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Estimating contribution of wildland fires to ambient ozone levels in National Parks in the Sierra Nevada, California

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We have demonstrated that it is possible to produce accurate forecasts of next-day hourly ozone levels in the Sierra Nevada, CA, during fire season.

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ABSTRACT

Data from four continuous ozone and weather monitoring sites operated by the National Park Service in Sierra Nevada, California, are used to develop an ozone forecasting model and to estimate the contribution of wildland fires on ambient ozone levels. The analyses of weather and ozone data pointed to the transport of ozone precursors from the Central Valley as an important source of pollution in these National Parks. Comparisons of forecasted and observed values demonstrated that accurate forecasts of next-day hourly ozone levels may be achieved by using a time series model with historic averages, expected local weather and modeled PM values as explanatory variables. Results on fire smoke influence indicated occurrence of significant increases in average ozone levels with increasing fire activity. The overall effect on diurnal ozone values, however, was small when compared with the amount of variability attributed to sources other than fire.

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1. Introduction

Concerns about smoke from large high-intensity wildfires and low-intensity managed fires in the Sierra Nevada, California have been increasing during the past decade. Smoke from large high-intensity fires is known to contain and generate secondary fine particles ($PM_{2.5}$) and ozone precursors. Ozone (O_3) is a secondary pollutant not emitted directly by fire into the air, but formed by complex photochemical reactions between nitrogen oxides (NOx), carbon monoxide (CO) and volatile organic compounds (VOCs) (Finlayson-Pitts and Pitts, 1993). The availability of NOx downwind of fires may affect O_3 concentrations. The intensity and spatial distribution of O_3 formation has been shown to be mostly controlled by the presence and properties of the available hydrocarbons (Derwent and Jenkin, 1991). There are numerous examples of elevated O_3 concentrations downwind of wildland fires (Goldammer et al., 2009). Analysis of more than 20 years of fire and air quality data demonstrated correlations between ozone levels and total area burned and biomass consumed by fires in the Western United State (Jaffe et al., 2008).

High ozone concentrations have been shown to be harmful to plants even in remote regions such as the National Forests and Parks in the USA (Grulke et al., 2003). In addition to fires, fossil fuel consumption in urban areas as well as use of nitrogenous fertilizers in the San Joaquin Valley upwind both contribute to making the National Parks in Southern Sierra Nevada region some of the most

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ozone polluted Parks in the United States (Unger, 1978; Grulke et al., 2003). Attributing increases or decreases in ozone to a specific source, such as fire, is a major challenge because it is hard to quantify all sources of ozone variation. Various process-based models have been developed for forecasting PM and ozone levels in the presence of fires. One such model is the BlueSky Smoke Dispersion Modeling Framework developed by the Forest Service Air Fire Team (Larkin et al., in press) and is run operationally in many regions in the United States including California and Nevada. Predictions from process-based models often include only one source of pollutants (e.g. fires). Consequently they cannot be used to forecast total ozone or PM concentrations but rather only the contribution from the one source. To forecast ozone concentrations one needs to develop a statistical model that attempts to include all measured sources of variability and to estimate the distribution of the error due to all the unmeasured sources. The latter is essential for model evaluation; for attributing significant effects of fires or other sources of ozone and for forecasting with known precisions.

In this article we describe a statistical model that is used to forecast next-day ozone levels at given sites. The statistical model takes into account some of the known sources of ozone fluctuations, including changes in temperature, humidity, wind speed, wind direction and, during fire season, effects of smoke from fires. Seasonal patterns of ozone fluctuation characteristic for a given site are incorporated in the model by estimating background seasonal levels from historic data. Other sources of variation not directly accounted for in the model – e.g. variability in daily amount of ozone produced by sources other than fire – are included in the uncertainty measure as random effect variables. The advantage of a model that is capable of estimating mean effects and uncertainties simultaneously is that evaluation of model performance is immediate and predictions are available with specific precision levels.

The statistical model is developed in three stages. In Section 3.1 we estimate background seasonal trends and daily diurnal patterns for each site using 16–26 years of historic data depending on the site. Next (Section 3.2) we study the effects of daily fluctuations in weather and fire activity on the residual (de-seasoned) ozone series after removing the long-term background seasonal trends and diurnal patterns, as estimated in Section 3.1. Finally, in Section 3.3, we develop a statistical autoregressive model for forecasting next-day ozone levels at sites with continuous ozone and weather monitors. The goodness of fit and the skill of the statistical model in forecasting ozone are discussed in Section 4. It is anticipated that the forecasts from the statistical model can be used to support decision making by land and air resource managers regarding air quality and prescribed burns in the Class-I areas of the Sierra Nevada and other sensitive areas.

2. Study sites and data

The study employed long-term observational data from four sites operated by the National Park Service in the Sierra Nevada of California. Two of the sites, Ash Mountain (ASH) and Lower Kaweah (LWK), are located in Sequoia and Kings Canyon National Parks, while the other two, Turtleback (TRTB) and Yosemite Valley (Yos), are in Yosemite National Park (Fig. 1). Ozone values from these sites are used by air resource managers to support decisions regarding air quality and prescribed burns in these Class-I parks. A Class-I designation originates from the 1977 amendment to the Clean Air Act, identifying park and wilderness areas that are subject to the most stringent regulations of any designation to protect air quality related values. Section 118 of the 1970 Clean Air Act requires the parks to meet all federal, state, and local air pollution standards.

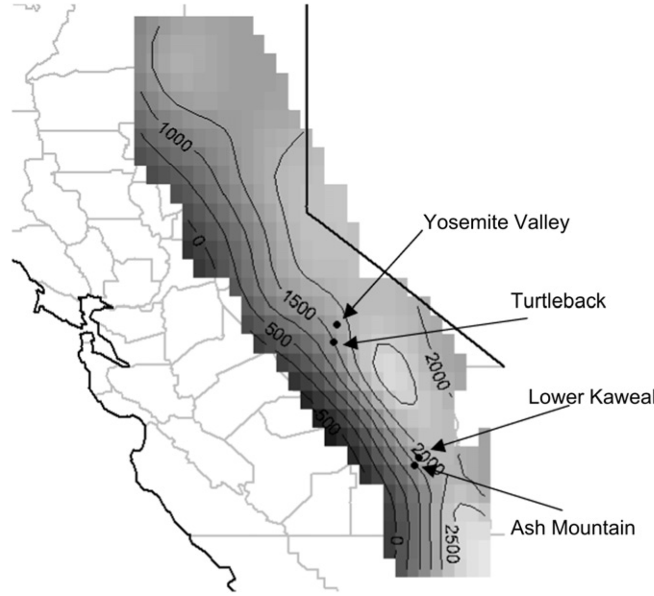

Fig. 1. Locations of the four monitoring sites in Sequoia and Yosemite National Parks with background contour map of elevation in the Sierra Nevada region of California.

At each of these four sites, ozone concentrations were measured with a Thermo Environmental Model 49C UV absorption instrument operated by the National Park Service. The ozone monitors are housed in a temperature controlled shelter, calibrated at the beginning of each season and checked against a calibrator on a weekly basis. Each site also had a surface weather station that measured standard meteorological variables. Rotronics humidity sensors, Climatronics wind speed/wind direction, temperature, and solar radiation sensors were mounted on a 10-m tower. Data are quality controlled and archived on at the NPS Air Resources Division office in Denver, CO.

The statistical analyses performed employed primarily the hourly averaged values of ozone concentrations and meteorological variables at these sites in the past three years (2006–2008) except for Yosemite Valley site that did not have data for 2008. To describe background ozone patterns at these locations, historical ozone data were also obtained which varied in record lengths from 16 years (1992–2007) for the two Yosemite sites to 24 years (1984–2007) and 26 year (1982–2007) for Lower Kaweah and Ash Mountain sites, respectively.

During the three-year (2006–2008) period, fires of various sizes and distances from the four sites were recorded (Fig. 2). There were a total of 809 reported fires with 59 fires that burned at least 40 ha (100 acres). The largest fire in the region was in 2008 and it burned a total of 6192 ha (15,300 acres). In our statistical model the impact of smoke on ozone concentrations from these fires was taken into account by using, as surrogate, $PM_{2.5}$ concentration obtained from the BlueSky model output at the nearest grid point to the four sites.

BlueSky is a smoke dispersion modeling framework that combines burn information with models of consumption, emissions, meteorology, and dispersion to yield a prediction of trajectories and surface concentrations of particulate matter (both $PM_{2.5}$ and PM_{10}) from managed low-intensity fires, wildfires, and agricultural burn activities (O'Neill et al., 2008; Larkin et al., in press). Currently BlueSky predictions of smoke from wildfires are available daily for many locations in the United States. In California and Nevada, BlueSky has been implemented by the California and Nevada Smoke and Air Committee (CANSAC), an

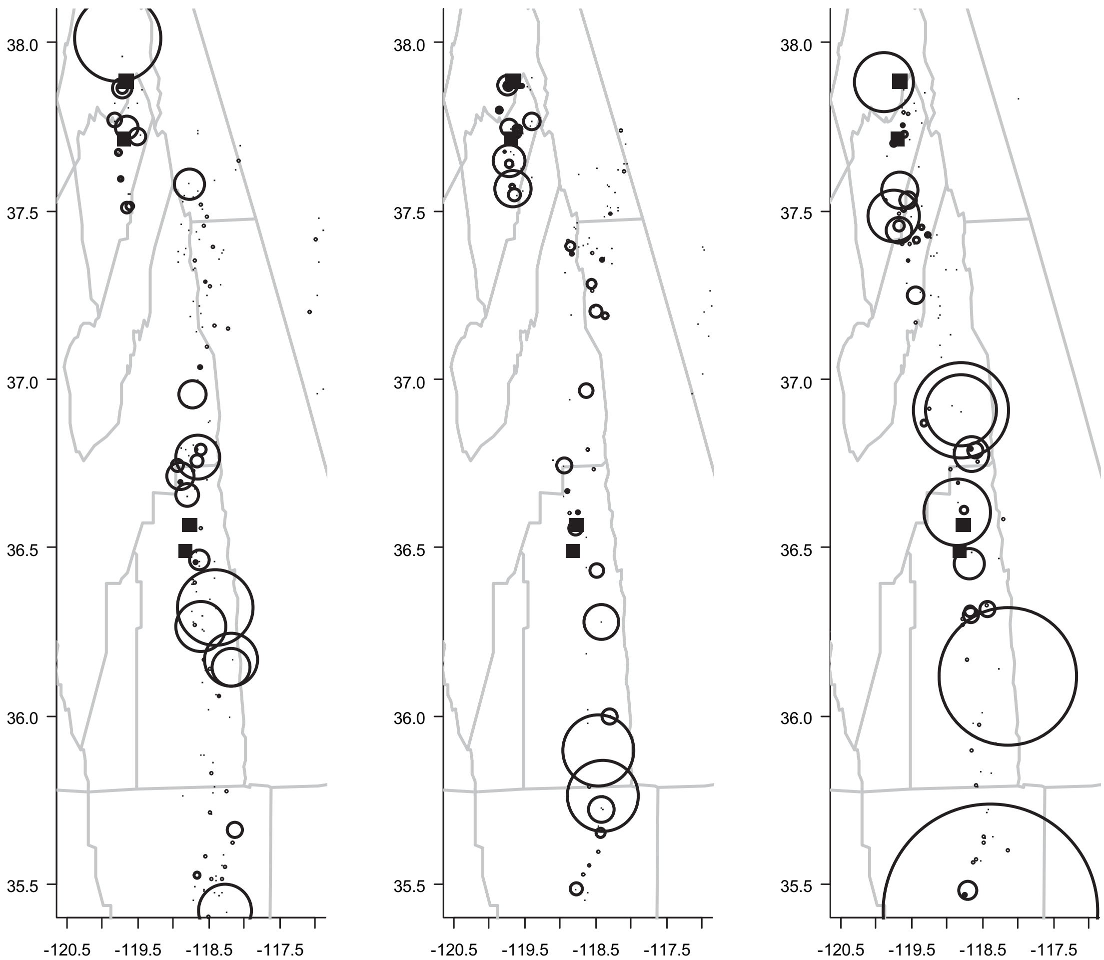

Fig. 2. Locations of four monitoring sites (■) and wildland fires (circles) in the region. Diameters of circles are proportional to the total acres burned (square-root scale) in a fire that started at the center of the circle. The circles are not the boundaries of the fires. Sizes of fires ranged from 0.04 ha (dots) to 15,000 ha (largest fire in 2008 map).

interagency consortium of meteorologists, air quality and fire managers and researchers (http://cefa.dri.edu/COFF/cofframe.php) (Brown and Kahyaoglu-Koracin, 2007) into its operational weather forecast system. The meteorological fields needed for smoke transport and dispersion are provided by real-time regional weather forecasting using the MM5 meteorological model (Grell et al., 1994) at 4 km grid resolution over a domain that covers California and Nevada. The transport and dispersion of $PM_{2.5}$ from fire emissions in the region are estimated using the CALPUFF (Scire et al., 2000) and HYSPLIT (Draxler and Hess, 1997) models. We used the predicted $PM_{2.5}$ values from BlueSky forecasts to characterize the amount of PM produced by smoke from both large high-intensity and managed low-intensity fires in the region. BlueSky forecasts are zero for days with no reported fires. Outputs from other transport models, with the capability to produce spatially and temporally explicit values in real time, may also be used.

There are several advantages of using modeled PM output from the state-of-art smoke dispersion model as surrogate for fire activities over previous methods. Previous studies (Preisler et al., 2005) make simple, subjective assumptions about fire impact that are based primarily on distance from the fires, an average wind direction and total acres burned by the fire. The metric used in Preisler et al. (2005) was not a forecast and can only be used for quantifying historic fire effects. The BlueSky model incorporates emissions from all fires and the transport of smoke from distant fires. The model results, however, may contain relatively large errors due to uncertainties in emissions, meteorological input, and assumptions on dispersion mechanisms and smoke deposition etc. Despite these limitations, the method of using BlueSky-predicted PM concentrations resulting from wildland and prescribed fires, which are available in real-time for California and many other regions in US, provides a better alternative to the distance- and average wind direction-based empirical approach or to the use of total area burned in the region as a measure of fire activity. In this study we are not trying to validate the output of BlueSky but rather use it as a reasonable indicator of the amount of fire activity in the region.

3. Statistical models

3.1. Estimating background ozone patterns

The long-term historic ozone data were used to estimate background seasonal and diurnal patterns at the four locations. The 24-h and 12-month ozone cycles were captured by using a semi-parametric additive regression model with periodic spline functions (Hastie et al., 2001). The statement for the statistical model is given by

$$y = \alpha + s(\text{day\_in\_year}) + s(\text{h}) + \varepsilon \quad (1)$$

where, y – is the hourly ozone level; α – is the intercept parameter; $s()$ – non-parametric periodic spline function; ε – is a random error term.

A separate equation was estimated for each of the four sites. Non-parametric splines are useful in particular when relationships are not expected to be linear. Periodic spline functions provide a flexible method for fitting relationships with a cyclical pattern such as the 24-h diurnal pattern or the 365-day seasonal patterns often seen in ozone values. We also estimated the between year standard deviation that was used later to incorporate between year variability in our overall error bounds. The between year variability was estimated by including a fixed year effect in model (1) above and then calculating the variance of the estimated year effects. All through our analysis we used the publicly available R statistical package (R Development, 2008).

It is anticipated that the estimated background seasonal and diurnal patterns will account for the majority of the natural ozone variability at these locations. The background pattern at each site also reflects the climatological conditions and the averaged ozone exposures. The estimated historic patterns at each site are used in the following sections to de-season the ozone data, before the second stage of fitting wherein contributions of fires and daily weather are estimated.

3.2. Quantifying sources of variation

Hourly meteorological data from the surface weather stations at each of the four sites were used to evaluate the simultaneous influence of temperature, humidity, wind speed, wind direction, and solar radiation to the diurnal variation in ozone values. The $PM_{2.5}$ concentration values from the nearest grid point from BlueSky model predictions is used as surrogate for the influence of smoke on the site as a result of wildland and prescribed fire activities in the region. The BlueSky-predicted $PM_{2.5}$ value would be zero if there are no fires in the region or if there are fires and the site is not in the path of the smoke plume. Although BlueSky output used here is not a predicted ozone value but rather forecasted $PM_{2.5}$ (here on referred to as *mpm* or modeled PM), it is assumed that if the amount of smoke and PM from surrounding fires increases, the amount of ozone precursors coming from the fires may also increase. Our task here is to quantify the significance and shape of the relationship between *mpm*, including lagged values of *mpm*, and ozone.

We used non-parametric spline functions in a linear regression model to estimate the simultaneous effects of the following explanatory variables: *temperature*, *solar radiation*, *relative humidity*, *wind speed*, *wind direction and mpm*. Because wind direction is periodic (with a 360° cycle) we estimated a periodic spline for this variable. After some exploratory analysis with various combinations of *mpm*, the variable that appeared to show the largest significance, and the one used in the final model, was the median of the *mpm* values of the previous 48 h (hereafter referred to as *mblue*).

The specific autoregressive model used to fit the data was

$$z_{kt} = \alpha + \sum_m s(X_{kmt}) + \varepsilon_t \tag{2}$$

where, z_{kt} – the residual series for site k and time t after removing the background seasonal and diurnal effects estimated from the historic data; $t \in \{1, \cdots, T\}$ h with $T = 3$ years × 365 days × 24 h; X_{kmt} – value of mth explanatory variable at time t and site k; $s()$ – non-parametric smooth spline or periodic spline function; $\varepsilon_t = \rho_1\varepsilon_{t-1} + \rho_2\varepsilon_{t-2} + \rho_3\varepsilon_{t-3} + \xi$ – is an autoregressive process of order 3 with ξ independent Gaussian error terms.

The autoregressive model of order 3 was found necessary to account for the serial correlation in the data (see goodness of fit analysis in Section 4). The program *gamm* of the R statistical package was used to fit the generalized additive mixed model in equation (2).

3.3. Forecasting next-day ozone levels

This section is concerned with fitting a predictive model for next-day ozone levels at sites where hourly ozone and weather values are available. We used an autoregressive model with historic time series information on hourly ozone values up to the previous day in addition to modeled PM and local weather. The specific time series model used to forecast next-day ozone levels for a particular site was as follows

$$z_t = \beta_1 z_{t-24} + \beta_2 mblue^{1/3} + \beta_3 \Delta temp + \beta_5 \Delta windd + \varepsilon_t \tag{3}$$

where, z_t – the residual series for time t after removing the background seasonal and diurnal effects estimated from historic data; $t \in \{1, \cdots, T\}$ h with $T = 3$ years × 365 days × 24 h; $z_{t-24} = (y_{t-24} - hy_t)$ – difference between historic average (hy_t) and previous-day ozone level (y_{t-24}); *mblue* – median modeled $PM_{2.5}$ value for the previous 48 h; The variable *temp* indicates temperature, *winds* and *windd* denote wind speed and direction.

$$\Delta temp = temp_t - temp_{t-24}$$

$$\Delta windd = \begin{cases} WSW & \text{if } windd_t \in \{200-275\} \text{ and } windd_{t-24} \notin \{200-275\} \\ 0 & \text{otherwise} \end{cases}$$

$$\varepsilon_t = \rho_1\varepsilon_{t-1} + \rho_2\varepsilon_{t-2} + \rho_3\varepsilon_{t-3} + \xi - \text{is an autoregressive process of order 3 with } \xi \text{ independent Gaussian error terms.}$$

$\beta_1, \beta_2, \beta_3, \beta_4, \beta_5, \rho_1, \rho_2, \rho_3$ — parameters estimated from the data.

The 24-h lagged difference in the others weather variables; relative humidity; solar radiation and wind speed were not found to be significant when added to the model in (3).

4. Results and discussion

The background ozone levels at the four sites, as estimated from the historic data, exhibit strong seasonal cycles (Fig. 3). At all four sites, ozone concentrations appear to be lowest on average in winter and highest in summer, with peak values occurring in late summer (August). The two Yosemite sites appear to be less polluted on average (using the ozone metric) than the Sequoia–Kings Canyon sites. The Ash Mountain site in Sequoia National Park appears to have the highest average ozone concentration values that reach the critical level of 90 ppb in the months of July and August. Some of the between site differences may be due to differences in topography, elevation and distance from urban sources.

There were significant between-station differences in the estimated diurnal patterns (Fig. 4). The Yosemite Valley site shows the largest diurnal amplitude, while Turtleback is nearly flat with only a small diurnal signal. Similar difference in diurnal ozone variation among different sites in the Sierra Nevada was also found

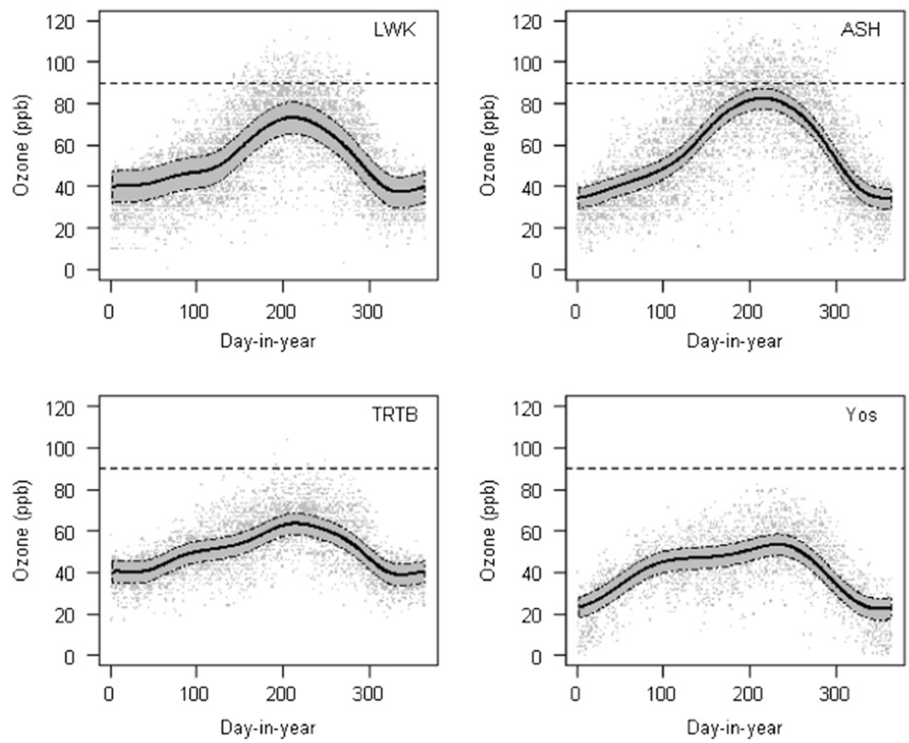

Fig. 3. Estimated historic average ozone patterns (black curve) for 1400 h superimposed on observed values for that hours (gray dots) at four stations. Gray band is 95% CL for between year variability. Dashed horizontal line is the critical ozone level of 90 ppb.

by Van Ooy and Carroll (1995) in a climatological analysis of ozone data and they attributed the difference to the topographic sitting rather than to the remoteness to urban areas that typically explains large and small diurnal variation in ozone concentration. At Ash Mountain and Lower Kaweah, ozone peaks in late afternoon, while at the Yosemite Valley site, the peak appears to occur around noon.

The overall variability in ozone values, as given by the standard deviation over all sites; hours-of-day and days-of-year, was ±18.3 ppb with a mean of 54 ppb and coefficient of variation of 33%. Table 1 shows the percentage reduction in the variance as various explanatory variables are included in the regression model.

As would be expected, a relatively large portion of the variability in ozone values was attributed to between site variations (19.7%). Background seasonal and diurnal patterns (as described in Section 3.1 above) appear to account for 54% of the variation and local meteorological variables listed above account for an additional 6%. This indicates that most of the ozone variability may be attributed to 'average weather' or climate conditions with an additional 6% variability due to departure of a given day's weather from the seasonal averages at that site. Fire activity, as measured by previous 48 h median BlueSky values (*mblue*), accounts for an additional 1.0% of the variation in the de-seasoned data and 1.4% of the variation when local weather (but not seasonal trends) and topography were included in the model.

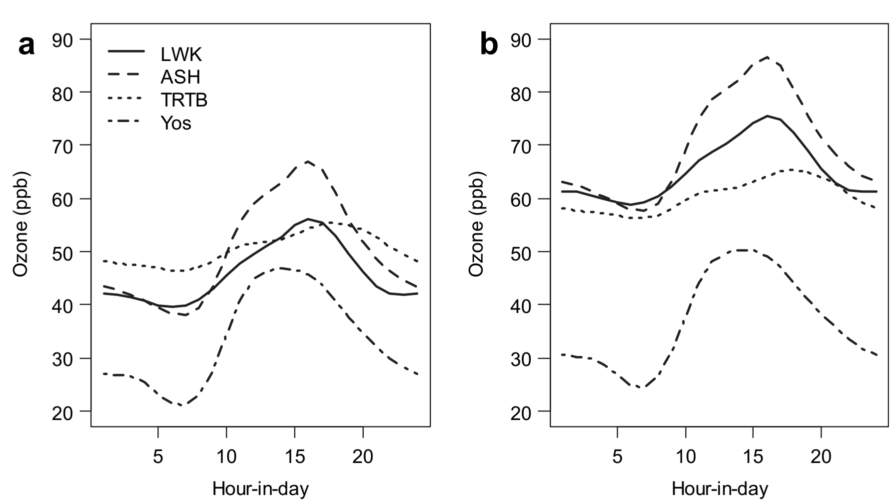

Fig. 4. Estimated historic diurnal ozone patterns for the four stations evaluated for two dates (a) 15 May and (b) 15 June. Note that, although the diurnal shapes for each site do not change between the dates, the overall increase between 15 May and 15 June is larger for the Sequoia sites as compared to the Yosemite sites.

Table 1
Estimated reduction in variability of ambient ozone as various sources of variation are added to a semi-parametric statistical regression model.

Model	Variables	Reduction in variance
I	Between sites	19.7%
II	Site specific seasonal + diurnal trends (historic average)	54.4%
III	I + local met variables	38.1%
IV	II + local met variables	5.6%
V	III + *mblue*	1.4%
VI	II + *mblue*	1.0%
VII	IV + *mblue*	0.6%

Estimated partial effects for each of the explanatory variables were developed by fitting equation (2) to three years of de-seasoned ozone data from the four sites. The estimated effects, standardized to have mean zero, are presented in Fig. 5. A variable does not have a significant effect (at the point-wise 5% level) if the horizontal line at zero is almost completely within the confidence bounds. The meteorological variable that appears to have the largest effect on the de-seasoned ozone values was temperature, which is not surprising given that ozone is produced by photochemical reactions that depend on temperature. Partial effects of solar radiation and relative humidity appear to be small but significant. Wind speeds of greater than 10 m s^{-1} appear to decrease average ozone levels, however, the standard errors are large because of the small sample sizes at high winds making the variable only marginally significant (*P*-value = 0.07). Wind directions from south and southwest (180–270°) appear to increase ozone levels above the seasonal averages. Given the locations of these four sites on the western slopes of the Sierra Nevada, the increase in ozone levels with winds from south and southwest suggests the contributions to ozone concentrations at these sites from transport of ozone precursors from pollution sources in the Central Valley, especially the heavily polluted cities like Fresno and Bakersfield in the southern part of the valley.

Finally, there seemed to be a small but significant effect of fires as measured by the increase in ozone levels with increasing values of *mblue* (the median amount of modeled PM levels in the last 48 h). On average there appeared to be about 2.1 (SE = 0.7) ppb increase in ozone for each one point increase in the cube root value of *mblue* (*mblue*$^{1/3}$). The standard errors around the estimated curve are large, in particular for larger values of *mblue*. Part of the reason for the large variably is possibly due to the fact that BlueSky predictions of amounts and directions of smoke dispersion are not very accurate. The other reason is that there were very few observed cases with *mblue* values greater than 1. During the three years and the four sites included in the study, the maximum observed *mblue* value was 7.3, leading to an estimated increase in ozone of 4 ppb (1.2–6.8 ppb) attributed to fire. The estimated curve

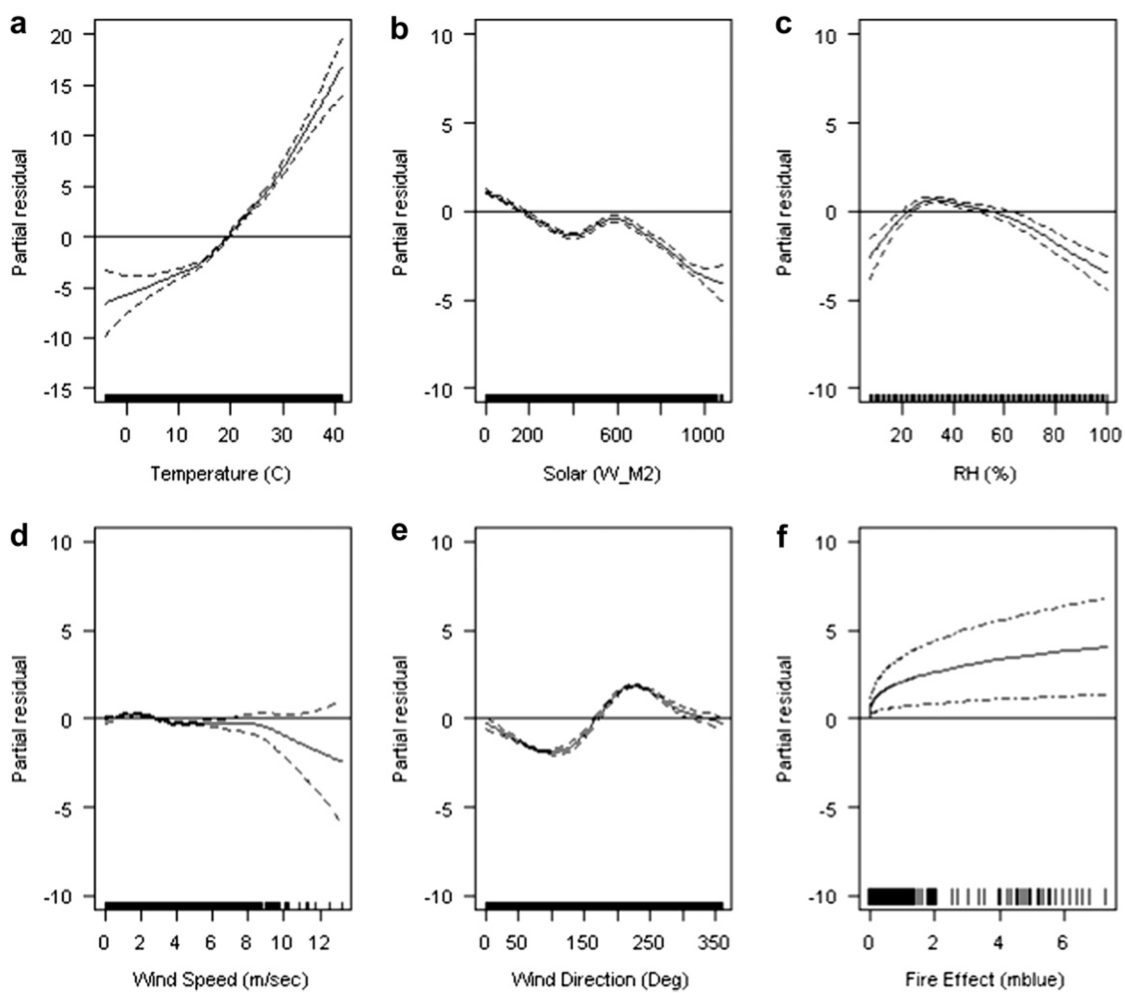

Fig. 5. Estimated effects (partial residuals) of weather and fire variables on the de-seasoned ozone levels – where historic background levels are remove. Dashed lines are approximate point-wise 95% confidence limits. Hatch marks at the bottom indicate observed levels of the explanatory variable.

seems to be leveling off with increasing values of *mblue*. However, it is not clear if this trend will continue for values of *mblue* outside the range observed during the period of this study.

The estimated regression line for forecasting next-day ozone values was

$$\underset{(\pm \text{s.e.})}{\hat{y}_t} = hy_t + \underset{(\pm 0.005)}{0.4 \times z_{t-24}} + \underset{(\pm 0.72)}{2.7 \times mblue^{1/3}} + \underset{(\pm 0.03)}{0.28 \times \Delta temp} - \underset{(\pm 0.02)}{0.6 \times (\Delta windd = O)} + \underset{(\pm 0.21)}{0.4 \times (\Delta windd = WSW)} \quad (4)$$

The average standard error around the forecasted model as given in (4) was ± 10.2 ppb.

The particular autoregressive model in equations (3) and (4) was used because the diagnostic tests of the residuals indicated the need for a third order autoregressive process (Fig. 6). The autocorrelation function of the de-seasoned series shows the amount of serial correlation between lagged hourly ozone values (Fig. 6). An autoregressive model of order one (AR1) did not seem to remove all the serial correlations with some significant correlations persisting at 2 and 3 h lags in addition to those at 24 h lags (Fig. 6b). There were no apparently significant serial correlations in the residual series from the final model in (4) (Fig. 6c).

We developed graphs of observed and forecasted ozone values for various periods of time in order to further asses the skill of the statistical model and to further study the contribution of fires to ozone variability. Although the parameter estimates in equation (4) above were developed using data from all three years, in what follows forecasts for a given year were developed by first excluding data from that year. This is similar to cross-validation where estimates for a time period *delta* are produced by excluding the data from period *delta* before performing the model fit. Observed and forecasted ozone values for various periods are shown in Figs. 7 and 8 as a demonstration of our results. During the period 15–25 July 2006 there were many lightening-caused fires in Yosemite National Park. On July 20 and 22 (day-in-year 201, 203) there were three large fires (>40 ha) with one fire on July 22 that burned 2441 ha.

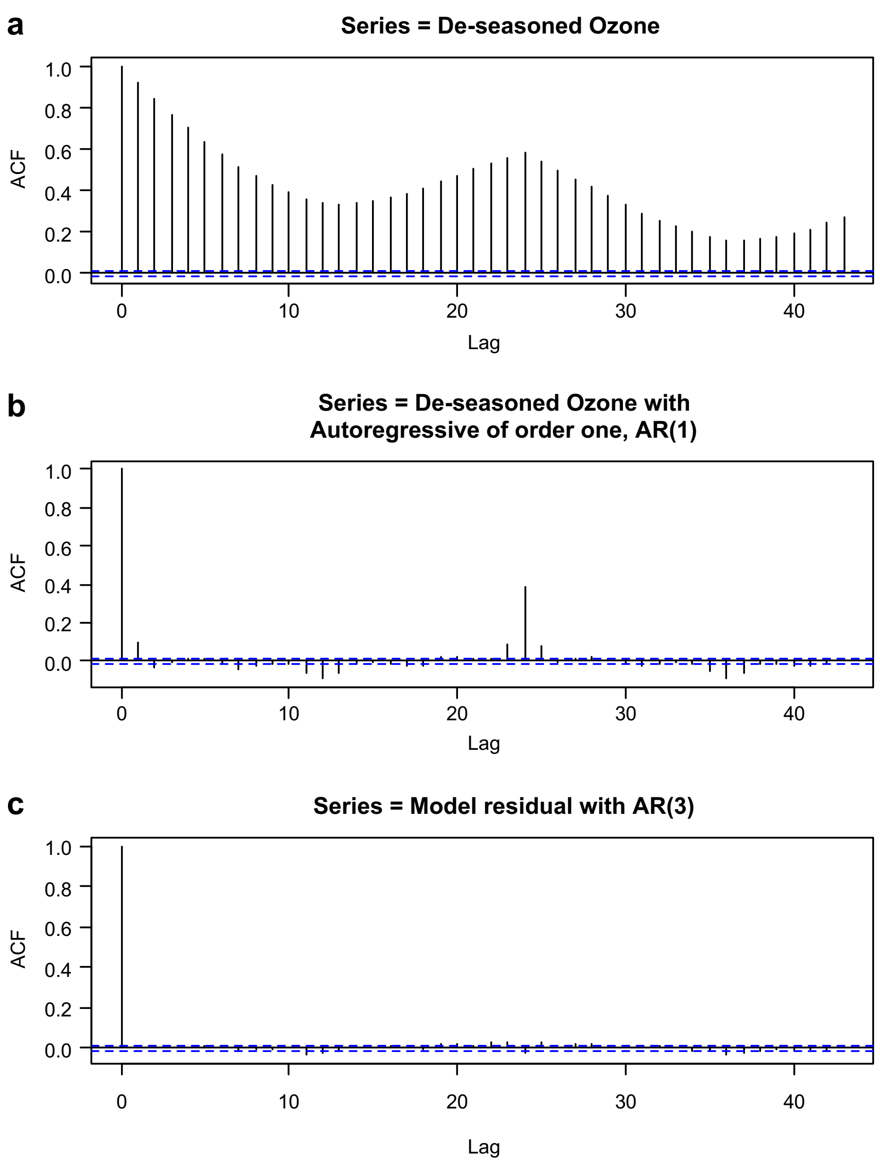

Fig. 6. Estimated auto-correlation functions for (a) ozone series with seasonal trends removed (b) residual series from fitting an AR(1) to the de-seasoned data and (c) residual series from fitting the model in equation (2).

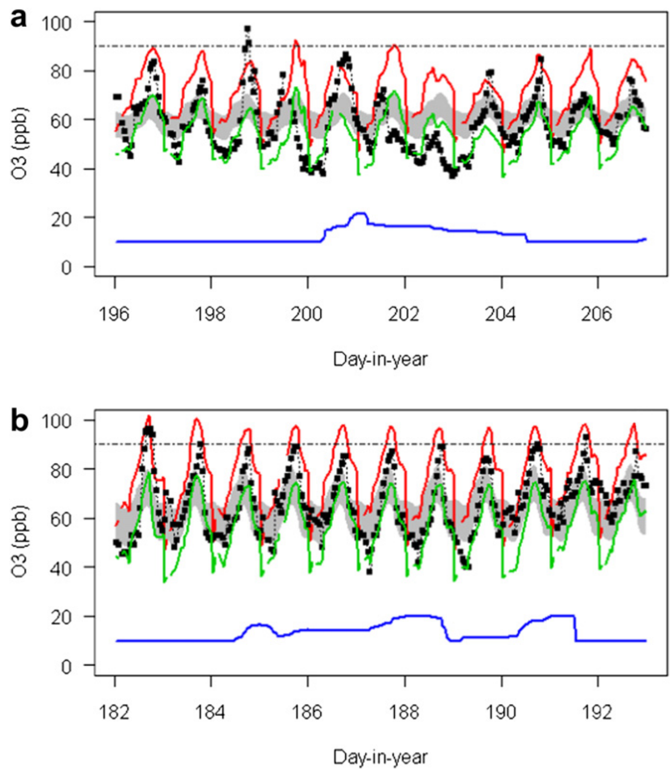

Fig. 7. Observed (black dots); forecasted 50th (green) and 99th (red) percentile ozone values (a) for July 15–25, 2006 at Yosemite – Turtleback – site and (b) for July 1–11 at Ash Mountain site. Gray band is the historic average level for the given site. Blue curve at the bottom indicate the fire activity level as measured by the median BlueSky value for the last 48 h. Dashed line at top is the critical 90 ppb level. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

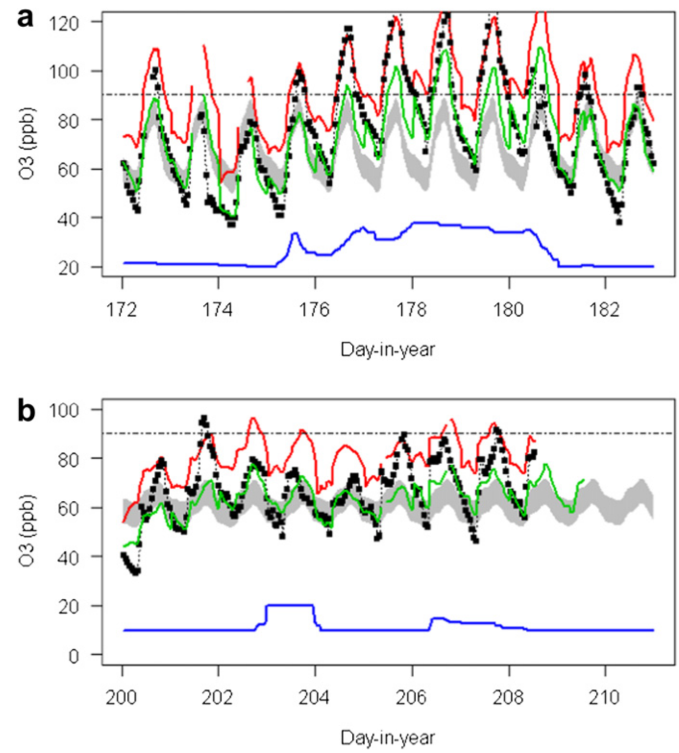

Fig. 8. Observed (black dots); forecasted 50th (green) and 99th (red) percentile ozone values (a) for June 20–30, 2008 at Lower Kaweah – site and (b) for July 18–28 at TurtleBack Dome site. Gray band is the historic average level for the given site. Blue curve at the bottom indicate the fire activity level as measured by the median BlueSky value for the last 48 h. Dashed line at top is the critical 90 ppb level. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

These fires were detected by the BlueSky model as indicated by the above-zero values of the modeled PM levels at the two Yosemite sites (Fig. 7a). The amount of increase in ozone that was attributed to the fires, as estimated by the model, was a maximum of 2 ppb. On 5th July 2006 (day-in-year = 186) there was a fire in Sequoia National Park that burned 250 ha. This was detected by the BlueSky model as indicated by the above-zero values of the modeled PM levels at the Ash Mountain – Sequoia NP site (Fig. 7b). For this period the estimated amount of increase in ozone that was attributed to the fire was a maximum of 1.6 ppb.

During the period June 20–30, 2008 there were 7 fires in Sierra Nevada over 40 ha. On June 21, 2008 three fires were discovered that eventually burned 243; 890 and 11,700 ha, respectively. Both the observed and predicted ozone values during this period are elevated at the Lower Kaweah site (Fig. 8a). On June 28 a fire was detected in the southern part of Sequoia National Park that eventually burned approximately 15,000 ha. This was the largest fire in the region (Fig. 2). This fire appears to have no effect on either observed or predicted ozone level or the BlueSky value at the Lower Kaweah site (Fig. 8a starting at day-in-year 180). The latter is probably due to many reasons, including the fact that the fire was not near the site; the fire, although eventually burned large number of hectares, was only a few hectares the day it was discovered (day 180); the site might not have been downwind from the fire. These are some of the difficulties encountered when trying to relate a given fire to changes in ozone levels. However, the statistical model was able to predict next-day ozone levels with confidence bounds. On July 18, 2008 a fire was detected in the region that eventually burned about 3000 ha. The effect of this fire at the Turtleback Dome site was minimal as indicated in Fig. 8b.

The hourly forecasts may also be used to predict next-day 8-h moving average levels. California ambient air-quality standard for ozone is 75 ppb for the forth highest 8 h concentration averaged over three years (http://www.arb.ca.gov/research/aaqs). As an example we produced predicted 8-h average levels for three periods in 2008 and for the Lower Kaweah site in Sequoia National Park (Fig. 9). We note that the forecasted next-day 50th and 99th percentiles appear to give a good fit to the actual observed levels. We anticipate forecasts such as those in Fig. 9 to be useful to air resource managers in particular since ozone values from the four monitoring sites in this study are used to support decisions regarding air quality and prescribed burns in the Class-I areas of the Sierra Nevada and other sensitive areas.

5. Conclusions

We have performed statistical analyses of multi-year ozone and meteorological data at four critical sites in Sierra Nevada. Using long-term historical data, we established the average ozone pattern for each site. At all four sites, averaged ozone pattern shows a strong seasonal cycle with maximum ozone values occurring in late summer and minimum in winter. The ozone levels are generally higher at sites in the Sequoia and Kings Canyon National Park than those in the Yosemite National Park, suggesting the importance of transport from the Central Valley to the western slopes of the Sierra Nevada. All sites also show a diurnal ozone cycle, but the

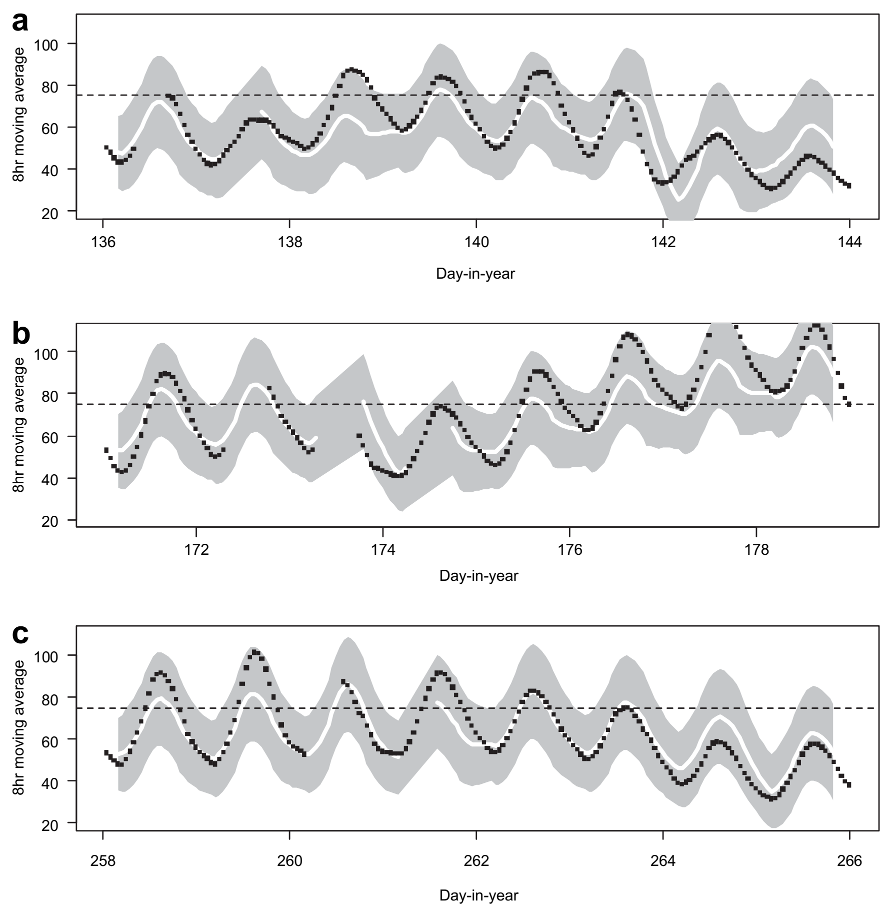

Fig. 9. Observed (black dots) and forecasted 99% CL (gray band) for 8 h moving average ozone values for (a) 15–22 May 2008; (b) 19–26 June, 2008 and (c) 14–21 September, 2008 at the Lower Kaweah site in Sequoia National Park. The dashed line at 75 ppb indicates the critical 8 h California ambient air-quality standard for ozone.

amplitudes of the diurnal cycle vary significantly between sites, possibly due to differences in topography.

The sources for ozone variation are quantified using simultaneous measurements of ozone and meteorological variables in the past three years. As expected, the majority of ozone variation at these sites may be attributed to temperature variation. It was found that wind directions from south and southwest appear to increase ozone levels above the seasonal average levels, suggesting the importance of transport of ozone precursors from the sources in the Central Valley.

To assess the contribution of smoke to ozone variations, we used PM concentrations predicted using the BlueSky smoke dispersion model that incorporated emissions from wildland and prescribed fires in the region. The statistical analysis detected a small but significant effect of fires on ozone variation. Ozone levels appear to increase on average when fire activity in the region increases, but the overall effect, however, seems to be small compared to the variations due to meteorological factors and sources other than fires.

We have demonstrated that accurate forecasts of next-day hourly ozone levels may be achieved by using the estimates from a time series model with previous-day ozone values, historic average values, expected local weather and modeled PM values as explanatory variables. The model-produced forecasts accounted for 68% of the variability in next-day's diurnal ozone levels. For sites with no historic data, forecasting with estimates from a time series model, with only previous day's values as explanatory variables, accounted for 60% of the variability. Comparisons of the model predictions with actual observed ozone values indicate that the model had considerable skill in forecasting ozone for the next 24 h during the fire season.

The statistical model and computer program developed in this study can be implemented to produce forecasts in real time. The forecasts may support decision making by land and air resource managers regarding air quality and prescribed burns in the Class-I areas of the Sierra Nevada and other sensitive areas.

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