

Article

Determining the Impact of Wildland Fires on Ground Level Ambient Ozone Levels in California

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Abstract: Wildland fire smoke is visible and detectable with remote sensing technology. Using this technology to assess ground level pollutants and the impacts to human health and exposure is more difficult. We found the presence of satellite derived smoke plumes for more than a couple of hours in the previous three days has significant impact on the chances of ground level ozone values exceeding the norm. While the magnitude of the impact will depend on characteristics of fires such as size, location, time in transport, or ozone precursors produced by the fire, we demonstrate that information on satellite derived smoke plumes together with site specific regression models provide useful information for supporting causal relationship between smoke from fire and ozone exceedances of the norm. Our results indicated that fire seasons increasing the median ozone level by 15 ppb. However, they seem to have little impact on the metric used for regulatory compliance, in particular at urban sites, except possibly during the 2008 forest fires in California.

Keywords: wildland fire smoke; autoregressive model; exceptional events; estimating ozone norms

1. Introduction

Recent droughts in California, in addition to the history of fire suppression in the United States, have contributed to increased wildland fire [1,2] Additions are highlighted yellow throughout the text and old citation style removed. Ecologically beneficial fire may be used to reduce these stressors [3,4]. Whether from large high intensity, prescribed, or ecologically beneficial fire, smoke from wildland fire will be present regardless of management policy.

Forested systems, especially in California, are at a critical point where large high intensity fires are poised to change forested landscapes [5]. Fire suppression has created a fuels backlog that is ready to be released. Large high intensity fires have had significant impacts on air quality [6], increasing health risks from exposure to smoke [7–9]. Minimizing human interference and utilizing fires that have desired fuel and ecosystem benefits reduces the extreme smoke exposures attributed to unwanted large high intensity fires [10,11].

The Clean Air Act [12] is the federal law that regulates anthropogenic emissions and established National Ambient Air Quality Standards (NAAQS). State regulation in California are set forth in the California Code of Regulations with Title 17 Subchapter 2 [13] dealing with smoke management for agriculture and prescribed burning. Wildland fire smoke, unique from other natural process emission sources, is regulated as if it is an anthropogenic source and not an integral ecosystem component. The Exceptional Events Rule [14] established criteria to determine if an event was controllable and preventable and air quality data could be excluded from regulatory decisions. EER requires air

regulators to determine if impacts to air quality from wildland fire smoke could have been reasonably controlled or prevented and that there exists a clear causal relationship between the smoke event and the monitored exceedance by comparison to historical concentrations. Determining clear causal relations between air quality exceedances and wildland fire smoke events has been difficult to prove particularly when attempting to demonstrate an exceedance from the historical for a given day in an area that frequently exceeds the standard. Validation requires statistical approaches that can help to inform this decision.

Wildfires emit ozone precursors, but there are large variations in the emissions, plume heights, and ozone chemistry that makes it difficult for Eulerian models to predict ozone production [15]. Alternative to Eulerian models are statistical relationships developed between surface ozone concentrations and satellite detected smoke levels above a site [15–24]. Our study utilizes a statistical model developed between hourly ground level ozone values and real time data observed by satellites from the National Oceanic and Atmospheric Administration (NOAA) Hazard Mapping System (HMS). The ultimate goal of our study was to provide plots comparing observed hourly ozone levels at a given site to the expected “norm” for that day (Figure 1) and to determine whether the day was impacted by fire. To this end we, (1) developed a statistical model to estimate the ozone “norm” for each site in our study, (2) estimated the probability of an ozone level significantly exceeding the norm under various levels of satellite detected smoke. Depending on whether the estimated probability of exceedance, on days with potential fire impact, is similar or significantly different than the probability on no fire impact days, one may ascertain whether the satellite smoke is sufficient for detecting fire impact or if further investigation is needed.

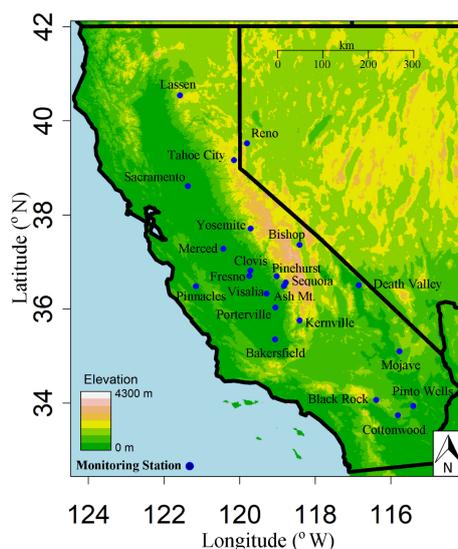


Figure 1. Study area and air monitoring locations.

2. Experiments

2.1. Study Area

Ground based monitors were selected across California focusing on the Central Valley, foothill areas, and east of the Sierra Nevada. Included are 22 ground level ozone monitors with data for the years 2006–2016 (Figure 1). Locations were grouped into Urban, Rural, or Baseline. Monitors were selected to include state run compliance run urban areas (Urban), rural areas where federally protected land begins (Rural), and baseline areas (Baseline) where monitors were typically adjacent to federally designated wilderness.

2.2. Ozone and Weather Data

Hourly values of ozone and meteorological data (wind speed and direction, temperature, barometric pressure, relative humidity, and solar radiation) were compiled where available for each monitoring site. Data for the sites: Bakersfield, Clovis, Fresno, Merced, Porterville, Sacramento, Tahoe City and Visalia are from the California Air Resources Board Air Quality and Meteorological Information System [25]; Bishop is from the Tribal Environmental Exchange Network [26]; Reno is from the State of Nevada Division of Environmental Protection Bureau of Air Quality Planning [27]; and Ash Mountain, Death Valley, Black Rock (Joshua Tree NP), Cottonwood (Joshua Tree NP), Pinto Wells (Joshua Tree NP), Lassen, Sequoia (Lower Kaweah), Mojave, Pinnacles, and Yosemite (Turtleback Dome) are from the U.S. Department of the Interior National Park Service [28]. Monitoring stations at Kernville and Pinehurst are not regulatory sites and are operated by the U.S. Forest Service.

All sites use the Federally Equivalent Method of ultraviolet absorption. This technique has limitations where interference is possible particularly with humidity [29,30]. Well maintained monitors have no significant interferences [31] and reliable in high concentration wildfire plumes with small positive bias [32]. Our model to determine the expected ozone at a given site incorporates humidity to help determine the norm for the individual site. Additionally, positive bias interference from wildland fire smoke would more likely trigger exceedances of the expected norm and thus provide an additional protection for an exceptional event in the case of monitor interference.

2.3. HMS Data

The National Environmental Satellite, Data and Information Service's Office of Satellite and Product Operations with NOAA generates daily HMS fire and smoke analysis [33]. Visible wavelength is used to identify smoke and provide a spatial estimate of smoke as low, medium, or heavy concentrations. HMS data were chosen to aid our present study because of the large geographic area of coverage and because of the utility of this product in assessing ground level smoke impacts to fine particulate matter (PM_{2.5}) [34]. In the Preisler (2015) study [34], we found different levels of satellite smoke to have an impact on ground level PM_{2.5} where the probability of ground level PM exceeding the norm on days with heavy concentrations of HMS smoke was ~37% larger than the corresponding probability on days with low or no HMS smoke above the site. In this study, because we were interested in the impact of smoke from fires on hourly ozone values, daily HMS smoke data were converted to hourly using the provided detection start and end times to determine smoke (low, medium, or high) concentration over a given site. Absent an HMS polygon over a site was considered a time of no smoke.

2.4. Wildland Fire

Wildland fire data are from the National Wildfire Coordinating Group [35]. This includes wildland fires of all sizes on state and federal lands. All fires less than 10 km from each site were compiled to determine if smoke present from a localized wildland fire not large enough to be detected by satellite or obscured by cloud cover impacted a site. Additionally, wildland fires over 4046 ha and less than 100 km from a site was compiled and used to help determine if smoke was missed by HMS data. For a more detailed description of HMS and wildland fire data handling see Preisler et al. (2015) [34] as the same methods are used in this analysis.

2.5. Statistical Models

We used a two-step procedure to analyze the data. In the first step we estimated the expected hourly ozone value using only data from times with no observed HMS smoke above the site in past seven days and no fires within 10 km. The model took into account the seasonal and diurnal pattern of ozone at that site, in addition to the weather conditions (temperature, relative humidity, wind speed and direction) at that hour. This gave us an estimate of ozone "norms" for each site and hour. In the

second step we estimated the probabilities of departure from the ‘norm’ for hours with various levels of smoke (including no smoke) at the hour and three days prior. Specifics of the model are as follows:

Step 1—Estimating ozone norm: We used the hourly ozone values observed at the monitoring sites as the dependent variable in our statistical model. Each monitoring site has its unique, characteristic, diurnal and seasonal patterns [36–39] and distinctive relationships with weather. In order to account for these site specific relationships, a separate model was developed for each site, with the square root of the ozone values as the dependent variable. The distribution of the square root values was better approximated by the Gaussian distribution. The following generalized additive autoregressive model was used to estimate the norms:

$$Y_{ijk} = \beta_o + g_1(\text{day}) + g_2(\text{hr}) + g_3(\text{ws}) + g_4(\text{wd}) + g_5(\text{temp}) + g_6(\text{rh}) + k \tag{1}$$

for $i \in \text{NS}$,

where:

Y_{ijk} = square root of observed ozone value at site i , date j , hour k (hourly ozone data)

β_o = intercept of the regression line

$g_3(\text{ws}), g_5(\text{temp}), g_6(\text{rh})$ = are smooth spline functions [40] of the hourly wind speed, temperature, and relative humidity, respectively. Two sites, Mojave and Pinto Wells, did not have wind data, consequently only temperature and relative humidity were used to estimate the norm for those two sites.

k = autoregressive error of order one used to account for potential serial correlation in the hourly ozone values not accounted for by the diurnal or seasonal trends already in the model.

NS = set of all observations with no fire impact (see below for definition of fire impact).

The above autoregressive model was used to estimate site specific expected hourly ozone levels on days with no fire impact (i.e., the norms).

Step 2—Estimating probability of exceedance: The model in step 1 above accounts for the variability in hourly ozone values due to weather and site specific diurnal and seasonal patterns. At times with no fire impact we expect approximately 5% of the observed values to be greater than 1.64 standard deviations (on the square root scale) from the norm (upper 95th percentile). Ozone values above the expected 95th percentile are considered significant departures, or exceedances, from the norm. Consequently, the impact of fire may be measured by the frequency of times exceedances are observed (or the probability of exceedance) at times with no fire impact compared to the value for times with a potential of fire impact. The probability of an ozone value exceeding the norm is estimated using the following model:

$$\begin{aligned} r &= 1 && \text{if } O_3 > (\hat{\mu} + 1.64\hat{\sigma})^2 \\ r &= 0 && \text{otherwise} \end{aligned} \tag{2}$$

where $\hat{\mu}$ and $\hat{\sigma}$ are estimates of the expected value and standard deviation, respectively, evaluated for each site and time using the model in Equation (1). The probability of departure from the norm, i.e., $p = \text{Pr}[r = 1] = [1 + \exp(-\theta)]^{-1}$ is estimated using the following logistic regression model

$$\text{logit}(p_{ijkl}) = \theta_{ijkl} = \beta_l + \gamma_k \text{smk}_{ijkl} s(\text{shr}_{ijkl}) + \varepsilon_{ijkl} \tag{3}$$

where:

p_{ijkl} —probability of observed ozone level exceeds the norm at hour i , day j , year k and site l

α_o —overall mean of log-odds

β_l —categorical effect of site type (urban, rural, base)

γ_k —categorical indicator for 2008 fires. This variable was included in model to ascertain whether the 2008 fires during June and July were different (as far as their impact on ozone) from fires in other years in the study

shr_{ijkl} —number of hours in previous 72 h with High or Medium level HMS smoke

smk_{ijkl} —categorical variable for hour i , day j , year k with the following seven categories:

- (1) None—no HMS smoke above site at hour i , day j , year k
- (2) Low—Low level of HMS smoke at the hour i , day j , year k
- (3) Mid—Medium level of HMS smoke at the hour i , day j , year k
- (4) (U Hi, R Hi, B Hi)—High level of HMS smoke at hour i , day j , year k at urban, rural or base sites, respectively
- (5) NSFN—Fire within 10 km of the site but no HMS smoke above the site at hour i , day j , year k

ε_{ijkl} —random noise; $s(\cdot)$ —a non-parametric smooth spline function

Estimation for both models (Equations (1) and (2)) were done within the R-statistical package [41].

No fire impacted hours—From here on, we define “no fire impact” as all times with no HMS smoke above at the time (hour) or in the prior three days (72 h) and no known fires within 10 km of the site. The metric, total hours with smoke in past three days, was used in the logistic regression model in step 2 because previous work suggested that ozone production increases for plumes aged two days or more [42] and because the three-day smoke metric gave a better fit (smaller Akaike Information Criteria, AIC) for our models when compared with other metrics (e.g., 48 h or seven-day). The three-day metric was also a way to account for smoke in the area that may have been missed on a specific day or hour due to cloud cover, nighttime occurrence, etc.

Hours with potential fire impact—The complement of “no fire impact” is the set of all hours with some level of HMS smoke observed above the site or in the past three days, or the existence of fires within 10 km of the site. Similar approaches have been used previously for identifying fire impacted days [15,16]. In these previous studies, as indication of smoke impacted days, ozone exceedance from the daily norm was used in addition to presence of HMS smoke and exceedance of ground level PM_{2.5} from the norm. In our present study, we did not use the PM_{2.5} exceedance metric as an explanatory because we did not have PM_{2.5} data at many of the rural locations and we wanted a method that will apply to all locations. While more information (e.g., amount of ozone precursors at the site, or ground level PM values) would help determine fire impact with more certainty, our analysis below demonstrates that one can still ascertain whether a given exceedance may be attributed to fire using only a metric based on HMS smoke.

3. Results

Plots of observed ozone values for the Reno and Sacramento sites, against the expected values (i.e., the norm as estimated using only hours with no fire impact), demonstrated how our model defines days with ozone exceedance (Figure 2). Ozone values above the 95th percentile are considered to be significantly exceeding the norm for that site, day, hour and weather conditions.

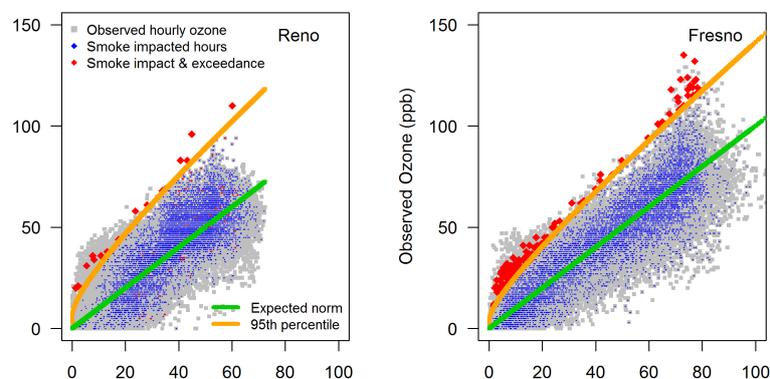


Figure 2. Observed ozone values against the expected norms for a given site day-in-year and weather condition at that hour. Observed ozone values above the estimated 95th percentile line are considered values in exceedance of the norm. Smoke impacted cases are times with “potential fire impact” as defined in text.

From our model (Equation (2)), the estimated odds of exceeding the norm for times with potential fire impact, relative to the odds at times with no fire impact depended on the level of HMS smoke above the site and the number of hours with high or medium smoke in the past three days (Figure 3).

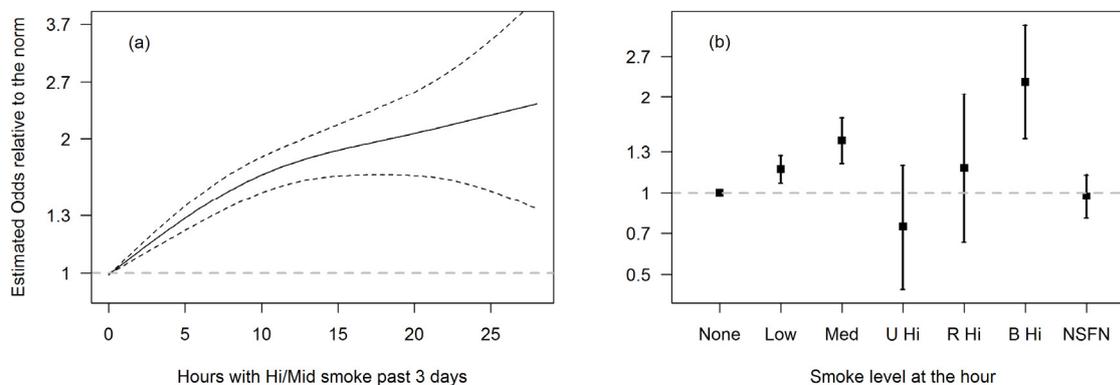


Figure 3. Estimates odds (and approximate 95% CL) of ozone exceeding the norm, relative to the odds of exceedance when there is no fire impact (horizontal line at one). The “no smoke but fire nearby” (NSFN) category in the right panel includes cases with observed fire within 10 km of site but no observed smoke above the site. (a) for number of hours with high or medium level smoke in the past 3 days, (b) for smoke level at that hour (None, Low, Medium Urban High, Rural High, Baseline High, and NSFS).

The first thing to note is that there seem to be no fire impact at times when there is no smoke above, even if there were fires within 10 km of the site. The latter is indicated by the fact that the estimated odds of exceedance on “NSFN” days (no smoke above but fire nearby) is not significantly different from times with no fire impact (Figure 3b). At times with medium level smoke above (Med), the odds of exceedance is 1.5 (95% CL = 1.2–1.7) times the corresponding odds for times with no fire impact. At times with low level of smoke above the odds of exceedance is 1.2 (95% CL = 1.1–1.3) times the odds when there is no fire impact. On the other hand, when there are high levels of HMS smoke above the site, the effect appears to depend on the location of the site. In particular, whether the site was in an urban setting vs rural or base. The largest ozone increase seems to be at Base sites with the odds being 2.3 (95% CL = 1.5–3.4) times the corresponding odds at times with no fire impact, regardless of the level of smoke above. At urban sites, high level of smoke appears to dampen the effect of fire precursors on ozone when compared with the effect at times with medium level smoke. One possible explanation is that solar radiation decreases when smoke above is high and the plume is not mixing to the anthropogenic emission dominated air at ground level. Ozone generated in the plume would therefore not be added to the ground level concentration nor would ozone precursors in the plume be available to react with urban pollutants. In the model estimating the norm (Equation (1)), we use time of day as surrogate for solar radiation. However, on smoke impacted days, high smoke above results in lower than average solar radiation levels. Since ozone production is reliant on radiant energy, lower than normal solar radiation at a given hour may result in lower ozone production with the plume aloft. Smoke from the Sierra Nevada does not routinely mix to the Central Valley [43,44]. The low level mountain parallel winds formed by the Sierra Nevada along the California Central Valley has long been established [45]. Prevailing westerly winds create a decoupled layer along the Central Valley [46]. This Sierra barrier jet has typically been associated with winter precipitation events [47,48] but has been noted in spatial distribution of fine particulate matter in the Sierra Nevada [37]. This can lead to a decoupling of the higher elevation from Central Valley sites leading to Sierra Nevada smoke emissions remaining aloft over the Central Valley [43]. The combination of lower solar radiation during high smoke hours, together with the potential of the plume remaining aloft, may explain the lower impact of smoke from fires on ozone levels in the Central Valley urban sites [11,49]. Wildland fire in the

Sierra Nevada largely transports east and often has minimal impact on the California Central Valley floor [10,43,44,49].

Another interesting result is that the number of hours with high or medium smoke in the past three days had a significant impact on ozone, with the odds of exceedance at 10 h of high or medium smoke being 1.67 (95% CL = 1.5–1.8) times larger than times with no fire impact (Figure 3a). Impact of fires on Base sites was 1.44 (95% CL 1.4–1.5) times larger than the corresponding effect on urban site (as evaluated by the term β in Equation (3) above). Additionally, fires in 2008 had a significantly larger impact on ozone than all other years (i.e., the term γ in Equation (3) was significant). The estimated odds of impact for 2008 fires was 4 (95% CL = 3.7–4.2) times larger than the impact of fires in other years. During the 2008 June and July months, conditions in the western United States produced multiple large wildfires that largely covered the entire state of California in smoke [50,51]. Smoke from wildfires in California during 2008 was well mixed as it primarily came from northern California and settled throughout the Central Valley for multiple days. Other years, the fires were more dispersed throughout the Sierra Nevada of California and had very a different smoke exposure pattern.

In summary, fires in 2008 had the largest impact on ozone levels, in particular when there were multiple hours of high or medium level smoke in previous three days. For all other fire seasons in our study the presence of low, medium or high smoke only marginally increased the probability of ozone exceedance above the norm, and the effect was dampened when the smoke level above was high, especially at urban sites, making the increase in probability significant only at the Base and marginally at the Rural sites (Figure 3b).

Identifying Exceptional Events

The US Environmental Protection Agency (EPA) requires the annual 4th highest daily maximum 8 h ozone value (MDA8) to be less than 70 ppb (averaged over three years) for a site to be in compliance. The EPA also has a policy for ‘exceptional events’ where data may be excluded from the calculations of averages if it can be demonstrated that the air quality would have been different (better) but for the exceptional event of a wildland fire or other source of pollution that cannot be reasonably controlled and caused concentrations at the monitor to exceed the standard. Documentation to receive EPA approval is extensive and for the area of this study occurred for three days in Sacramento in 2008 and three days in Reno in both 2015 and 2016 [52]. Below we present a simple and objective procedure that uses the satellite smoke metric developed in this study to help identify, with high level of certainty, ozone values impacted by fire. The method is particularly useful for identifying whether a given hour with exceedance may be designated as an exceptional event and hence excluded from the calculation of compliance to NAAQS.

Given the results of our analysis above, we can divide the data into two groups:

(1) Ozone values with the potential of being impacted by fire (Imp). For Urban and Rural sites, any ozone value observed when there was medium level of smoke above the site and/or when at least two hours of medium or high level of smoke was observed in the past three days are considered potentially impacted (see Figure 3). For Base sites, any ozone value observed when there was medium or high level of smoke above the site and/or when at least two hours of medium or high level of smoke was observed in the past three days are considered potentially impacted (see Figure 3). Note that simply having high level of smoke above an Urban or Rural site is not sufficient to designate the ozone value as potentially impacted by fire.

(2) Values not impacted by fire (Not Imp). All ozone observations not satisfying the conditions for group 1 (Imp) are assumed to be not impacted by fire.

Ozone values exceeding the norm in the Imp group defined above are candidates for the ‘exceptional event’ designation to be excluded from the calculation of averages used for EPA compliance.

The relationship between distributions of ozone levels for the impacted and not impacted groups gives an estimate of the average size of the impact of fires over all the years and sites in our study (Table 1). Overall the median hourly ozone levels at times with fire impact was ~7 ppb (Imp 2008) or ~1 ppb

(Imp other) significantly higher than the median of times with no fire impact. Additionally, at times with no fire impact 11.4% of the positive departures were greater than 70 ppb. The corresponding percentages for fire impacted cases were 43% during the 2008 fires and 32% during all other years (Table 1). All this indicated that fires have a significant impact on ozone levels for cases designated as “fire impacted” by our rules, with the highest positive impacts seen during the 2008 fires.

Table 1. Median ozone departures from normal and percentage of positive departures above 70 ppb for various levels of fire impact (Imp) and year. Imp (other) category includes fire impacted cases for all years but 2008.

Fire Impact	Departure from the Norm (ppb)	Departure above the Norm (ppb)	Departure below the Norm (ppb)	Positive Departure above 70 ppb (%)
None	0.34	6.2	−5.7	11
Imp 2008	7.8	14	−6.8	43
Imp (other)	1.5	7.4	−6.1	32

Finally, we calculated two values for the EPA metric for compliance, i.e., the annual 4th highest value of the mean daily maximum 8hr value (MDA8) averaged over three years, for each of the sites in our study (Figure 4). The first calculation was done using all observed values and the second calculation was done after removing all values that were considered impacted by fire using our procedure as described above.

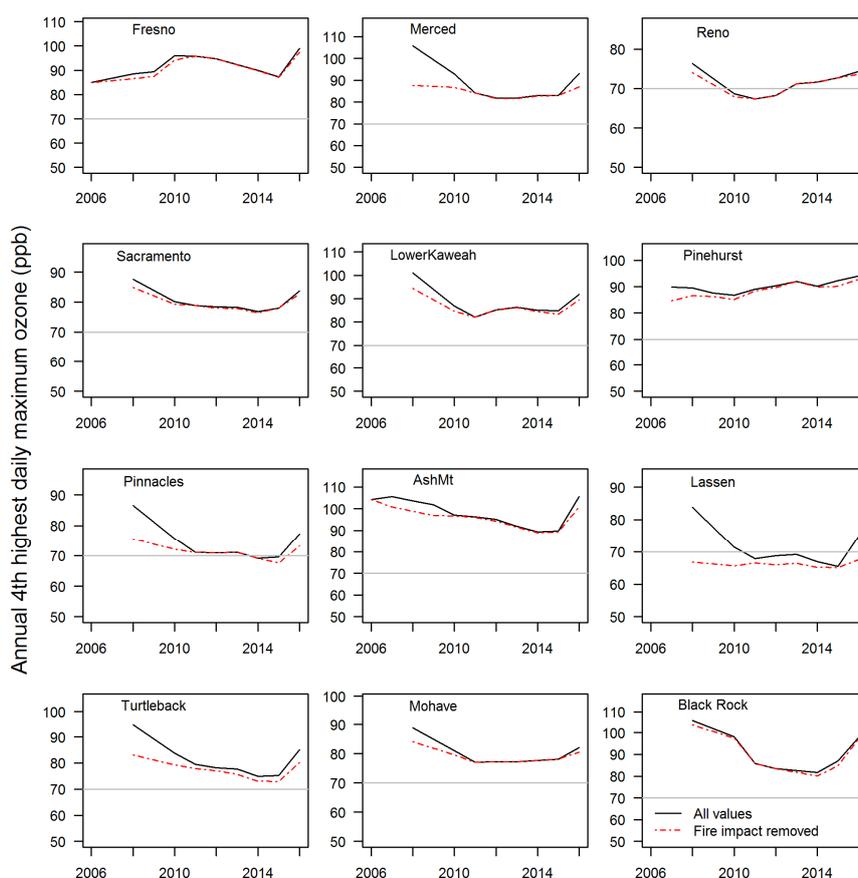


Figure 4. Annual 4th highest mean daily maximum 8hr value (MDA8) (averaged over three years) for 12 of the sites in our study, calculated using all values (black) and only values not impacted by fire (red).

It is of interest to note that, while wildfires seem to have a significant impact on the hourly ozone values, with some fire seasons increasing the median ozone level by 15 ppb, they seem to have little

impact on the metric used by EPA to consider compliance, in particular at urban sites, except possibly during the 2008 fires.

4. Discussion and Conclusions

EER analysis currently looks extensively at a single event to determine ozone values to be excluded from regulatory compliance. Here we suggest using remotely sensed smoke extent data to determine whether a given site has been impacted through a simple process. If the monitor has a higher than expected level of ozone and has been under HMS medium or high smoke density polygons for a few hours in the past 72 h it may be attributed to wildland fire smoke with ~95% confidence. An increase in the number of hours of remotely sensed smoke in the last 72 h increases the probability of exceeding the norm. With 1 h of smoke, the odds relative to the normal are near 1 (no significant increase), by 3 h it is nearly 1.3, and around 10 h of smoke the odds begins to stabilize at 2 (Figure 3a).

Exceedances at times with a few hours of smoke at a site may be attributed to fire with confidence (>95%) regardless of the fuel, fire size, emissions, plume chemistry, timing, or proximity to the monitor. It does not matter how much smoke, where the fire is located, how long it takes to transport, or any other descriptor. This method shows, in these cases, smoke has a significant impact on the monitor which, under the EER, is enough to justify removing the data if concentrations at the monitor exceed the standard. Therefore, no further analysis is needed for EER decisions. However we also found, removing these data have little or no impact on the regulatory compliance in most cases (Figure 4). This method may decrease the ambiguity and time consuming work using individual fires to determine ozone exceedance and rationalize a simpler approach. However, work still needs to be done to estimate the norm at a given site by using historic data from multiple years.

Wildland fire smoke both increases and decreases ozone in California. The most significant increase was for 2008 with a departure from the hourly normal of 7.8 ppb. Other year impacts were also significant but much smaller at 1.5 ppb. Forest fires that occurred in the Sierra Nevada decreased ozone concentrations at urban sites including the Central Valley (Figure 3). This is similar to what has been found for PM_{2.5} where smoke generated in the Sierra Nevada had little or no impacts on the Central Valley [43,49]. Current smoke management policy pressures regulators to minimize most burning, effectively shifting a larger burden of emissions from today to the future [53]. Wildland fire smoke that impacted the Central Valley sites (similar to 2008) tended to be generated from large wildfires west of the Central Valley or in northern California [44]. This smoke impacting ozone was produced outside of a given air district's jurisdiction.

Emissions within jurisdictional boundaries are the only regulatory control for local air districts to limit during high pollution events. Ecologically beneficial fires can be restricted to slow emissions or sped up (fire strategically added to increase burned area) to increase emissions during periods of desired transport because of the perceived impacts to air pollution that may not be present [11]. There is no empirical evidence that limiting emissions in the Sierra Nevada during these times leads to improved regulatory compliance of ozone at sites in the California Central Valley.

Our findings of ozone impacts from wildland fire smoke suggest regulating fire does not provide significant improvement to air quality in the Central Valley. While in some cases the EER might reduce regulatory compliance ozone measurements for rural and baseline sites, the effects on urban ozone compliance are almost non-existent. Further investigation is needed to understand the health implications.

Smoke is inevitable in California. Timing and intensity of emissions are the unknown. Reducing the extent and dispersal of the smoke plume seems to be paramount to reducing exposure to ozone. While in the wildland urban interface dispersal is obviously important because of the proximity to the fire, the most effective way to reduce smoke impacts at the landscape level appears to be having the patience to allow ecologically beneficial fires to burn [10]. This effort will come with a level of vocal public opposition that should be heard but should not drive fire management decisions [54].

A simple method of smoke impact determination will ideally improve compliance documentation. While wildland fire smoke is a part of life in California, extreme smoke events do not need to be the normal. Smoke management is needed particularly during an event when warnings and advice to avoid smoke is the best means to reduce exposure.

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