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# Analysing the effects of the 2002 McNally fire on air quality in the San Joaquin Valley and southern Sierra Nevada, California

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**Abstract.** Smoke from wildfires can expose individuals and populations to elevated levels of particulate matter (PM) and ozone ($O_3$). Between 21 July and 26 August 2002, the McNally Fire burned over 150 000 acres (61 000 ha). The fire occurred in the Sequoia National Forest, in the southern Sierra Nevada of California. This study evaluated the effects of the McNally Fire on air quality, specifically particles <10 μm in diameter ($PM_{10}$) and $O_3$. Downwind of the fire on the eastern side of the Sierra Nevada, 24-h concentrations of $PM_{10}$ more than doubled. The $PM_{10}$ federal standard was exceeded four times during the fire. Violations of the California $PM_{10}$ standard increased drastically during the fire. The California $PM_{10}$ standard was violated six times before the fire and 164 times during the fire. Most of the $PM_{10}$ exceedances occurred at the Kernville Work Center and sites east of the fire. Compared with the other sites, the highest 2-week average $O_3$ concentrations occurred in the eastern part of the Sierra Nevada and north of the fire, where $O_3$ increased by a factor of two at two locations.

**Additional keywords:** ozone, particulate matter, smoke.

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## Introduction

Wildland fires are complex combustion processes dependent on fuel loading and fire behaviour that change over time and with weather conditions. In the United States, the patterns of fire frequency, intensity and severity have been gradually altered by the prevailing management strategy of fire suppression, which has contributed to conditions that encourage high-severity wildfires (Radke *et al.* 2001; USDA Forest Service 2001; Syphard *et al.* 2007). Smoke from wildfires has been increasing owing to the increased size of significant wildfires during the past decade (Jaffe *et al.* 2008). There is a growing awareness that smoke from wildfires can expose individuals to elevated levels of the criteria pollutants (pollutants with established air-quality standards under the Clean Air Act), with significant increases for particulate matter (PM) and ozone ($O_3$) (Chandra *et al.* 2002; DeBell *et al.* 2004; Linping *et al.* 2006; McMeeking *et al.* 2006; Pfister *et al.* 2008).

The San Joaquin Valley and portions of the southern Sierra Nevada in California are two of the most polluted regions in the United States (Cisneros and Perez 2007). Monitoring sites in Yosemite, Sequoia and Kings Canyon National Parks and Sequoia and Sierra National Forest, which are located in the southern Sierra Nevada, often violate the federal 8-h health standards for $O_3$ (Cisneros and Perez 2007; USDI NPS 2008).

A major concern for air regulatory and land-management agencies is the production of PM and $O_3$ precursors from wildfire, particularly because wildfire smoke can be transported long distances and across counties, states, countries and continents. Smoke from fire in North America has been shown to travel thousands of kilometres from Canada to the south-eastern United States (Wotawa and Trainer 2000; DeBell *et al.* 2004; Sapkota *et al.* 2005), and from Alaska and Canada to Texas (Morris *et al.* 2006). Smoke from fire has also travelled across continents, from Canada to Europe (Forster *et al.* 2001) and from Russia to different locations in the northern hemisphere such as Alaska, Canada, Scandinavia and eastern Europe, and even back to Russia in ~17 days (Damoah *et al.* 2004). Smoke from fire is composed of hundreds of chemicals in gaseous, liquid and solid forms (Ottmar and Reinhardt 2000; Urbanski *et al.* 2009). Particulate matter produced during catastrophic fire events is of great concern because it can adversely affect public health and can be one of the principal causes of visibility reduction in National Forests and Parks during the summer (Park *et al.* 2007). Burning of forests also releases large amounts of $O_3$ precursors, which has the potential to generate substantial concentrations of $O_3$ downwind of the fire (Cheng *et al.* 1998).

Both ambient $O_3$ and PM have detrimental effects on human health, with $O_3$ causing decreased lung function, asthma exacerbation and even premature mortality (Broeckaert *et al.* 2000). PM concentrations are associated with increased mortality and morbidity, reduced lung function, increased respiratory symptoms (such as chronic cough or bronchitis), aggravated respiratory and cardiovascular disease, eye and throat irritation, coughing, breathlessness, blocked and runny noses, and skin rashes (Radojevic and Hassan 1998; US EPA 2004). Additionally PM affects visual air quality and contributes to regional haze (Malm *et al.* 2000; Park *et al.* 2007). Visual air quality in national parks and wilderness areas receives special protection under the Clean Air Act. The 1977 amendments to the Act established a national goal to prevent and remedy any future and existing visibility impairment resulting from man-made air pollution to Class I Federal areas while recognising wildfire as a natural background source. Contributions from wildfire to air quality and the effects on human health from different fire size and intensity scenarios are not understood and thus not easily accounted for by regulators. This lack of scientific understanding of the effect on air quality from different wildfire events in the Sierra Nevada has led to confusion and inconsistency by regulators as to what is the best strategy to protect human health.

To meet air-quality regulatory and management requirements for the communities that surround the southern Sierra Nevada, more information is needed regarding fire effects on air quality. New methods and information are necessary to better assess, monitor and predict smoke plumes and the resulting contributions to pollutant concentrations downwind of fires. The McNally Fire was a large, high-intensity wildfire that burned over 60 000 ha (150 000 acres). It occurred in the Sequoia National Forest during the period of 21 July 2002 through 26 August 2002. Smoke from this fire was transported hundreds of kilometres within California and across the state boundary, affecting air quality and impairing visibility locally and regionally.

It is hypothesised that the McNally Fire affected air quality in the surrounding areas. This study evaluates and quantifies the magnitude of the effects of the McNally Fire on air quality, specifically $PM_{10}$ (particulate matter smaller than 10 μm in diameter) and $O_3$. This paper presents measurements of air pollutants before and during the fire to evaluate the extent and duration of wildfire smoke and provides insight into the magnitude of the effect of smoke on air quality. Wildfire episodes are hard to predict; therefore, this study took advantage of existing monitoring efforts available during the time of the fire.

## Methods

Particulate matter ($PM_{10}$) and $O_3$ concentrations were measured in California's Central Valley, the Owens Valley in the Great Basin and the Sierra Nevada during the McNally Fire. Site locations are shown in Fig. 1. The fire burned areas from 1200 to 2900 m in elevation (Fig. 1). Air-quality data were provided by the California Air Resources Board (CARB) network (see http://www.arb.ca.gov/aqmis2/aqmis2.php, accessed 24 July 2012), Interagency Monitoring of Protected Visual Environments (IMPROVE), Great Basin Unified Air Pollution Control District (GBUAPCD), Sequoia National Park (SNP) and the US Forest Service (USFS). Sites were selected based on data availability and the likelihood of the site being affected by the fire.

The CARB network consists of 11 sites, mostly located in large urban locations in Fresno and Kern Counties, except for Mojave and China Lake, which are located in smaller rural locations in Kern County. $PM_{10}$ data from the IMPROVE network included seven mountain sites. IMPROVE is a long-term monitoring program established to monitor visibility trends in National Parks and Wilderness areas located in the United States (http://vista.cira.colostate.edu/improve/, accessed 10 July 2012). Data from the GBUAPCD were collected from six sites located in the Owens Valley, east of the southern Sierra Nevada. The data from the USFS came from a monitoring station set up at the Kernville Work Center at the beginning of the fire. The Kernville Work Center is located in the Kern River drainage south of the fire and was the closest site to the fire.

Particulate matter mass concentrations in the CARB network sites were collected using SLAMS Hi-Volume Sierra Anderson sampler monitors (Andersen Instruments, Thermo Electron, Waltham, MA). The collections were performed every sixth day during 24 h, and $PM_{10}$ mass was determined gravimetrically. The IMPROVE network uses differences in pre- and post-collection weight of a Teflon filter to gravimetrically determine

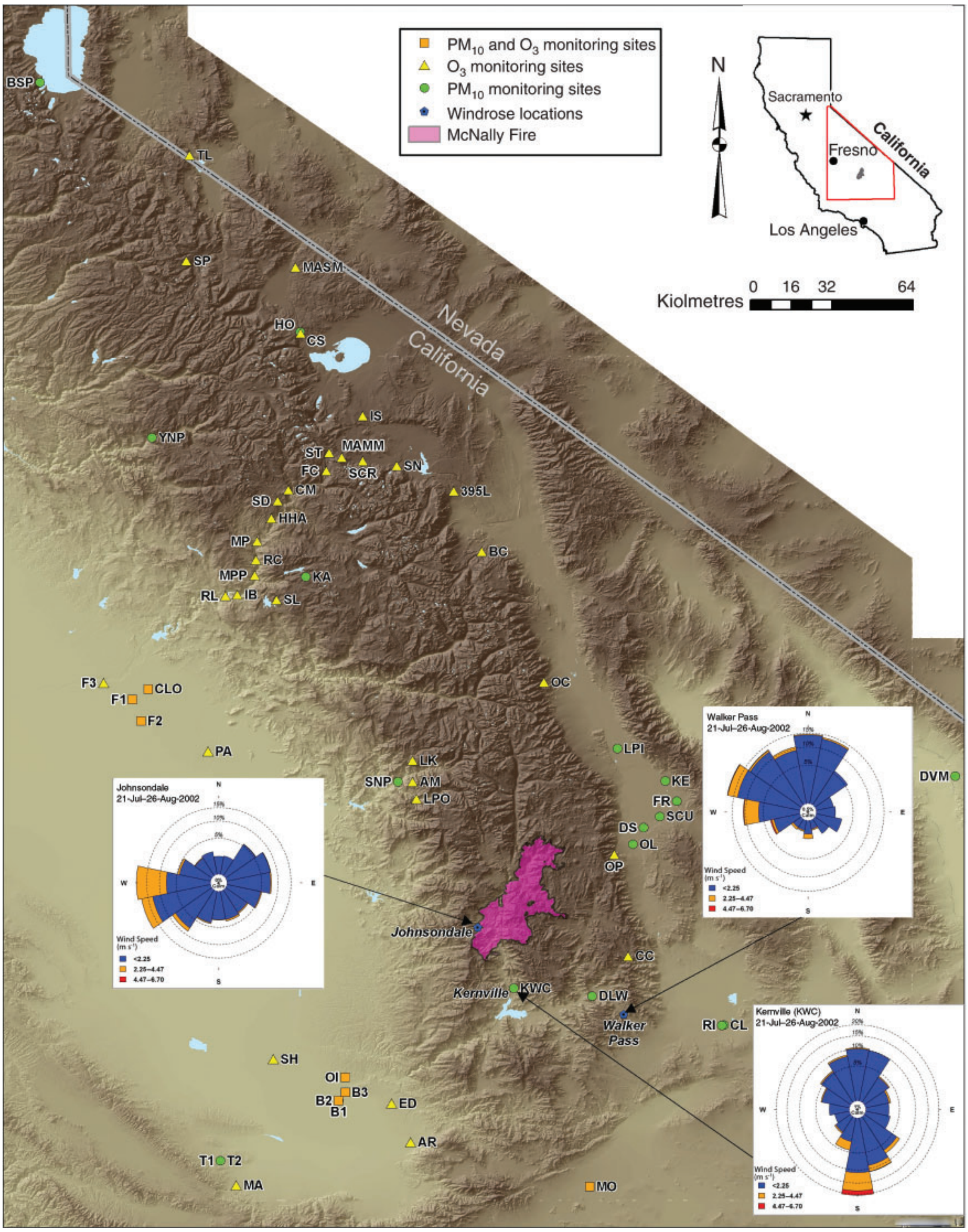


**Fig. 1.** Location of $PM_{10}$ (particulate matter smaller than 10 µm in diameter) and $O_3$ monitoring sites: Arvin (AR), Ash Mountain (AM), Bakersfield (B1), Bakersfield2 (B2), Bakersfield3 (B3), Bishop Creek (BC), Bliss State Park (BSP), Cattle Mountain (CM), China Lake (CL), Chimney Creek (CC), Clovis (CLO), Conway Summit (CS), Death Valley Monument (DVM), Dirty Sox (DS), Dome Lands Wilderness (DLW), Edison (ED), Fish Creek (FC), Flat Rock (FR), Fresno (F1), Fresno2 (F2), Fresno3 (F3), Hells Half Acre (HHA), Hoover (HO), Indiana Summit (IS), Italian Bar (IB), Keeler (KE), Kernville Work Center (KWC), Lone Pine (LPI), Lookout Point (LPO), Lower Kaweah (LK), Mammoth Mountain (MAMM), Mammoth Pool (MP), Mammoth Pool Powerhouse (MPP), Maricopa (MA), Masonic Mountain (MASM), Mojave (MO), Oak Creek (OC), Oildale (OI), Olancha (OL), Olancha Pass (OP), Parlier (PA), Redinger Lake (RL), Ridgecrest (RI), Rock Creek (RC), Sequoia National Park (SNP), Shafter (SH), Shaver Lake (SL), 395 Lookout (395 L), Shell Cut (SCU), Sherwin Creek (SCR), SNARL (SN), Sonora Pass (SP), Squaw Dome (SD), Starkweather (ST), Taft (T1), Taft2 (T2), Topaz Lake (TL) and Yosemite National Park (YNP).

**Table 1. Summary of mean, range and standard deviation of $PM_{10}$ (particulate matter smaller than 10 µm in diameter; $\mu g\ m^{-3}$) 24-h average concentrations before and during the fire**

Data in bold indicate statistically significant differences between pre-fire and during-fire concentrations at the 0.05 significance level using the Mann–Whitney Test. Statistical test was conducted on Great Basin Unified Air Pollution Control District (GBUAPCD) sites only

| Site (abbreviation, network) | Before fire | | | During fire | | |
|---|---|---|---|---|---|---|
| | *n* | Mean ± s.d. | Range | *n* | Mean ± s.d. | Range |
| GBUAPCD | | | | | | |
| Dirty Sox (DS) | 20 | **29 ± 15** | 15–81 | 37 | **56 ± 28** | 12–120 |
| Flat Rock (FR) | 20 | **20 ± 7** | 10–41 | 37 | **41 ± 22** | 10–100 |
| Keeler (KE) | 20 | **18 ± 6** | 11–36 | 37 | **43 ± 25** | 9–90 |
| Lone Pine (LPI) | 20 | **23 ± 7** | 15–42 | 37 | **52 ± 28** | 14–111 |
| Olancha (OL) | 20 | **27 ± 8** | 19–50 | 37 | **63 ± 35** | 17–167 |
| Shell Cut (SCU) | 20 | 20 ± 9 | 12–52 | 37 | 44 ± 24 | 11–90 |
| USFS | | | | | | |
| Kernville Work Center (KWC) | | NA | | 33 | 80 ± 33 | 47–178 |
| IMPROVE | | | | | | |
| Bliss State Park (BSP) | 7 | 6 ± 2 | 4–11 | 10 | 11 ± 4 | 3–17 |
| Death Valley Monument (DVM) | 7 | 25 ± 8 | 14–37 | 12 | 26 ± 13 | 6–53 |
| Yosemite National Park (YNP) | 7 | 13 ± 3 | 9–18 | 12 | 18 ± 4 | 12–24 |
| Dome Lands Wilderness (DLW) | 6 | 24 ± 3 | 20–29 | 8 | 36 ± 13 | 22–50 |
| Hoover (HO) | 7 | 9 ± 6 | 4–21 | 9 | 12 ± 4 | 6–19 |
| Sequoia National Park (SNP) | 7 | 26 ± 2 | 22–29 | 9 | 32 ± 12 | 6–45 |
| CARB | | | | | | |
| Bakersfield (B1) | 4 | 44 ± 9 | 35–56 | 6 | 56 ± 5 | 51–62 |
| Bakersfield2 (B2) | 4 | 43 ± 10 | 33–57 | 6 | 56 ± 4 | 52–63 |
| Bakersfield3 (B3) | 4 | 52 ± 8 | 43–61 | 6 | 68 ± 8 | 59–81 |
| China Lake (CL) | 5 | 21 ± 7 | 15–30 | 6 | 32 ± 12 | 19–50 |
| Clovis (CLO) | 4 | 35 ± 7 | 24–40 | 6 | 46 ± 11 | 34–62 |
| Fresno (F1) | 4 | 30 ± 7 | 20–35 | 6 | 38 ± 9 | 24–47 |
| Fresno2 (F2) | 4 | 42 ± 7 | 33–49 | 6 | 47 ± 11 | 34–65 |
| Mojave (MO) | 4 | 26 ± 4 | 20–31 | 9 | 33 ± 8 | 23–45 |
| Oildale (OI) | 4 | 37 ± 5 | 31–44 | 6 | 57 ± 9 | 46–69 |
| Ridgecrest (RI) | 4 | 27 ± 7 | 21–36 | 6 | 40 ± 15 | 18–62 |
| Taft (T1) | 4 | 33 ± 7 | 22–37 | 6 | 47 ± 7 | 38–57 |
| Taft2 (T2) | 4 | 32 ± 7 | 22–37 | 6 | 46 ± 6 | 39–55 |

$PM_{10}$ every 3 days. The six monitoring sites in the GBUAPCD collected data by the use of TEOM (tapered element oscillating microbalance) instruments (Rupprecht & Patashnick, Albany, NY) to measure $PM_{10}$ concentrations. $PM_{10}$ in the GBUAPCD network was collected as a 1-h average every day. An Environmental Beta Attenuation Monitor (EBAM; Met One Inc., Grants Pass, OR) was used by the USFS at the Kernville site. The EBAM use a vacuum pump to draw a sample of ambient air and deposits particles onto the filter paper. A carbon-14 source emits $\beta$ particles that pass through the tape and are counted by a detector. To determine the particulate mass, a $\beta$ count is taken before and after the sample is taken. The air flow measured is used to calculate the concentration. $PM_{10}$ in the USFS network was collected as a 1-h average every day.

Ozone data were collected during 2002 from 41 monitoring stations (Fig. 1), including 17 active monitoring stations and 24 passive monitoring stations. The CARB network provided $O_3$ data from 14 active monitoring sites. Sequoia National Park provided data from three active sites. The active $O_3$ monitoring sites used the US Environmental Protection Agency-approved collection and analysis method, employing an ultraviolet absorption instrument that provided 1-h average concentrations. Ozone 2-week averages were determined with passive samplers operated by the USFS Pacific Southwest Research Station in Riverside, California, for 24 sites. As $O_3$ passive samplers provide 2-week average $O_3$ concentrations, all of the active instruments' hourly $O_3$ concentrations were also averaged over the same 2-week period. The passive sampling methodology is explained in Arbaugh *et al.* (2001) and in Arbaugh and Bytnerowicz (2003). For the purpose of comparing $O_3$ 2-week averages across the entire study area sites were denoted as Urban (U: F1, F2, F3, CLO, PA, SH, OI, B1, B2, ED), Rural (R: AR, MA, MO), West Mountain Sites (WMS: LK, AM, LPO), San Joaquin River Drainage (SJRD: SL, RL, IB, MPP, RC, MP, HHA, SD, CM, FC, ST) and East Mountain Sites (EMS: TL, SP, MASM, CS, MAMM, SCR, IS, SN, 395 L, BC, OC, OP, CC) (Fig. 1).

## Results and discussion

### *$PM_{10}$*

The $PM_{10}$ 24-h mean concentration increased at every monitoring site in the GBUAPCD when compared with pre-fire concentrations measured in the Owens Valley east of the southern Sierra Nevada (Table 1). Dirty Sox, Flat Rock, Keeler, Lone Pine and Olancha experienced statistically significant

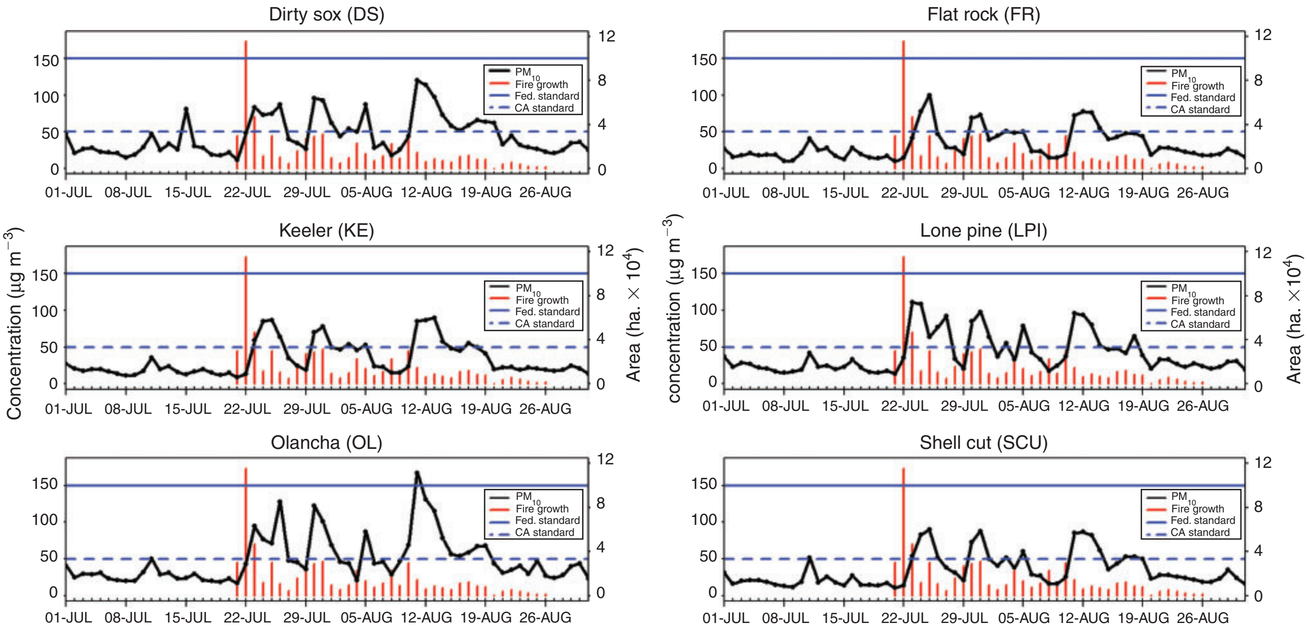


**Fig. 2.** $PM_{10}$ (particulate matter smaller than 10 μm in diameter) 24-h average concentrations in the Owens Valley (Great Basin Unified Air Pollution Control District, GBUAPCD).

increases ($P < 0.05$) in $PM_{10}$ during the fire. The biggest effect on $PM_{10}$ 24-h mean concentration occurred on 11 August at the Olancha and Dirty Sox sites, although the rate of fire growth for the McNally Fire was not the largest at this time (Fig. 2). HYSPLIT (Hybrid Single Particle Lagrangian Integrated Trajectory) (Draxler and Rolph 2011) back-trajectories (not presented in the current paper) indicated that smoke from the fire was transported towards these locations. Also, smoke plumes visible from MODIS (Moderate Resolution Imaging Spectroradiometer) satellite images were observed above these locations on 9 and 11 August (MODIS data provided by the USFS Remote Sensing Application Center, 2002). Without fire emissions, $PM_{10}$ 24-h averages were ~20–30 μg m$^{-3}$ (Fig. 2). However, when the fire started, $PM_{10}$ concentrations at these sites increased three to five times (Fig. 2). Similar daily temporal distribution patterns of $PM_{10}$ concentrations were seen for GBUAPCD sites in the Owens Valley east of the fire (DS, FR, KE, LPI, OL and SCU) with typical afternoon and evening high concentrations, whereas concentrations at the KWC site typically peaked in the late morning (Fig. 3). Fig. 3 presents the diurnal cycle for the GBUAPCD sites and the Kernville Work Center site. $PM_{10}$ concentrations at GBUAPCD sites were higher during the fire than the pre- and post-fire concentrations for all hours of the day, with peak hourly $PM_{10}$ concentrations typically occurring between 1500 and 1900 hours during the fire. Hourly $PM_{10}$ peak concentration at the Kernville Work Center site during the fire occurred between 0600 and 0900 hours.

The Kernville Work Center experienced the highest $PM_{10}$ concentrations, but comparisons with pre-fire levels were not possible because the monitor was installed at the onset of the fire. The Kernville Work Center is located south of the fire and was the closest site to the fire. Kernville is located on the Kern River at an elevation of ~1000 m. The Kern River runs in a north to south direction, which can be seen in Fig. 1. Winds typically flow down-canyon at night and up-canyon during the day, and wind direction during the fire followed this pattern. Smoke emissions from the McNally Fire were transported down-canyon at night and vented in the late morning to early afternoon. As the inversion in the drainage broke, smoke from the fire began moving up and out of the drainage. High concentrations of $PM_{10}$ were seen during the entire duration of the fire, with the highest hourly concentration of 603 μg m$^{-3}$ occurring on 27 July at 0900 hours.

The sample size for the CARB and IMPROVE sites is not sufficient to draw any statistical conclusions as to the effects of the McNally Fire on air quality. Thus, statistical tests are only relevant for sites in the GBUAPCD. Particulate matter increased at all monitoring sites in the GBUAPCD, but the increase was statistically significant only at five sites (Dirty Sox, Flat Rock, Keeler, Lone Pine and Olancha). The Yosemite National Park site showed an increase in $PM_{10}$ 24-h concentrations during the fire. It has been suggested by Cahill *et al.* (2005) and Carrico *et al.* (2005) that Yosemite air quality was affected mainly by southern Oregon fires and to a lesser extent by the McNally Fire. In the CARB network, the four sites closest to the fire (Bakersfield3, Oildale, Taft1 and Taft2) experienced what appeared to be higher levels of $PM_{10}$ but this could not be statistically determined. Therefore, CARB and IMPROVE sites data indicate no obvious effects on $PM_{10}$ that can be attributed to the McNally Fire.

Air-quality exceedances presented here use the current Federal and California standards for $PM_{10}$ and $O_3$. Table 2 shows the number of exceedances of the 24-h average California (50 μg m$^{-3}$) and Federal (150 μg m$^{-3}$) $PM_{10}$ standard.

During the pre-fire period, there were no exceedances of the Federal $PM_{10}$ standard. During the fire, the Federal standard was exceeded on four occasions. Three of the exceedances occurred

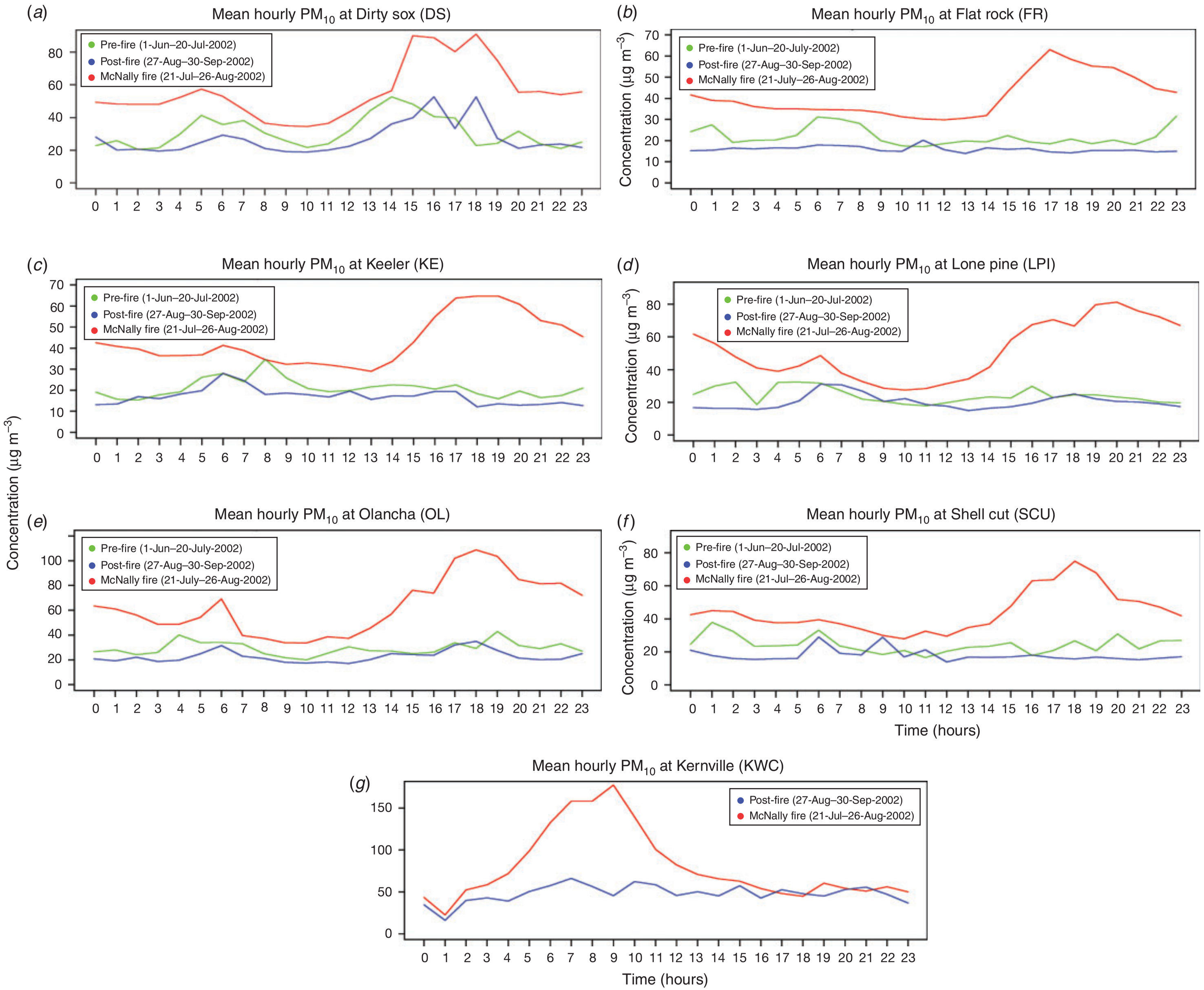


**Fig. 3.** Hourly average $PM_{10}$ (particulate matter smaller than 10 µm in diameter) diurnal cycle before, during and after the McNally Fire for the GBUAPCD (Great Basin Unified Air Pollution Control District) sites: (*a*) DS, (*b*) FR, (*c*) KE, (*d*) LPI, (*e*) OL, (*f*) SCU and the Kernville Work Center site (*g*) KWC (see Fig. 1 caption for site codes).

**Table 2. Count of exceedances of the California (50 µg $m^{-3}$) and the Federal (150 µg $m^{-3}$) ambient air quality standards for $PM_{10}$ (particulate matter smaller than 10 µm in diameter)**

| Site | Before fire | | During fire | |
|---|---|---|---|---|
| | 24-h mean (≥50 µg $m^{-3}$) | 24-h mean (≥150 µg $m^{-3}$) | 24-h mean (≥50 µg $m^{-3}$) | 24-h mean (≥150 µg $m^{-3}$) |
| Bakersfield (B1) | 1 | 0 | 6 | 0 |
| Bakersfield2 (B2) | 1 | 0 | 6 | 0 |
| Bakersfield3 (B3) | 2 | 0 | 6 | 0 |
| Bliss State Park (BSP) | 0 | 0 | 0 | 0 |
| China Lake (CL) | 0 | 0 | 1 | 0 |
| Clovis (CLO) | 0 | 0 | 2 | 0 |
| Death Valley Monument (DVM) | 0 | 0 | 1 | 0 |
| Dirty Sox (DS) | 1 | 0 | 20 | 0 |
| Yosemite National Park (YNP) | 0 | 0 | 0 | 0 |
| Dome Lands Wilderness (DLW) | 0 | 0 | 2 | 0 |
| Flat Rock (FR) | 0 | 0 | 10 | 0 |
| Fresno (F1) | 0 | 0 | 0 | 0 |
| Fresno2 (F2) | 0 | 0 | 2 | 0 |
| Hoover (HO) | 0 | 0 | 0 | 0 |
| Keeler (KE) | 0 | 0 | 15 | 0 |
| Kernville Work Center (KWC) | NA | | 32 | 3 |
| Lone Pine (LPI) | 0 | 0 | 15 | 0 |
| Mojave (MO) | 0 | 0 | 0 | 0 |
| Oildale (OI) | 0 | 0 | 5 | 0 |
| Olancha (OL) | 1 | 0 | 19 | 1 |
| Ridgecrest (RI) | 0 | 0 | 2 | 0 |
| Sequoia National Park (SNP) | 0 | 0 | 0 | 0 |
| Shell Cut (SCU) | 0 | 0 | 16 | 0 |
| Taft (T1) | 0 | 0 | 2 | 0 |
| Taft2 (T2) | 0 | 0 | 2 | 0 |

**Table 3. Daily 8-h maximum $O_3$ concentrations (ppb) with descriptive statistics and standard deviation before and during the fire**
During the fire, a trough (low-atmospheric pressure) occurred; the data for these days (31 July–4 August) are not included in the comparisons. Data in bold indicate statistically significant differences ($P < 0.05$) when comparing the before-fire and during-fire periods

| Site name | Before fire | | | During fire | | |
|---|---|---|---|---|---|---|
| | *n* | (Mean ± s.d.) | Range | *n* | (Mean ± s.d.) | Range |
| Arvin (AR) | 19 | (91 ± 15) | 61–118 | 32 | (97 ± 15) | 72–121 |
| Ash Mountain (AM) | 19 | (91 ± 8) | 80–109 | 32 | (92 ± 8) | 76–108 |
| Bakersfield (B1) | 20 | (72 ± 15) | 41–101 | 31 | (80 ± 16) | 26–106 |
| Bakersfield2 (B2) | 20 | (77 ± 13) | 51–103 | 31 | (84 ± 12) | 51–106 |
| Clovis (CLO) | 18 | (78 ± 12) | 64–105 | 31 | (77 ± 15) | 52–103 |
| Edison (ED) | 20 | **(81 ± 13)** | 60–109 | 32 | **(90 ± 14)** | 61–116 |
| Fresno (F1) | 20 | (78 ± 16) | 45–112 | 31 | (82 ± 14) | 53–111 |
| Fresno2 (F2) | 20 | (80 ± 18) | 42–120 | 32 | (82 ± 15) | 54–110 |
| Fresno3 (F3) | 20 | (88 ± 18) | 53–129 | 32 | (90 ± 14) | 64–120 |
| Lookout Point (LPO) | 16 | (91 ± 6) | 83–107 | 32 | (95 ± 10) | 77–177 |
| Lower Kaweah (LK) | 20 | (88 ± 7) | 60–100 | 32 | (93 ± 11) | 74–118 |
| Maricopa (MA) | 20 | (79 ± 13) | 53–109 | 32 | (86 ± 15) | 45–110 |
| Mojave (MO) | 20 | (72 ± 12) | 53–102 | 31 | (79 ± 13) | 48–102 |
| Oildale (OI) | 20 | **(76 ± 13)** | 51–102 | 32 | **(84 ± 12)** | 59–104 |
| Parlier (PA) | 20 | (86 ± 15) | 58–124 | 32 | (92 ± 14) | 67–118 |
| Shafter (SH) | 20 | **(73 ± 12)** | 47–93 | 32 | **(80 ± 12)** | 56–100 |
| Shaver Lake (SL) | 20 | (79 ± 10) | 56–96 | 32 | (82 ± 11) | 60–105 |

at KWC and one at OL. Smoke was visible in satellite imagery throughout the upper Kern River drainage including Kernville. The KWC monitor was likely representative of areas that were close to the fire and where emissions did not readily mix with upper air parcels and remained relatively concentrated at ground level.

The California $PM_{10}$ standard was exceeded six times in the pre-fire period. Four of the exceedances occurred in Bakersfield, which is recognised as one of the most polluted sites in the south valley floor for $PM_{10}$. During the fire, the number of exceedances increased to 164. The most obvious reason is the fire. The most affected site was the Kernville Work Center, follow by the sites located east of the fire. The major effects on air quality of the McNally Fire occurred in the eastern Sierra Nevada closest to the fire and downwind of the fire in the Owens Valley.

### *Eight-hour maximum ozone concentrations*

All of the active $O_3$ monitoring sites are located in a non-attainment area (i.e. an area that does not meet the US EPA's standard for $O_3$, PM or any of the six criteria pollutants). During 31 July and 4 August, there was a weak low-pressure system that developed over the eastern Pacific and moved into the San Joaquin Valley. This trough enhanced the marine layer, resulting in lower temperatures across the San Joaquin Valley Air Basin (S. Ferreira, pers. comm., 2006). Ozone levels were lower at all sites between 1 and 5 August (Table 3; Fig. 4). Therefore, $O_3$ concentrations are presented for two time periods: before the fire and during the fire without the trough. Back-trajectories calculated for the Fresno site using HYSPLIT (Draxler and Rolph 2011) for the months of July and August (not presented in the present paper) indicate no air parcel originating from the McNally Fire area or east of the Central Valley. Back-trajectory simulations during the low-pressure system for the Fresno site indicated that for 2 August through 4 August, the air parcels were coming from less polluted areas such as Monterey Bay, Carmel Valley and Paso Robles. The decrease in the 8-h $O_3$ maximum during the trough is attributed to lower temperatures, a small decrease in solar radiation and possibly air parcels moving into the valley originating from lower-emission sources (Fig. 4).

When comparing the pre-fire and during-fire periods without the trough, statistically significant increases of $O_3$ ($P < 0.05$) were found at three sites: Edison, Oildale and Shafter (Table 3). At Edison, there was an increase in the mean daily 8-h $O_3$ maximum during the fire of 9 ppb. Oildale experienced a mean increase of 8 ppb for the daily 8-h $O_3$ maximum. The increase of the daily 8-h maximum during the fire at the Shafter site was 7 ppb. Table 3 also shows that sites in Bakersfield, Maricopa and Mojave experienced higher daily 8-h maximum $O_3$ during the fire. All the sites above are close in proximity to each other and to the fire. Even though the data indicate an increase in daily 8-h maximum $O_3$ concentrations, it is not possible to attribute the increase to any one source because of the complicated chemical process of $O_3$ formation. The small increase in 8-h maximum $O_3$ has the potential to be misinterpreted owing to factors beyond the scope of this paper, such as increased local production of $O_3$ precursors (oxides of nitrogen, $NO_x$, and volatile organic compounds, VOCs) and the unusually cool meteorological conditions witnessed during this event. HYSPLIT (Draxler and Rolph 2011) back-trajectories (not shown in this paper) indicated that no air parcel originating from the McNally Fire area reached these sites.

Fig. 4 presents information about the maximum daily 8-h mean $O_3$ for the duration of the fire and compares it with the distribution of maximum daily 8-h mean $O_3$ over the same time period for 1997 through 2001 and 2003 through 2005. In Fig. 4, the daily 8-h maximum $O_3$ is shown for Edison, Oildale and

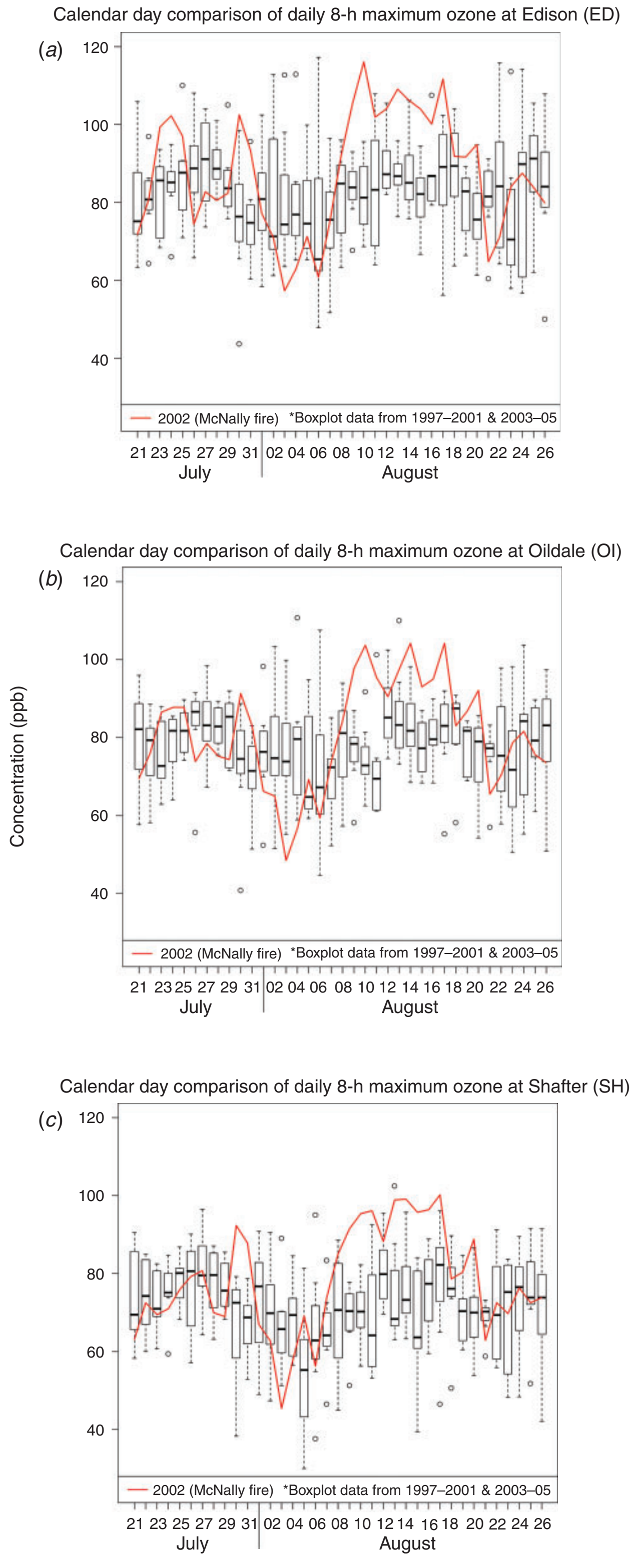


**Fig. 4.** Comparison of the daily 8-h $O_3$ (ppb) maximum during the McNally Fire and the calendar 8-h $O_3$ maximum box plots during 21 July–26 August for the years 1997–2001 and 2003–05 for: (*a*) ED, (*b*) OI, and (*c*) SH (see Fig. 1 caption for site codes).

**Table 4. Two-week ozone averages (ppb) for 40 monitoring stations before (first period) and during (all other periods) the McNally fire 2002**
See Fig. 1 for monitor locations

| Sites | 2–16 July | 16–30 July | 30 July–13 August | 13–28 August | Mean (s.d.) |
|---|---|---|---|---|---|
| Urban | | | | | |
| Bakersfield (B1) | 45 | 49 | 44 | 47 | 46 (2) |
| Bakersfield2 (B2) | 43 | 44 | 43 | 44 | 44 (1) |
| Clovis (CLO) | 51 | 46 | 47 | 47 | 48 (2) |
| Edison (ED) | 53 | 54 | 58 | 59 | 56 (3) |
| Fresno (F1) | 52 | 50 | 49 | 50 | 50 (1) |
| Fresno2 (F2) | 51 | 49 | 50 | 49 | 50 (1) |
| Fresno3 (F3) | 61 | 57 | 58 | 57 | 58 (2) |
| Oildale (OI) | 51 | 52 | 54 | 56 | 53 (3) |
| Parlier (PA) | 56 | 54 | 54 | 55 | 55 (1) |
| Shafter (SH) | 40 | 44 | 42 | 46 | 43 (2) |
| Rural | | | | | |
| Arvin (AR) | 62 | 65 | 67 | 68 | 66 (3) |
| Maricopa (MA) | 59 | 62 | 67 | 69 | 64 (5) |
| Mojave (MO) | 65 | 57 | 68 | 73 | 66 (7) |
| West Mountain sites | | | | | |
| Ash Mountain (AM) | 73 | 74 | 75 | 82 | 76 (4) |
| Lower Kaweah (LK) | 72 | 72 | 75 | 87 | 77 (7) |
| Lookout Point (LPO) | 74 | 74 | 75 | 84 | 77 (5) |
| San Joaquin River Drainage | | | | | |
| Cattle Mountain (CM) | 74 | 80 | 79 | 94 | 82 (9) |
| Fish Creek (FC) | 58 | 62 | 90 | 78 | 72 (15) |
| Hells Half Acre (HHA) | 80 | 80 | | 95 | 85 (9) |
| Italian Bar (IB) | 84 | 80 | 87 | 95 | 87 (6) |
| Mammoth Pool (MP) | 82 | 70 | 79 | 80 | 78 (5) |
| Mammoth Pool Powerhouse (MPP) | 83 | 89 | 90 | 97 | 90 (6) |
| Redinger Lake (RL) | 88 | 89 | 94 | 98 | 92 (5) |
| Rock Creek (RC) | 70 | 72 | 76 | 92 | 78 (10) |
| Shaver Lake (SL) | 56 | 56 | 58 | 66 | 59 (5) |
| Squaw Dome (SD) | 70 | 76 | 87 | 186 | 105 (55) |
| Starkweather (ST) | 61 | 61 | 41 | 88 | 63 (19) |
| East Mountain sites | | | | | |
| Bishop Creek (BC) | 61 | | 78 | 79 | 73 (10) |
| Chimney Creek (CC) | 64 | 67 | 50 | 80 | 65 (12) |
| Conway Summit (CS) | 62 | | 100 | 78 | 80 (19) |
| Indiana Summit (IS) | 64 | | 55 | 75 | 65 (10) |
| Mammoth Mountain (MAMM) | 79 | 90 | 62 | 132 | 91 (30) |
| Masonic Mountain (MASM) | 50 | | 53 | 63 | 55 (7) |
| Oak Creek (OC) | 66 | 62 | 67 | 77 | 68 (6) |
| Olancha Pass (OP) | 68 | | 167 | 80 | 105 (54) |
| Sherwin Creek (SCR) | 61 | | 95 | 86 | 81 (18) |
| SNARL (SN) | 46 | 58 | 63 | 76 | 61 (12) |
| Sonora Pass (SP) | | 51 | 57 | 59 | 56 (4) |
| Topaz Lake (TL) | | | | 106 | 106 |
| 395 Lookout (395 L) | 59 | | 59 | 68 | 62 (5) |

Shafter. These sites experienced similar effects during the fire. The sites are located north and east of the city of Bakersfield and are small towns, but for the purposes of this paper considered urban locations. The maximum daily 8-h mean $O_3$ at Edison increased during 23–25 July, 30–31 July and 8–20 August. The highest 8-h mean $O_3$ at Oildale during the fire occurred during 23–25 July, 30–31 July and 9–17 August. The Shafter site experienced the highest mean daily 8-h $O_3$ levels during 30–31 July and 7–20 August.

Given the complicated nature of $O_3$ formation chemistry, Fig. 4 presents the distribution of $O_3$ for multiple years. Fig. 4 suggests that the maximum daily mean 8-h $O_3$ occurring during the fire is higher than the distribution of ozone occurring during the same time period for other years. Thus, it is possible but not conclusive that a portion of the increases of the daily 8-h maximum $O_3$ at locations close to the fire may be a result of the fire.

### *Ozone 2-week averages*

Before the McNally Fire started, the highest $O_3$ 2-week average concentration occurred in the mountain (WMS, SJRD, EMS) and rural monitoring sites (Table 4). Mountain, rural and urban monitoring sites may experience different diurnal $O_3$ cycles and have different sources of $O_3$ precursors. Adequate air-quality modelling is not possible in these mountain locations because of the complex topography and lack of real-time $O_3$, $NO_x$, VOC and weather information. The approach presented here is an attempt to understand the effects of fire on ground-level $O_3$ using the limited available data in this area.

Before the fire (2–16 July 2002), the maximum 2-week average did not exceed 88 ppb. During the fire (30 July–28 August 2002), the EMS and SJRD sites experienced higher 2-week average ozone concentrations. The highest concentrations occurred in Olancha Pass in the eastern part of the Sierra Nevada, downwind from the fire, with a maximum 2-week average of 167 ppb. Table 4 provides evidence of the connection between large wildfire events and $O_3$ generation.

In the Sierra Nevada, owing to the proximity of the California Central Valley urban pollutant plume, which is rich in $NO_x$, there is a strong potential for generation of very high $O_3$ concentrations when elevated concentrations of VOCs, $NO_x$ and CO are present as a result of forest fires (Cheng *et al.* 1998). The data in Table 4 show that 2-week $O_3$ concentrations at Olancha Pass and Squaw Dome, which are sites located in the EMS and SJRD respectively, doubled.

Ozone concentrations in urban locations were similar for the pre-fire and during-fire 2-week periods (Table 4). Whereas the mountain (WMS, SJRD, EMS) and rural monitoring sites experienced effects from the McNally Fire, the urban locations did not (Table 4). The McNally Fire (Table 3) apparently did not affect the seven $O_3$ monitoring locations Fresno, Fresno2, Fresno3, Clovis, Parlier, Bakersfield and Bakersfield2, which were categorised as urban locations. Increases of $O_3$ at the other urban locations of Shafter, Oildale and Edison were potentially affected to a low extent by the McNally Fire.

### *Human health implications*

There are possible negative effects on human health caused by the fire with the observed $PM_{10}$ exceedances in the small communities near the fire, especially Kernville. It is evident that large wildfires pose a threat to human health, particularly when more than 2000 acres (809 ha) are burned per day, in communities near and downwind of the fire (Fig. 2). More work to prevent high-intensity large-area fires is needed in this area, preventing the higher daily fuel consumption that is evidently one of the biggest drivers of ambient particulate matter, which may be transported to downwind communities. More information is needed to understand the effects of biomass burned per day, fire intensity, distance from the fire, transport and dispersal on air quality to better characterise and understand public health impacts.

Significant effects to the forest ecosystem and to humans, who live, work and take recreation in the forest, are likely to occur during large fire events. Furthermore, the dynamics between urban pollutants and smoke from high-intensity fires need to be better understood. This is an important management issue considering that wildfire size and intensity are increasing because of past fire-suppression policies and a rise in spring and summer temperatures leading to earlier spring snowmelt and longer fire seasons (Westerling *et al.* 2006; Miller *et al.* 2009), resulting in overall increased exposure to unhealthy levels of air pollutants. From this perspective, better information about the magnitude of wildland fire effects on $O_3$ and PM in mountain locations is needed for science-based management of forests and air resources.

## Summary and conclusions

Mean 24-h average concentrations of $PM_{10}$ increased at every monitoring site in the study, with the highest $PM_{10}$ 24-h concentrations occurring on the eastern side of the Sierra Nevada, downwind of the fire. During the fire, 24-h concentrations of $PM_{10}$ more than doubled at sites on the eastern side of the Sierra Nevada. The Kernville site, which was the closest to the fire, experienced the highest 24-h average concentration of $PM_{10}$ of all sites in this study.

The results presented here show that the McNally Fire increased the $O_3$ 2-week average concentrations at some sites downwind of the fire by a factor of two. This increase of $O_3$ concentration is likely linked to high levels of $O_3$ formation precursors – $NO_x$ from urban areas of the upwind San Joaquin Valley and $NO_x$, CO and VOCs emitted from the fire. It is noted from this study that big wildfires could significantly increase the already high $O_3$ concentrations in mountain locations downwind of the fire.

Continuous monitoring by state and federal agencies proved to be valuable in determining the effects of the McNally Fire. However, there is a need for monitoring networks to be expanded in mountain areas, because those were the most affected by this high-intensity wildfire. For that reason, a network of densely distributed passive samplers aided by real-time portable $O_3$ monitors and portable PM monitors is essential for evaluating effects of wildland fire on ambient air quality. Large fire size and high fire intensity in combination with urban pollutants from the Central Valley may be the leading cause of increased concentrations of ozone and PM in rural mountain communities of the Sierra Nevada. A return to historic fire size and intensity may be the best solution for reducing $O_3$ and PM exposure in the Sierra Nevada.

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