (Figure 4.12(b)) where the Avalanche Fire in Yosemite National Park appeared to combine. The 2015 Rough Fire and the larger emissions from this megafire, as seen in HMS and satellite imagery, obviously increased spatial smoke impacts (Figure 4.13). This figure also illustrates the variance that can be seen in the modelled (HYSPLIT) and remote sensing (HMS) data.

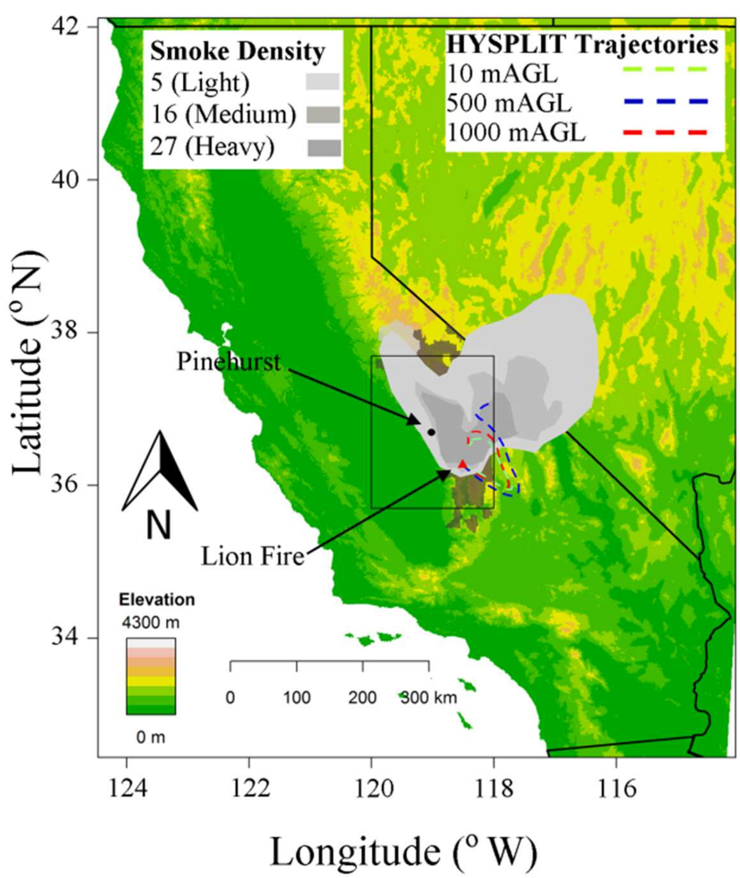


Figure 4.11 Smoke transport as described by the Hazard Mapping System Fire and Smoke Product (HMS) smoke density and Hybrid Single Particle Lagrangian Integrated Trajectory Model (HYSPLIT) forward trajectories from the Lion Fire on 7/26/2011.

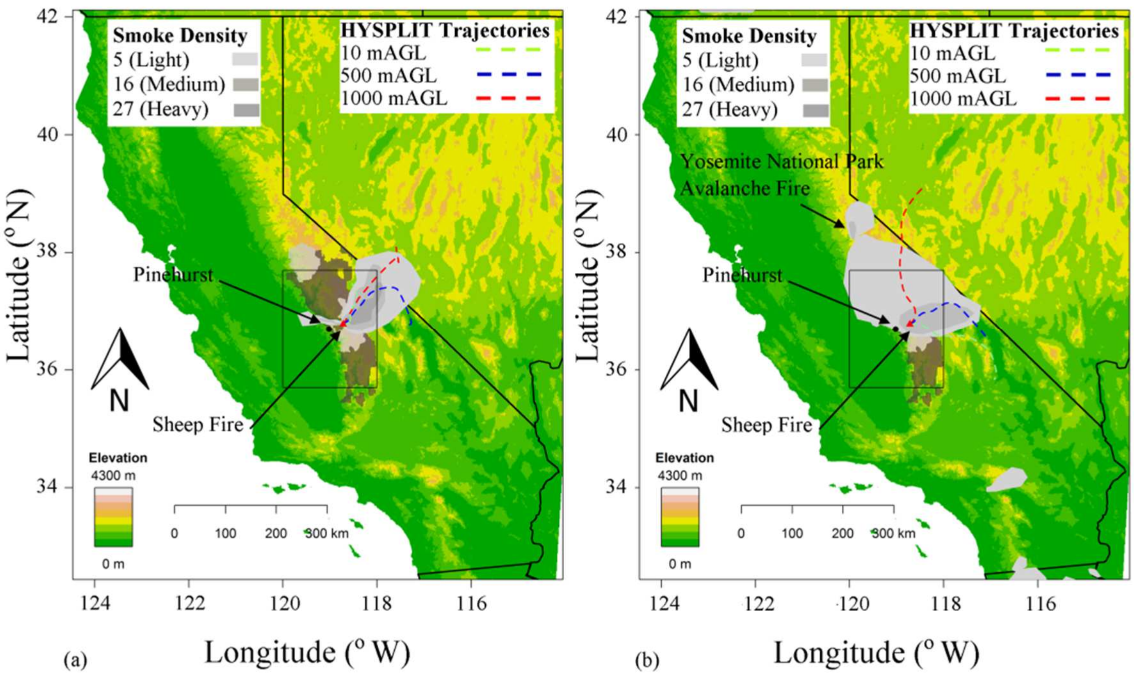


Figure 4.12 Smoke transport as described by the Hazard Mapping System Fire and Smoke Product (HMS) smoke density and Hybrid Single Particle Lagrangian Integrated Trajectory Model (HYSPLIT) forward trajectories from the Sheep Fire on (a) 8/17/2010 and (b) 9/3/2010 showing additional regional smoke from the Avalanche Fire in Yosemite National Park.

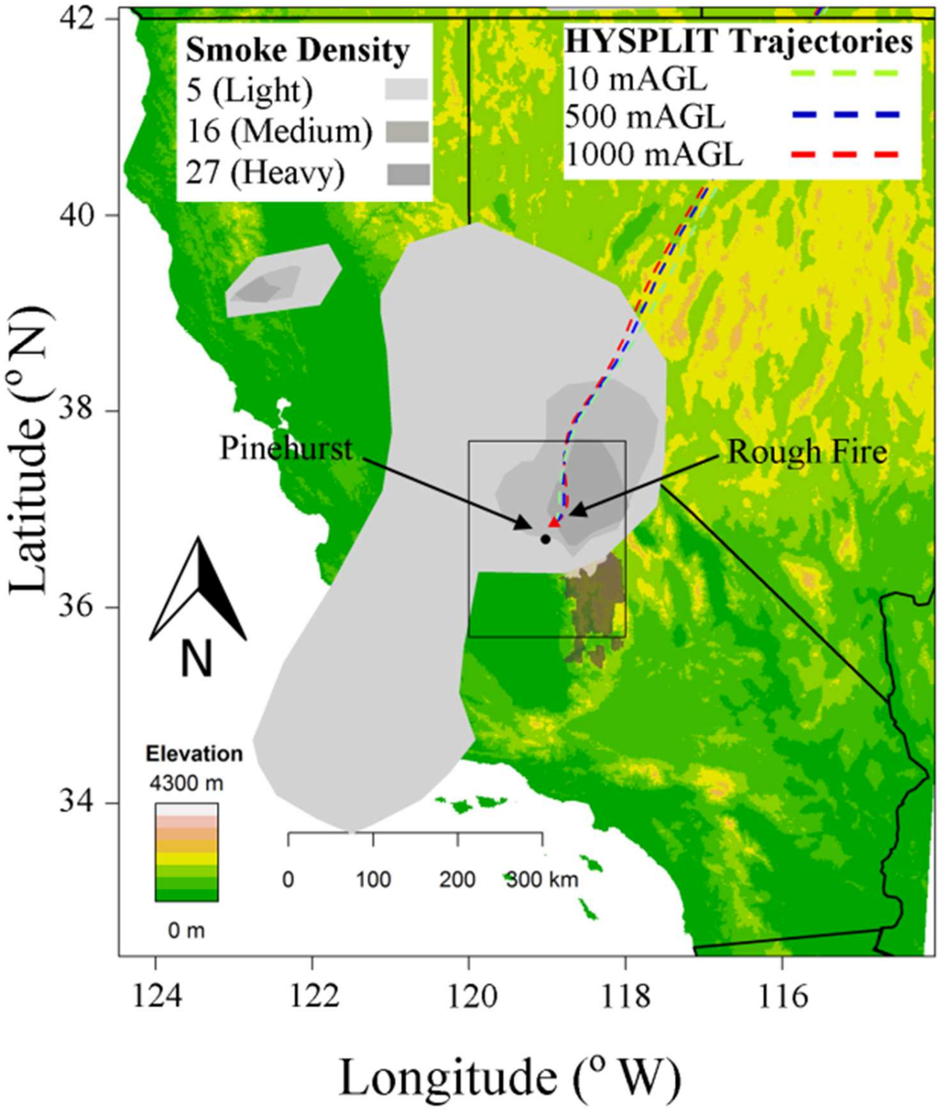


Figure 4.13 Smoke transport as described by the Hazard Mapping System Fire and Smoke Product (HMS) smoke density and Hybrid Single Particle Lagrangian Integrated Trajectory Model (HYSPLIT) forward trajectories from the Rough Fire on 9/2/2015.

Satellite imagery and modelling was especially helpful to assess smoke from the larger (~>1,000 ha) managed and full suppression fires. Typically the greatest impacts were largely found to be east of these fires. Because east of these fires was largely federally protected land until the Owens Valley (e.g. city of Bishop), smoke was primarily denser over the Pinehurst site and although Pinehurst was west, for all local fires excluding the Rough Fire, the impacts tended to be at or below sites on the east side of the Sierra Nevada (Preisler et al., 2015; Schweizer and Cisneros, 2014). The impacts from smoke were likely larger at select sites for each fire with Pinehurst regularly exhibiting the presence of smoke.

Vegetation

Land use and vegetation in the Pinehurst region was primarily developed for agriculture (27%) with Ponderosa (9%) and mixed conifer (9%). The western foothill region was primarily agricultural (57%) with 31% Blue Oak and Valley Oak woodland. Moving up in elevation, Dry and Mill Creek were similar with primarily Blue Oak and Valley Oak (40% and 37% respectively) and both having 37% Chaparral. The Kings and Kaweah River drainages, encompassing mid- to high-elevation federally protected lands included more timber. The Kaweah River drainage was found to have 37% Ponderosa Pine, Chaparral (24%), and Red Fir (22%). The Kings River drainage major vegetation types were Red Fir (36%) and Ponderosa Pine (15%).

Pre-historic fire was estimated to be between 35,300 ha and 92,400 hectares annually for the Pinehurst local area with the regional fire from 132,600 to 347,900 ha annually (Table 4.2). Wildland fire in this area is well below the estimated pre-historic levels. Although 2015, due to the Rough Fire, exceeded the high fire return interval estimate for local fires but was only 66% of the median pre-historic fire extent, regionally, even with the Rough Fire, total fire size was below pre-historic estimates. Additionally, the Rough Fire contributed 2 to over 5 times the expected fire area for the Kings River drainage for this year illustrating the exceptional conditions that led to this individual fire when compared to the pre-historic normal. Other years were all well below the calculated typically below 3% of the least expected. 2008 with managed fires in the Kings and Kaweah River drainages were 8-22% of the expected. The other high local fire years (2006 and 2010) were between 4-10% and 5-12% of calculated fire area.

Table 4.2 Estimates of pre-historic burned areas for the Pinehurst area in ha.

Bounding area	High fire return	Median fire return
Foothills	13,600	36,200
Dry Creek	1,500	3,800
Mill Creek	2,400	6,100
Kaweah River	6,400	16,200
Kings River	11,400	30,100
Total Local	35,300	92,400
Regional (excluding local)	97,300	255,500
Total Regional	132,600	347,900

$PM_{2.5}$ and AQI at Pinehurst

Hourly AQI for $PM_{2.5}$ at Pinehurst is largely good with 2015 having the worse hourly AQI and 2008 having the most unhealthy hours (Table 4.3). The highest hourly AQI at Pinehurst were almost exclusively caused by wildland fire with the large high-intensity full suppression

Rough Fire having the largest impact both in quantity and level of air quality impact. The managed local medium sized Sheep and Tehipite Fires also had impacts to air quality but were much less in both number of hours and level of impact. Local smaller prescribed fires including the nearest significant burns (Whitaker Rx, Redwood Mt., Upper Redwood, and Hart) had limited impacts with hourly $PM_{2.5}$ AQI typically good and nothing higher than moderate. These prescribed burns contributed to a total of 10 moderate hours.

Table 4.3 Hourly Air Quality Index (AQI) at Pinehurst from 2006 to 2015.

Year	NA	Good	Moderate	Unhealthy for Sensitive Groups	Unhealthy	Very Unhealthy	Hazardous	Max
2015	808	7749	163	35	4	1	0	455
2014	229	8431	95	5	0	0	0	110
2013	731	8009	19	1	0	0	0	95
2012	501	8251	29	1	2	0	0	185
2011	162	8557	41	0	0	0	0	64
2010	278	8451	30	1	0	0	0	136
2009	2084	6639	36	0	1	0	0	157
2008	193	8393	187	3	8	0	0	379
2007	609	8077	74	0	0	0	0	80
2006	1291	7431	37	1	0	0	0	103

The highest hourly AQI from 2006 to 2015 was very unhealthy. The highest hourly concentration of $PM_{2.5}$ (455 $\mu g\ m^{-3}$) was on 8/28/2015 and caused by the Rough Fire. The Rough Fire accounted for all the 2015 hourly AQI that were unhealthy for sensitive groups, unhealthy, and very unhealthy and additionally accounted for 137 of the Moderate hourly readings for the year. Hourly AQI was good during both large regional fires (Round and Cabin) during 2015. Neither the Round Fire to the northeast or the Cabin Fire to the south-southeast impacted Pinehurst with hours during this fire all good AQI. During 2008, all hourly unhealthy values and 2 unhealthy for sensitive occurred during the Tehipite Fire. The Tehipite Fire also contributed to the increased $PM_{2.5}$ at Pinehurst resulting in moderate AQI for 62 hours. Statewide in California 2008 was an active fire year with smoke covering most of the state. Before the start of the Tehipite Fire, large fires in northern California created high $PM_{2.5}$ in many areas across the state during late June (25-30) as the smoke moved more northerly $PM_{2.5}$ returned to normal but increased again in the Pinehurst area from about July 7 to 11. This June statewide smoke was associated with 70 moderate and 1 unhealthy for sensitive hourly $PM_{2.5}$ AQI while the July episode added 24 moderate hours. The Sheep Fire (2010) caused the 1 unhealthy for sensitive and contributed to 23 of the 30 moderate AQI hours.

Daily AQI was generally good with the highest AQI being unhealthy for sensitive $PM_{2.5}$ AQI (Table 4.4). Again the Rough Fire had the largest impact to AQI with 11 unhealthy for sensitive and 22 moderate days. The Tehipite Fire smoke contributed to 1 unhealthy for sensitive and 34 moderate days. Air quality for $PM_{2.5}$ during the Sheep Fire never went above moderate 24-hour AQI (42 days). The June 2008 period produced 3 moderate and 3 unhealthy for sensitive days with no days of good 24-hour AQI. The July 2008 period of smoke also had no good days with 4 moderate a 1 unhealthy for sensitive. Local prescribed and smaller fires did not go above moderate and typically remained in the good category. Larger fires in the local area (Rough, Tehipite, Sheep, Lion, and Aspen) all impacted Pinehurst and often increased the 24-hour AQI to moderate.

Table 4.4 Daily (24-hour) Air Quality Index (AQI) at Pinehurst from 2006 to 2015.

Year	NA	Good	Moderate	Unhealthy for Sensitive Groups	Unhealthy	Very Unhealthy	Hazardous	Max
2015	34	267	53	11	0	0	0	53.5
2014	12	274	78	1	0	0	0	45.7
2013	33	263	69	0	0	0	0	23.7
2012	23	283	59	1	0	0	0	36.4
2011	9	273	83	0	0	0	0	25
2010	36	239	90	0	0	0	0	24.4
2009	54	285	26	0	0	0	0	29.8
2008	10	269	82	5	0	0	0	53.9
2007	35	285	44	1	0	0	0	43.3
2006	68	248	49	0	0	0	0	34.2

Impacts from smoke to $PM_{2.5}$ air quality at Pinehurst were generally low for smaller fires (less than ~1,500 ha) throughout even the immediate area. The larger managed fires (Sheep and Tehipite) began impacting air quality noticeably. Smoke impacts to $PM_{2.5}$ were largely over a few hours (1-6) at the same time of day. The large high-intensity full suppression Rough Fire was distinctly worse for air quality including the only time Pinehurst was at very unhealthy for $PM_{2.5}$ which manifested in more unhealthy for sensitive days than all other years combined.

$PM_{2.5}$ and NAAQS Health Standards at Pinehurst

Because it falls within the regulatory district of the SJVAPCD, Pinehurst is in non-attainment for NAAQS health standards for $PM_{2.5}$. Higher elevation and less urbanized and developed federally protected land surrounding Pinehurst make it dissimilar to most of the regulatory defined SJVAPCD air basin where anthropogenic emission dominate $PM_{2.5}$ levels. NAAQS annual and NAAQS 24 hour are both in attainment at Pinehurst from 2006 to 2015 (Figure 4.14). NAAQS annual is consistently below the federal threshold with 2014 having the highest NAAQS annual (8.5 µg m^{-3}) and annual mean (9.3 µg m^{-3}). NAAQS 24 hour is also below the federal compliance threshold with the highest being 2015 (30 µg m^{-3}). The annual 24 hour for 2014 at 43.9 µg m^{-3} is the only year above the federal threshold. For Pinehurst to remain in compliance for $PM_{2.5}$ in 2016, the annual 24 hour cannot exceed 34.8 µg m^{-3}. For 2017 to remain in compliance both 2016 and 2017 need average 30.5 µg m^{-3} or less.

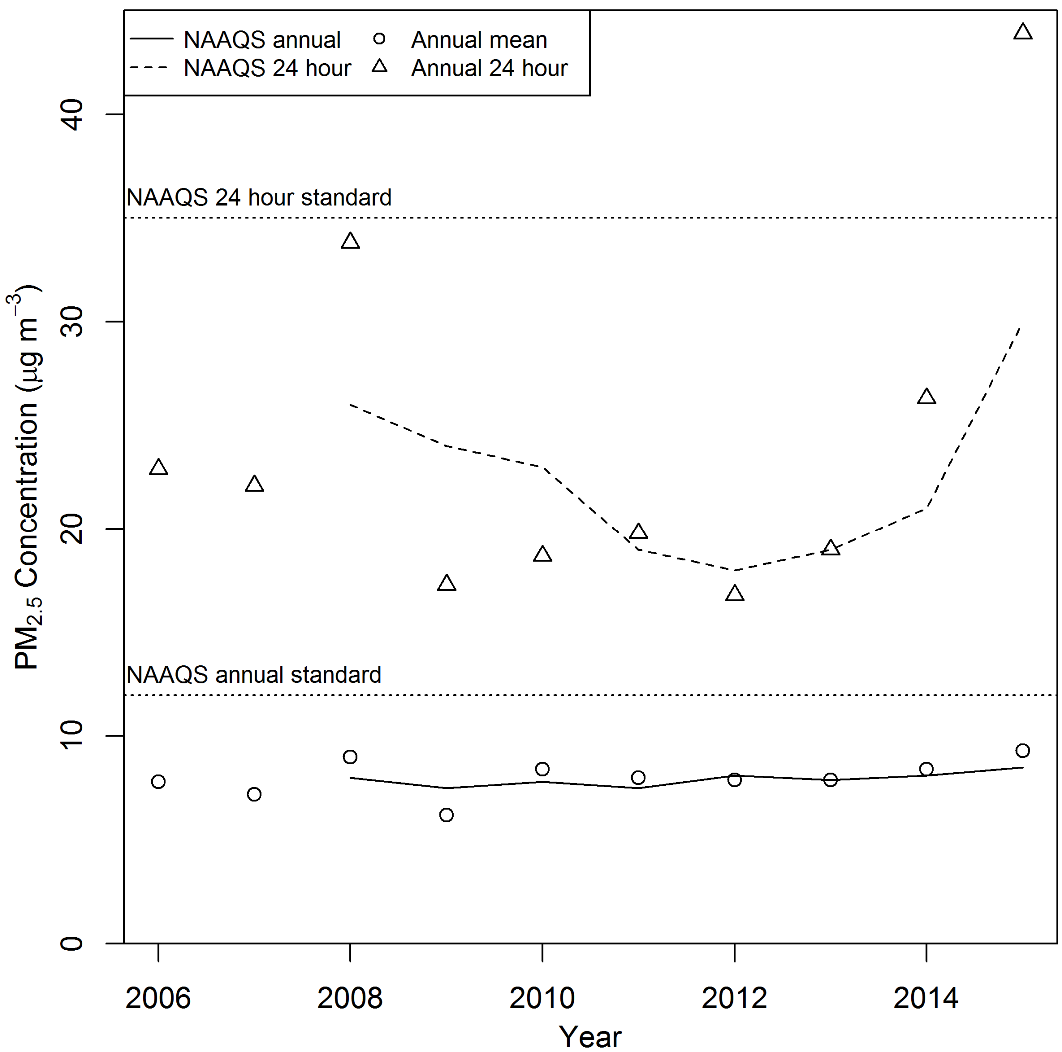


Figure 4.14 Fine particulate matter ($PM_{2.5}$)at Pinehurst for annual and 24 hour compliance to federal attainment standards.

Daily concentrations of $PM_{2.5}$ exceeded the NAAQS 24 hour standard (35 µg m^{-3}) 19 times from 2006 through 2015. The statewide smoke in California during 2008 accounted for 4 of these days while the Rough Fire for 11. The managed Tehipite Fire (2008) caused one while the other 3 were not associated with wildland fires.

Compliance with NAAQS 24 hour is an excellent indicator of smoke at Pinehurst. Smoke accounted for 39% of the highest 10 days for each year. The Rough Fire accounted for the all of the highest 10 days during 2015. Smoke from the Sheep Fire (2010) was associated with number 1, 2, 4, and 5 highest days. The Tehipite Fire (2008) accounted for 2 in the top 10 while the statewide smoke during this year caused 7. The only smoke from a prescribed fire in the highest days was from the Redwood Mountain burn in 2011 (2) and also had additional smoke input from the Lion Fire but the highest single day for 2011 was 25.0 µg m^{-3}. Other prescribed fires in the immediate area did not contribute to the highest $PM_{2.5}$ days at Pinehurst.

Discussion

Wildland fire is a difficult natural process to manage. Smoke from wildland fire is a complex component to using fire for ecological benefit and fuel reduction. Suppression of fire in the Sierra Nevada has led to increased fuel build-up and contributed to an increased number and severity of megafires (Williams, 2013). Smoke impacts from large high intensity full suppression fires such as the 2013 Rim (Peterson et al., 2015) and 2002 McNally (Cisneros et al., 2012) in the Sierra Nevada suggest these fires can be expected to widely distribute significant amounts of smoke that impact large populations. Tradeoffs between release of emissions through these high intensity fires versus release from self-regulated fire more typical of this forest system (Parks et al., 2015) must be confronted if the impacts to human health are to be accurately assessed.

We have attempted to analyze smoke impacts to air quality using a single site over multiple years. Overall air quality for $PM_{2.5}$ was found to be good for this site even though it is in regulatory non-attainment for this pollutant. This highlights the variance in an air district and the general benefits to air quality that distance from anthropogenic emissions and a protected natural area can provide. This fire prone area could include managed wildland fire to levels at or above the largest years in this study. Smoke from the Tehipite Fire in 2008 and the Sheep Fire in 2010 impacted the Pinehurst site and had larger impacts in more immediate areas adjacent to the fire and in the Kings River drainage but these impacts were largely only unhealthy for sensitive groups. Given that these fires were primarily in federally protected forests with little infrastructure or permanent residences, impacts were limited. These ecologically beneficial fires helped sustain this fire adapted forest and reduced fuels with measured emissions and included subsequent benefits to public health from a healthy and resilient ecosystem. In contrast, the Rough Fire (2015) increased air quality impacts to $PM_{2.5}$ to very unhealthy levels and led to an exceedance of the annual NAAQS 24 hour. Emissions slowed when the Rough Fire entered the 2010 Sheep Fire perimeter (Figure 4.8). Fire intensity dropped as the Rough Fire crept through this previously burned area and limited intensity and spread. Thus, impacts dropped in Pinehurst with a number of days of good AQI for $PM_{2.5}$. Later, fire became more active again in areas of higher fuel loads (caused by a century of fire suppression) and $PM_{2.5}$ again increased in Pinehurst to unhealthy levels. This drop in impacts at the Pinehurst site during the Rough Fire and the implications to air quality from previous recent ecologically beneficial fire needs further study.

There are inherent difficulties to determining smoke impacts on air quality from wildland fire. Smoke from any fire will have impacts. These impacts are largely dependent on location of the source and the receptor. Being in direct immediate contact with smoke even from a campfire will exceed air quality standards to protect human health. Wildland fire smoke is no exception. Indeed, being directly in the plume of a wildland fire of even the smallest size exposes a person to many pollutants. The Pinehurst site was never directly in a plume. Rather, we chose this site because it was not in the plume but is representative of local communities' exposure to $PM_{2.5}$.

We have also focused on $PM_{2.5}$ because it is a significant component of smoke and there is widespread availability of data for this pollutant. Other pollutants such as organics, ozone, or carbon monoxide are also important but are much more subjected to plume dynamics and chemistry and do not have such an easy and direct relationship with smoke impacts. Pinehurst was chosen to represent a typical location in a fire prone ecosystem at the wildland urban interface. Additionally, being in an area of regulatory non-attainment adds to the complexity of policy discussion and implications. Our intentions were to look at smoke impacts over time and various fire management actions.

Smoke impacted air quality at Pinehurst. Hourly AQI reached very unhealthy on one of the largest growth days (8/28/2015) of the Rough Fire when fire size increased approximately 2,900 ha. Smoke most definitely will impact the most sensitive groups but we found during a typical prescribed or managed fire, smoke impacts were largely avoidable due to consistency of timing and overall lower exposure to $PM_{2.5}$. Impacts from prescribed and managed fires could largely be mitigated through personal choices. The high severity full suppression Rough Fire smoke exposure did not provide this luxury. AQI was the highest and much more pervasive throughout the day. Although this is intuitive in that the Rough Fire was much larger, and therefore emitted substantially more smoke, lost is the idea that had the local Pinehurst area burned more nearly to the pre-historic normal this episode may have been avoided.

It is difficult to parse out health impacts and nuisance when determining smoke impacts. Additionally, it is much easier to urge full suppression when confronting any smoke impact. But, this tact assumes that these emissions will not come. In a fire prone area with a fire adapted ecosystem this assumption is flawed. Increased fire size and intensity in the western United States is the new normal (Westerling et al., 2006) as we move into a post suppression era. The Sierra Nevada will have more smoke in the coming years. These emissions can likely be mitigated by increased burning and a slower release of smoke rather than choosing a megafire through suppression policies.

Exceedance of the annual 24 hour for 2015 illustrates the type of public health impacts to be expected when full suppression fails to contain a wildland fire with heavy fuel loading and other extreme conditions. Burning in the Pinehurst area could easily be increased while still adhering overall to NAAQS $PM_{2.5}$ thresholds. Air and fire management in a fire prone area particularly one with large tracts of wilderness or other protected natural areas should consider managing to a local basin where a sensitive site or, as Pinehurst, a site located near the wildland urban interface may be used to estimate landscape level air quality. This would help in the assessment and weighing of impacts from a managed fire while allowing regulators to have a metric to decide between public health (NAAQS) and nuisance (AQI, visibility, smell).

Conclusions

Effective wildland fire policy and management is complex. There is no simple solution. Smoke from wildland fire will likely become increasingly contentious particularly in the public forum as long term benefits of managed and prescribed fire meet the short term benefits of suppression for an increasing population. Wildland fire in the Pinehurst area is well below historic levels and could be increased using prescribed and managed fire. Prescribed and managed fire exposes the Pinehurst area to less particulate matter that is more avoidable and of shorter duration than a megafire when full suppression fails. For large protected natural areas, a landscape level understanding of air quality impacts would help assess the tradeoffs and aid in management of fire prone areas.

Chapter 5.0 Conclusions and policy recommendations

The ecosystem benefits of smoke in a fire prone environmental system need to be further studied. Just as fire helped to form the Sierra Nevada ecosystem of today, smoke likely played a developmental role albeit less obvious. This connection between forest and human health is integral to public acceptance and sound policy. The role of smoke and air quality is a much more compelling research area particularly as global population increases and intact forested wilderness area declines. Smoke impacts to air quality combined with a smoke averse public solicits an atmosphere conducive to suppression in large part because smoke is easily sensed with sight and smell. Comparative analysis of smoke impacts to human health under burn scenarios that include prescribed, ecologically beneficial, and full suppression fire are necessary to best judge population level human health impacts and inform a robust policy discussion.

The widespread systematic smoke monitoring for $PM_{2.5}$ in the Sierra Nevada provides an excellent crucible for examining the confluence of air and fire policy. Additionally, this fire prone area typified historically by frequent low intensity timber fire provides a sensitive indicator system for the impacts of the management policy of fire suppression. The Sierra Nevada provides a particularly effective template for assessing smoke impacts from fuel accumulation from suppression that highlight a systemic problem less apparent in other fuel types (e.g. grasses not timber) and larger fire return intervals (i.e. the northeast forests).

Ecologically beneficial wildland fires such as the Lion Fire (or the Sheep, Tehipite, Avalanche, etc.) will have impacts to air quality. Management of these fires can include smoke management theory that incorporates the long term consequences to air quality and public health. Wilderness and large tracts of undeveloped forested land in the Sierra Nevada can be managed at the landscape level. Using a broader perspective that includes long term air quality and smoke impacts is as essential as including other ecosystem health indicators when determining the use of fire.

Chapter 5.1 Forest fire policy: change conventional thinking of smoke management to prioritize long-term air quality and public health

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DOI: 10.1007/s11869-016-0405-4

Abstract

Wildland fire smoke is inevitable. Size and intensity of wildland fires are increasing in the western United States. Smoke free skies and public exposure to wildland fire smoke has effectively been postponed through suppression. The historic policy of suppression has systematically both instilled a public expectation of a smoke free environment and deferred emissions through increased forest fuel loads that will lead to an eventual large spontaneous release. High intensity fire smoke is impacting a larger area including high density urban areas. Policy change has largely attempted to provide the avenue for increased use of ecologically beneficial fire but allows for continued reliance on suppression as a primary tool for a smoke averse population. While understanding the essential role of suppression in protection of life and property, we dispute the efficacy of attempting to eliminate smoke exposure through suppression in a fire prone area to protect human health at the population level. Sufficient consideration to future negative health outcomes needs to be considered in fire management decisions. It is likely that long term air quality is inextricably linked to ecosystem health in the Sierra Nevada. We contend landscape use of ecological fire is essential to forest and human health. Radical change is needed where beneficial wildland fire smoke is treated as natural background and exempted from much of the regulation applied to anthropogenic sources. Tolerance of the measured release of routine smoke emissions from beneficial fires is needed. Using present air quality standards in the more remote areas will provide an opportunity to increase burning in many forests while protecting public health.

Introduction

Wildland fire has long been understood to perform many beneficial ecosystem functions (Kilgore 1981; Stevens et al. 2014) including helping to maximize carbon sequestration in fire-prone areas (Hurteau et al. 2008). Recurring lower intensity wildland fire additionally limits fire spread, reduces fire progression (Parks et al. 2015), and may provide an avenue to control high-severity emissions of smoke (Steel et al. 2015); localizing the subsequent health impacts, decreasing exposure and population at risk. Past fire management policy has primarily been intended to prevent or contain wildland fire with the consequence of reducing ecological integrity in fire-adapted ecosystems (Dellasala et al. 2004). Climate change will likely contribute to increased fire size and frequency while also increasing the length of the fire season (Westerling et al. 2006). Although suppression at times is the correct response to protect life and property, currently it has the potential to be the default management action. This suppression bias in wildland fire fighting is a product of institutional practice from over 100 years of full suppression. Additionally, fire managers are influenced by political matters such as zero tolerance for an escaped prescribed fire, failure to achieve desired objectives of a managed lightning fire, and pressure from a public that is conditioned to expect smoke free skies and fires that can always be fully suppressed.

Future policy will likely continue to be based almost exclusively on property protection and suppression unless the entrenched disincentives of current policy are overcome and proactive use of ecologically beneficial fire is supported (North et al. 2015a). Full suppression is the path of least resistance, given the current fire management environment, where managing a fire often comes with additional complexities. These include smoke management and the additional temporary monitoring, nuisance complaints, airshed capacity limits, and public health concerns as well as increased local political pressure, lack of public support, and limited resources (both personnel and monetary). There is always a reason not to burn (Chung and Kim 2008; Boer et al. 2015) and current policy reinforces this by creating a path where managed fire can happen but the default remains suppression (Thompson et al. 2015). Policy has adjusted as the natural role and function of fire has become better understood and the policy discussion continues (Topik 2015; North et al. 2015b). This includes smoke management plans at state and local levels that attempt to allow for more burning. At the federal level, legislation such as the Exceptional Events Rule (2007) has provided guidance that has not been consistently applied across all regions of the United States.

Smoke from wildland fire is one of the many reasons there is a reluctance to use fire as a resource tool. Air quality will likely become a more significant factor in the decision process as the public is confronted with additional smoke and more nuisance complaints are generated. Research on smoke from wildland fire has primarily focused on health impacts from exposure during large canopy replacing events. This easily leads the public and health officials to an assumption that all wildland fires have the same impacts to air quality. Currently there is a limited understanding of the tradeoffs between more frequent use of smaller fires versus larger higher intensity fires that are fully suppressed and the product of full suppression.

Without understanding the impacts from wildland fire smoke under historic or natural fire regimes it is easy to understand how suppressing all emissions for public health would appear to be sound policy. Unfortunately this is a short term solution where future emissions and forest health are essentially ignored and priority is given to restricting wildland fire emissions as much as possible with the assumption that future fire will not occur.

The Sierra Nevada of California Example

The Sierra Nevada and adjacent areas of California are particularly compelling example of where wildland fire policy collides with air quality and public health policy. The combination of large tracts of federally protected land including multiple wilderness areas in the fire-adapted ecosystem of the Sierra Nevada bordered by the densely populated Central Valley of California, which has some of the worst air quality in the country (independent of wildland fire emissions), provides a crucible through which land and air managers attempt to navigate. In practice, air regulators are often confined by political boundaries which many times have little practical value for air quality determination when dealing with wildland fire emissions. Pollutants monitored at large urban areas are assumed to represent entire districts. Blanket policy and action options handcuff regulators to make decisions about wildland fire emissions in the wilderness in the same context as industrial, agricultural, or vehicle emissions in an urban area. Air basins which include large tracts of federally protected land within these districts can be considered for regulatory purposes non-attainment even when they are below federal and state thresholds (Cisneros et al. 2014). This leads regulatory decisions to favor suppression and allow burning only under certain meteorological conditions. These conditions of "good dispersal" are often simply when the smoke is blowing out of their district.

Anthropogenic emissions in the Central Valley leave little to no room for air managers to accommodate any increase in pollutant emissions. The perception that any wildland fire creates unwanted air quality impacts that would otherwise not occur also restricts air management decisions. The concept that ecologically beneficial wildland fire could be managed to minimize public health impacts from smoke is difficult for policy makers to embrace. The necessity to limit emissions today to meet air quality attainment and for protection of public health continually pressures land and air management to limit wildland fire smoke whenever possible; reinforcing suppression bias when managing wildland fire. Our ability to limit smoke today through suppression is giving a false sense that wildland fire smoke can be completely eliminated.

The Clean Air Act (CAA), the primary enabling legislation on air quality management, is intended to protect human health from the mounting dangers to the public health and welfare from growth in the amount and complexity of air pollution brought about by urbanization, industrial development, and the increase use of motor vehicles in the expanding metropolitan and other urban areas. Fire policy could meet CAA intentions by managing emissions through the use of ecologically beneficial fire today; reducing the risk of large emission events in the future from continued fuel accumulation and ecosystem degradation.

Impacts from fire suppression have the potential to greatly increase both smoke emissions and the subsequent impacts to public health by radically altering the current ecological system and its ability to self-regulate fire size and intensity. The large canopy replacing suppression fires becoming more frequent in the Sierra Nevada are creating more smoke that is impacting a larger geographic area and subsequently including more densely populated areas. Increases in both the spatial extent and exposure from large fires not typical of the Sierra Nevada will result in an increased potential for negative health impacts that would not occur with increased use of ecologically beneficial fire. Actively managing wildland fire can improve forest health and provide the best long-term air quality.

Wildland fire and the subsequent extent and impacts of smoke have increased since the beginning of widespread monitoring of air quality in California. This is due to uncontrolled wildland fires that have increased in size and intensity; impacting a larger area with an increased population. Megafires like the Rim and King Fires impacted regional air quality including multiple urban centers in addition to local communities. This increase is due to fire suppression, which has both temporarily limited emissions and increased fuel loads – creating a backlog of wildland fire emissions (Hurteau and North 2009) that will likely be increased with expected future climate scenarios (Hurteau et al. 2014). Smoke historically was present throughout the Sierra Nevada for much of the summer and fall as high-frequency ground fire was needed to sustain the Sierra Nevada ecosystem mosaic (Swetnam et al. 2009; Baker 2014). The routine emissions from this higher frequency fire likely made the spatial extent of these smoke events smaller and more localized.

The Central Valley of California frequently exceeds federal and state air quality standards. Air pollution is prevalent in this relatively economically poor area of California. This environmental injustice, primarily driven by anthropogenic emissions of air pollutants, may be subjected to further impacts through the historic fire management policy of fire suppression in the adjacent Sierra Nevada. As federal land managers attempt to restore fire to the landscape, pressure is increased from air managers who have no capacity to increase emissions in an already polluted air shed. This short term protection of human health may be leading to the degradation of environmental health. A more comprehensive decision needs include the mid and long-term impacts of this policy. We may very well be reducing the immediate human health impacts from smoke (Zu et al. 2015) but it comes at the expense of future generations who will face increased exposure. It is very possible without immediate and aggressive re-introduction of natural fire to the landscape, large high intensity fires such as the Rim Fire (2013), King Fire (2014), and Rough Fire (2015) that highly impact air quality over heavily populated areas will be the new normal.

Wider Policy Suggestions

It is likely the best long-term air quality is inextricably linked to ecosystem health in the Sierra Nevada and our current predisposition to suppress wildland fire is leading to adverse impacts to long-term forest health that is not being adequately represented in the decision process. The Sierra Nevada fire-adapted ecosystem may be the most obvious example of the failure of full suppression but the warnings from this system may be the best indicator for other wilderness areas where it is less obvious. This is not to say suppression is not needed. Suppression is necessary to protect life and property and indeed should be well-funded to protect communities, but when a wildland fire originates in and has little to no potential to burn outside of wilderness or into communities, why utilize limited funding and put fire personnel into harm's way suppressing it? Wilderness fires performing ecologically beneficial results in these areas could be allowed to burn by directing fire and air management to first consider the longer ecological benefits. Smoke will inevitably impact air quality no matter how or when it is emitted but could be managed to federal $PM_{2.5}$ compliance in the Sierra Nevada (Schweizer and Cisneros 2014).The Exceptional Events Rule can be used to help air managers during smoke events in wilderness. Further, the backlog of emissions from the management action of fire suppression in wilderness areas seems particularly appropriate when considering an exceptional event impact to air quality.

Although policy does not typically try to control natural events like blizzard and hurricanes, to some degree we can control this natural process, which has led to the accumulation of fuels from past fire suppression. Wildland fire is inevitable and smoke is coming. Further insight into smoke-created public health impacts is necessary to create a more informed decision through the understanding of the nuances encountered when trying to manage fire size, intensity, and proximity to populated areas. After all, are the consequences on air quality worth the trade-off for an ecologically beneficial wildland fire or is a single air pollution event from a stand replacing fire in the best interest of public health?

Radical change is called for when regulating wildland fire smoke emissions for air quality and public health. The more localized smoke impacts from fire of historic size and intensity should be encouraged. The impacts to human health must take into consideration that suppression of these fires is deferring the risk to the future. Not only are ecological benefits often lost but each large, high-intensity wildland fire (Cisneros et al. 2012) will impact a much larger area than the smaller, lower-intensity burns (Schweizer and Cisneros 2014). It is easy to understand how high-intensity fire emissions even from remote locations will then increasingly impact high-density urban areas as larger portions of the forest burn quicker lofting and dispersing the increased emissions more regionally as has been witnessed increasingly throughout the western United States. Local land and air managers need the support of sound policy. Current policy needs to be fundamentally changed so as to incorporate long term sustainability of air quality in and around areas with a fire-adapted ecosystem. Tolerance of routine emissions from wildland fire smoke both from the public and managers is needed. Natural ignition ecologically beneficial wildland fire may best be treated as natural background and exempt from much of the regulation necessary for anthropogenic sources. Regulating to present standards (i.e. 3-year average concentrations for $PM_{2.5}$) in the more remote areas where the ecologically beneficial fires typically burn would provide an opportunity to increase burning in many forests while protecting public health. Understanding smoke impacts and public health advisories to protect exposure during any event is necessary and should be increased to better understand the presence and absence of impacts across the landscape.

Public awareness to potential long-term benefits from ecologically beneficial fire is easily overlooked because of the immediate difficulties of tolerating smoke. An increase in public awareness of the complexity of wildland fire decisions based on air quality is absolutely necessary to provide the public support needed to allow landscape level reintroduction of fire. It is much easier, from a smoke tolerance point of view, for fire and air mangers to suppress fire and remove the potential of immediate public health consequences and nuisance complaints, but policy makers need to question the path of full suppression and ask the question – is fire suppression the most appropriate way to protect air quality or just the easiest way for us today to handle a difficult decision while we mortgage the health of future generations?

Chapter 5.2 Conclusions

This dissertation describes the limitations of mobile EBAM monitors when using comparatively with permanent regulatory monitors such as the BAM. EBAMs are particularly useful in near real time use for daily (24 hour) mean concentrations. Although these monitors have limitations, they are effective at providing a good data source during quick deployment during a wildland fire.

The case study using the Lion Fire illustrates the potential for using landscape level managed fire for ecological benefit while still being cognizant of air quality compliance. This managed fire trigger by an unplanned natural ignition (lightning) contributed to air quality impacts that were largely under federal health standards for $PM_{2.5}$.

$PM_{2.5}$ at Pinehurst from 2006-2015 shows the possibility of managing fire over a large geographic area in an area of non-compliance for $PM_{2.5}$. The full suppression high intensity Rough Fire at Pinehurst exceeded other years and $PM_{2.5}$ impacts to air quality may have largely been managed had ecologically beneficial fire been managed in previous years.

Further research into smoke and impacts to human health from various fire management strategies needs to occur. Future work in understanding tradeoffs to smoke impacts from fire policy implementation should include studies in other ecosystems and pollutants. Monitoring sites should include areas with different anthropogenic background emissions and terrain. Wildland fire smoke needs further study to better quantify dispersal and transport. A better understanding also needs to account for changes in emissions from both fire size and intensity and fuel loading particularly after repeated burns in the same area. It is necessary for remote sensing data and transport models to be better correlated to ground level concentrations. Improvements are needed in emission estimations. Modeling, remote sensing, and ground level monitoring need to continue to improve.

Wildland fire in California is inevitable. Large high intensity fires prevalent when attempting full suppression management lead to greater impacts. A proactive fire program is needed to manage $PM_{2.5}$ from emissions. Although public policy does not typically try to control natural events like blizzard and hurricanes, to some degree we can control this natural process, which has led to an accumulation of fuels from past fire suppression. Smoke is coming. Insight into smoke created public health impacts is necessary to create a more informed decision by understanding some of the nuances encountered when trying to manage fire size, intensity, and proximity to populated areas. After all, are the consequences on air quality worth the trade-off for an ecologically beneficial wildland fire or is a single air pollution event from a stand replacing fire in the best interest of public health?

The large canopy replacing suppression fires becoming more frequent in the Sierra Nevada are impacting a larger geographic area and subsequently more densely populated areas than would have been impacted if full suppression was never instituted. Because of the transport distance of smoke, particularly in large high intensity fires, public awareness to potential long term benefits from ecologically beneficial fire are easily overlooked because of the immediate difficulties of tolerating smoke. Increases in both the spatial extent and exposure from large fires not typical of the Sierra Nevada will result in an increased potential for negative health impacts that would not occur with increased use of ecologically beneficial fire. Wildland fire managed for ecological benefit will provide the best long term air quality and be most beneficial to public health.

The Central Valley of California cannot claim to be an area of clean air. Air pollution is prevalent in this relatively poor area of California. This environmental injustice, primarily driven by anthropogenic emissions of air pollutants, may be subjected to further impacts through the historic fire management policy of fire suppression in the adjacent Sierra Nevada. As federal land managers attempt to restore fire to the landscape, pressure is increased from air managers who have no capacity to increase emissions in an already polluted air shed. This short term protection of human health may be leading to the degradation of environmental health. A more comprehensive decision needs include the mid and long term impacts of this policy. We may very well be protecting human health impacts from smoke at the expense of future generations. It is very possible without immediate and aggressive re-introduction of natural fire to the landscape large high intensity fires such as the Rim Fire (2013), King Fire (2014), and Rough Fire (2015) highly impacting air quality over heavily populated areas will be the normal.

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