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Analysis of the effects of combustion emissions and Santa Ana winds on ambient ozone during the October 2007 southern California wildfires

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ABSTRACT

Combustion emissions and strong Santa Ana winds had pronounced effects on patterns and levels of ambient ozone (O_3) in southern California during the extensive wildland fires of October 2007. These changes are described in detail for a rural receptor site, the Santa Margarita Ecological Reserve, located among large fires in San Diego and Orange counties. In addition, O_3 changes are also described for several other air quality monitoring sites in the general area of the fires. During the first phase of the fires, strong, dry and hot northeasterly Santa Ana winds brought into the area clean continental air masses, which resulted in minimal diurnal O_3 fluctuations and a 72-h average concentration of 36.8 ppb. During the-second phase of the fires, without Santa Ana winds present and air filled with smoke, daytime O_3 concentrations steadily increased and reached 95.2 ppb while the lowest nighttime levels returned to $\sim$0 ppb. During that period the 8-h daytime average O_3 concentration reached 78.3 ppb, which exceeded the federal standard of 75 ppb. After six days of fires, O_3 diurnal concentrations returned to pre-fire patterns and levels.

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1. Introduction

Tropospheric ozone (O_3) is a naturally occurring greenhouse gas formed during photochemical reactions between nitrogen oxides (NO_x), carbon monoxide (CO) and volatile organic compounds (VOCs) (Finlayson-Pitts and Pitts, 2000). Since the Industrial Revolution, global background O_3 concentrations have been growing due to increasing emissions of O_3 precursors from fossil fuel combustion, industrial activities and wildland fires (Crutzen and Andreae, 1990; Sitch et al., 2007; Goldammer et al., 2009). At the end of the 19th century in Europe, and likely elsewhere in the Northern Hemisphere, O_3 levels were less than10 ppb (Guiherit and Roemer, 2000). At present, across that hemisphere, average annual O_3 concentrations are typically 40–50 ppb, with 50–60 ppb frequently occurring in mid-latitudes (Brasseur et al., 2001; Oltmans et al., 2006), including western regions of the United States (CASTNET, 2007; Jaffe and Ray, 2007). Since O_3 causes serious human health problems, it is listed as a federally and state-regulated air pollutant with a primary 8-h standard set at 75 ppb (http://www.epa.gov/air/ozonepollution/standards.html).

Emissions from combustion engines are a major contributor to O_3 precursors (Finlayson-Pitts and Pitts, 2000) and result in high O_3 concentrations in suburban, rural and remote areas downwind of the pollution source areas (Bytnerowicz et al., 2007). The highest O_3 concentrations occur in urban agglomerations with dense traffic, such as the Los Angeles Basin, where the term "photochemical smog" was coined in the 1950s as O_3 levels sometimes exceeded 500 ppb (Seinfeld and Pandis, 1998). Since the introduction of reformulated gasoline and the advent of strict controls on emissions ambient levels of O_3 in southern California rarely exceed 150 ppb (Bytnerowicz et al., 2008). In urban areas, O_3 concentrations have a clearly defined diurnal cycle with a minimum in early morning and a maximum in late afternoon. Such a pattern results from the daytime photochemical O_3 production combined with O_3 loss by dry deposition and reaction with nitric oxide (NO) after sunset when photochemical reactions stop. In locations where NO concentrations are high at night, such as urban areas or in the vicinity of major transportation routes, the nighttime drop of O_3 concentrations may be very pronounced, often resulting in a complete O_3 disappearance (Seinfeld and Pandis, 1998).

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Biomass burning results in elevated O_3 concentrations in adjacent downwind areas as photochemical reactions are fed by the NO_x, CO and VOC emissions. There are many examples of elevated concentrations related to wildland fires in various areas of the World (Goldammer et al., 2009). Production of O_3 from fires depends on the age of fire plume. Fresh plumes have thick smoke inhibiting photochemical reactions and contain high NO concentrations, effectively titrating O_3. Old plumes typically have less dense smoke allowing for photochemical reactions and O_3 generation from the abundant precursors such as NO_2, VOCs and CO (Urbanski et al., 2009). Intense wildfire periods can significantly increase surface O_3 levels in remote areas of the western United States (Jaffe et al., 2008). Increases of O_3 concentrations in areas downwind from the October 2007 southern California wildfires resulted in increased frequency of violations of the U.S. federal air quality standard for O_3 during that period of relatively low photochemical activity (Pfister et al., 2008).

In southern California, wildland fires commonly occur during Santa Ana wind conditions. Santa Ana winds are blustery, dry and warm blowing into southern California from the Mojave Desert. These winds develop when the desert is cold, and thus they take place during the cool season (October through March). When high pressure builds over the Great Basin in Nevada, the cold air begins to sink and is forced downslope, compresses and warms at a rate of $\sim 10\,^{\circ}C\,km^{-1}$. As the temperature rises, the relative humidity drops, the air picks up speed and is channeled through passes and canyons into southern California (http://www.atmos.ucla.edu/~fovell/ASother/mm5/SantaAna/winds.html).

Information gained from a detailed analysis of the ambient O_3 concentrations is important for understanding potential air quality problems caused by wildland fires. Such fires may become more intense and frequent in the western United States as the climate warms up (Westerling et al., 2006). Improved information on the effects of fires on air quality is crucial for air resources and land managers of the federal, state and local agencies deciding on a potential use of prescribed (controlled) fires as a tool for fuel reduction and mitigation of catastrophic fire effects. Prescribed burning has to be carefully planned and applied to assure compliance with the federal and state air quality standards (Arbaugh et al., 2009).

The objective of this study was to provide an analysis of ambient O_3 during the October 2007 southern California fires at a remote receptor site from a perspective of detailed changes of meteorological conditions. Such analysis was supported by characterization of ambient O_3 at several air quality monitoring sites in the general area of the fires. We were mostly interested if these fires produced violation of the national O_3 air pollution standard, and how the Santa Ana winds influenced the distribution of ambient O_3 in the area affected by the fires.

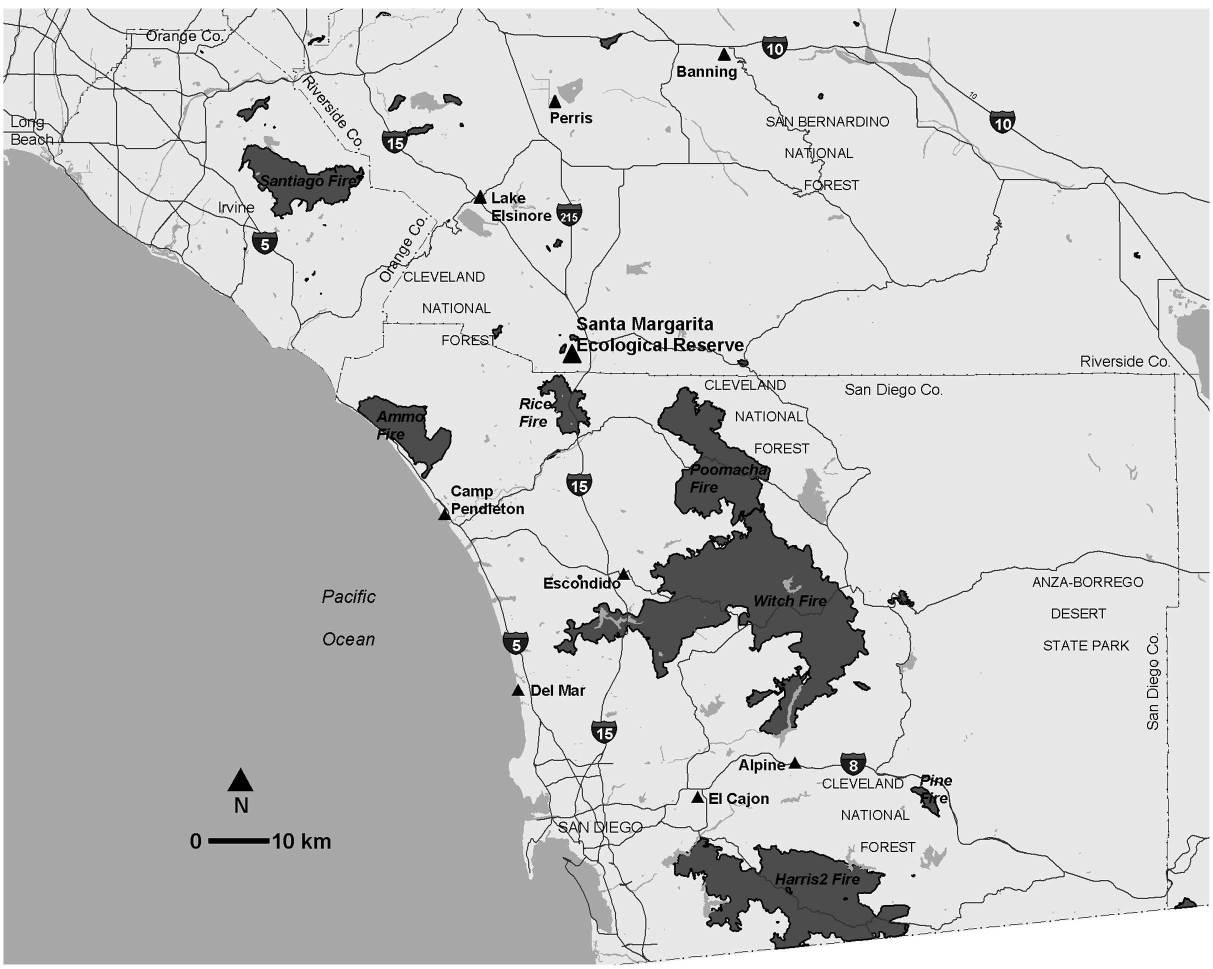

Fig. 1. October 2007 wildland fires and locations of the Santa Margarita Ecological Reserve (SMER) and other selected air quality monitoring stations in San Diego and Riverside counties.

Table 1
Southern California fires of October 2007.

Name	Start date	End date	Hectares	Distance from SMER (km)	Direction from SMER
Harris	October 21	October 31	36 600	80–100	SE
Santiago	October 21	November 8	11 490	45–63	NW
Poomacha	October 23	November 10	20 000	17–37	SE
Ammo	October 23	October 28	8500	22–35	W
Witch	October 21	October 31	80 125	37–67	S & SE
Rice	October 22	October 28	3830	2–11	S
Rosa	October 22	October 24	165	1	N
Pine	September 12	September 13	855	87–93	SE

2. Material and methods

2.1. Study location

The main receptor area for this study was the Santa Margarita Ecological Reserve (SMER). The SMER was established in 1962 and provides protected sites for southern California ecosystem research and education. The SMER lies on the Riverside/San Diego county line between the City of Temecula and the community of Fallbrook. The 1758 ha reserve encompasses an 8 km reach of the Santa Margarita River, the longest protected coastal river in southern California, and a variety of upland chaparral and agricultural habitats. The SMER is located to the west and in the vicinity of interstate freeway I15 which is the major north-to-south inland route between San Diego and Riverside counties with traffic of about 200 000 cars a day (http://webpawner.com/users/beachbuminda650/) (Fig. 1).

2.2. Meteorological measurements

An array of 20 meteorological stations has been installed in the SMER to sample a suite of weather variables, including precipitation, wind speed and direction, air temperature, relative humidity, barometric pressure, and solar radiation. The weather variables are sampled once per second and averaged each minute. The meteorological instruments are mounted on a 10 m Rohn tower. Winds are sampled at an elevation of 10 m, and temperature and humidity are sampled at an elevation of 2 m. Here we present weather observations from the SMER Devils Creek Overlook, which is located north of the Santa Margarita River and approximately midway between the communities of Fallbrook and Temecula at an elevation of ~350 m. The Devils Creek Overlook lies 4 km west of I15 and is ~400 m east-northeast of the North Station where air quality measurements were performed.

2.3. Ozone measurements

Ozone measurements have been performed at the SMER North Station since May 2005 with a 2B Technologies UV-absorption monitor Model 202 instrument (Bognar and Birks, 1996; http://www.twobtech.com/). The inlet to the instrument was mounted at an elevation of 2 m. Air entering the instrument was filtered through 5 μm pore Teflon filter. The instrument was calibrated by the manufacturer in May 2007 and its accuracy was ±1.5 ppb (http://www.twobtech.com/). Ozone results were presented as one-hour averages of twelve 5-min readings. Ozone concentrations for other discussed locations in the San Diego and Riverside counties were obtained from (http://www.arb.ca.gov/aqmis2/aqinfo.php), and for the Yellowstone, Great Basin, Grand Canyon and Joshua Tree National Parks (NP) from the CASTNET site (http://www.epa.gov/castnet/data.html).

2.4. Satellite images

The southern California images before and during the described fires were obtained from the Moderate Resolution Imaging Spectroradiometer (MODIS), which flies onboard NASA's Aqua and Terra satellites. The original image data were taken from the USA5 subset

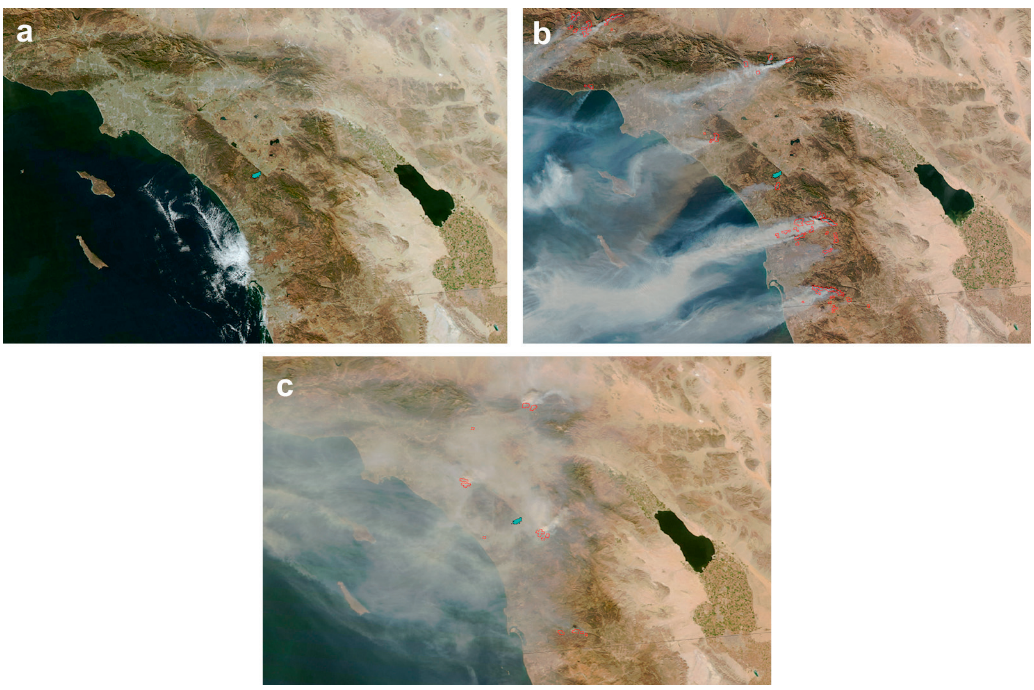

Fig. 2. MODIS images of smoke plumes during different phases of the southern California fires: (a) Phase I, normal, pre-fire conditions – October 20, 2007; (b) Phase II, Santa Ana winds, wildland fires and freeway closure – October 23, 2007; (c) Phase III, wildland fires after Santa Ana winds and freeway re-opening – October 25, 2007. The red boxes show the location of a thermal anomaly (fires) that was detected by MODIS using data from the middle infrared and thermal infrared bands. These boxes indicate the perimeter of 1 km resolution pixels containing the thermal anomaly detected by MODIS. Area of the SMER is shown as a cyan color polygon.

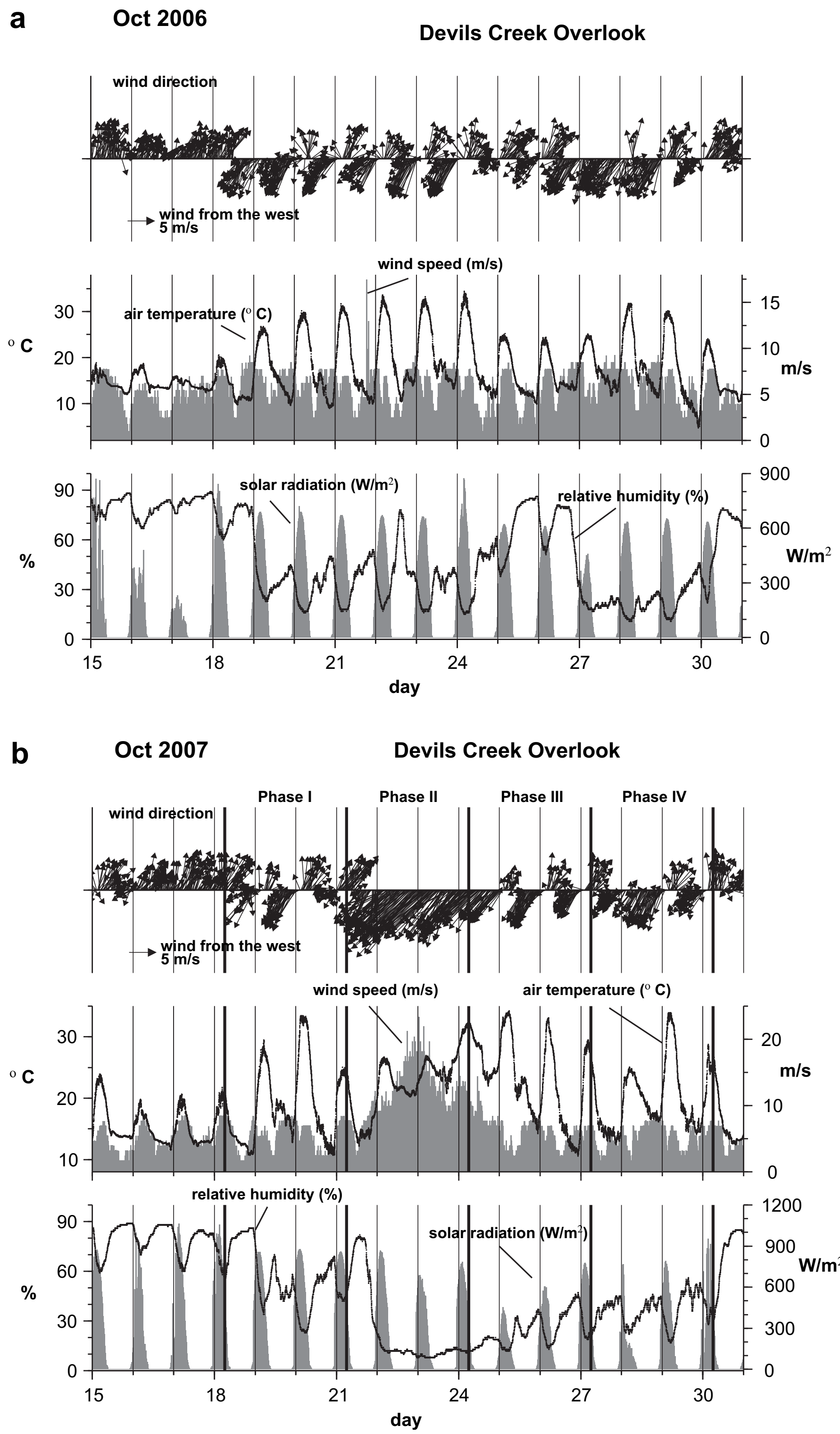

Fig. 3. Meteorological conditions, including vector wind [m s^{-1}] (upper panel), wind speed [m s^{-1}] and air temperature [°C] (middle panel), and relative humidity [%] and downward solar radiation [W m^{-2}] (lower panel) at the SMER for (a) October 2006 and (b) October 2007. Wind vectors point (arrows) in direction that wind is blowing, and the length of the arrow is proportional to the magnitude of the wind velocity. Values plotted are averages for every 1 min; wind vectors are plotted every 20 min. Horizontal axis marks days in Pacific Standard Time (PST).

of the USDA Foreign Agriculture Service (FAS) MODIS Rapid Response System Subsets collection and trimmed to show the 258 km (north-to-south) × 373 km (east-to-west) area centered on the reserve.

2.5. Back trajectories

The National Oceanic and Atmospheric Administration (NOAA) Air Resources Laboratory (ARL) provides an interactive Hybrid

Table 2
Summary of O_3 concentration changes at the Santa Margarita Ecological Reserve in various phases of the described event.

Phase	Period	Date (Oct-07)	O_3 avg. (ppb)	8 h O_3, avg. (ppb)	O_3 max. (ppb)	O_3 min. (ppb)
I. Normal pre-fire	1	18, day	29.4	39.1	47.7	4.6
	2	18–19, night	11.5		36.2	5.9
	3	19, day	28.0	36.7	47.1	7.2
	4	19/20 night	23.4		30.2	11.4
	5	20 day	34.2	42.8	50.2	5.4
	6	20/21 night	22.2		30.5	15.4
		Avg. (S.D.)	**24.8 (7.8)**			
II. Fires, Santa Ana winds, I15 closed	7	21 day	34.7	36.3	39.4	30.8
	8	21/22 night	39.0		40.0	37.7
	9	22 day	42.6	44.0	47.2	38.1
	10	22/23 night	40.5		46.6	35.8
	11	23 day	35.2	35.8	36.8	31.9
	12	23/24 night	28.8		36.6	24.0
		Avg. (S.D.)	**36.8 (5.0)**			
III. Fires, Santa Ana winds stopped, I15 open, smoke from the fires	13	24 day	39.3	46.8	55.4	21.9
	14	24/25 night	21.8		35.7	6.6
	15	25 day	33.4	49.3	67.4	0.6
	16	25/26 night	34.7		66.4	5.6
	17	26 day	55.4	**78.3**	**95.2**	4.1
	18	26/27 night	28.0		59.9	2.9
		Avg. (S.D.)	**35.4 (11.5)**			
IV. Normal, post-fire	19	27 day	32.6	41.0	46.4	2.5
	20	27/28 night	16.5		37.6	7.4
	21	28 day	36.2	45.9	53.9	9.6
	22	28/29 night	26.7		45.6	4.1
	23	29 day	26.1	37.5	47.2	0.9
	24	29/30 night	17.2		27.3	11.4
		Avg. (S.D.)	**25.9 (7.9)**			

Single-Particle Lagrangian Integrated Trajectory (HYSPLIT) model (http://www.arl.noaa.gov/HYSPLIT.php). For this study we employed the HYSPLIT model to compute air parcel trajectories for an intensive phase of the Santa Ana winds conditions. These trajectories used information from the EDAS 40 km dataset. Air parcel locations were entered and backward trajectories were started at 1000 m above ground level at 0000 UTC, October 23, 2007 (1600 PST, October 22, 2007) and were run back for 24 h until 1600 PST, October 21, 2007 (Draxler and Rolph, 2003; Rolph, 2003).

3. Results

A complex of wildland fires ignited in several locations in Southern California on October 21, 2007 and rapidly spread with strong, dry Santa Ana winds (http://www.oes.ca.gov/Operational/OESHome.nsf/ALL/876C5DEE11FE66808825737C005B8754). The largest of the fires – the Witch, Harris and Poomacha – were situated south and southeast from the SMER air quality monitoring site. The smaller Ammo fire was located west of the SMER. The Santiago Fire in Orange County was northwest of the SMER (Fig. 1, Table 1). Because of the fires, several roads and highways closed, some of them remaining closed until mid November, 2007. The I15 freeway closed on October 21 and reopened on October 24. MODIS images help with better visualization of wildfire smoke dispersion and show that compared to the pre-fire conditions when the air was clear (Fig. 2a), smoke from several Southern California fires was moved by strong Santa Ana winds in the southwestern direction into the Pacific Ocean, although it did not affect the SMER (Fig. 2b).

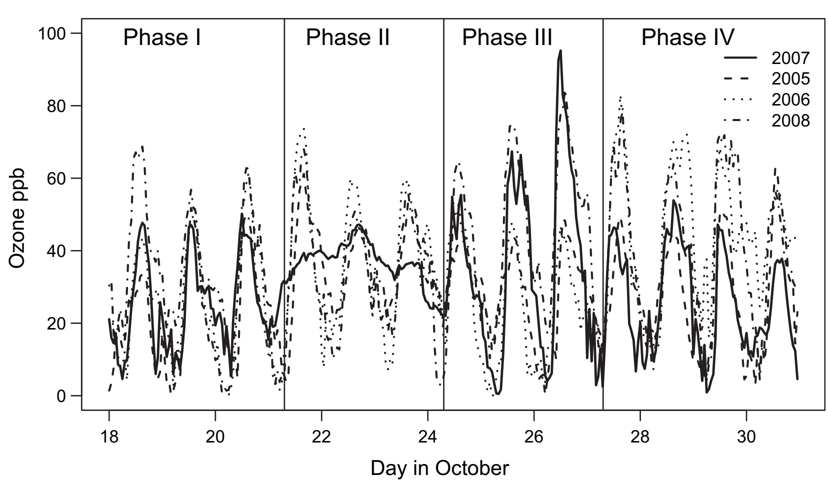

Fig. 4. Comparison of the October 2007 O_3 concentrations with those for the same month in 2005, 2006 and 2008 when fires were absent.

In the second phase of the fires, after the Santa Ana winds ceased, most of the southwestern California was covered with smoke, including the SMER (Fig. 2c).

Wind and other meteorological parameters at the SMER during the October 2007 Santa Ana-driven fires significantly differed from the background pre- and post-fire periods and from those in October 2006 (Fig. 3). Before the fires (Phase I), winds were mostly southwesterly and occasionally northeasterly. The early phase of the fires (Phase II), through October 24th, was marked by persistent northeasterly Santa Ana winds with average speed of 15 m s^{-1} and relative humidity which dropped markedly and held to very low levels near 10%. Thereafter, in the second period of the fires (Phase III), wind speeds dropped and the northeasterly winds continued, but were spelled during mid-day by onshore ventilation up the Santa Margarita River canyon from the southwest. During that phase, the afternoon relative humidity increased to ∼20–40% as an onshore flow developed. It is noteworthy that during the fires, and particularly during Phase III, solar insolation was reduced for several days despite the lack of large-scale cloudiness. This dimming effect was likely due to persistent smoke and aerosols in the lower atmosphere.

Diurnal patterns in measured O_3 concentrations at the SMER were substantially altered over the course of the fires (Table 2,

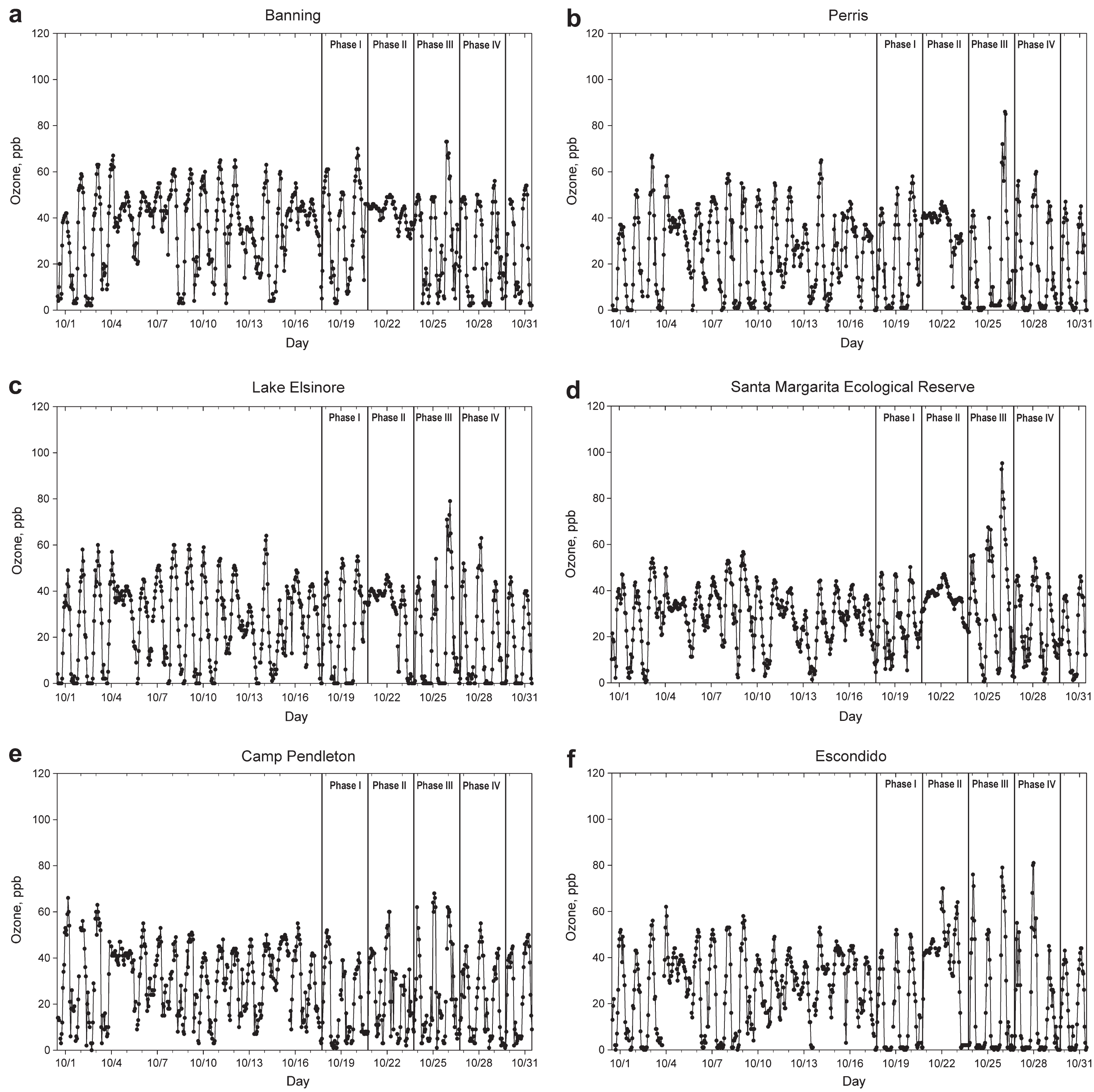

Fig. 5. October 2007 O_3 concentrations at SMER and selected San Diego and Riverside counties monitoring stations: (a) Banning; (b) Perris; (c) Lake Elsinore; (d) SMER; (e) Camp Pendleton; (f) Escondido; (g) Del Mar; (h) Alpine; (i) El Cajon.

Fig. 5d). The pre-fire Phase I had a clearly defined diurnal pattern with highest concentrations reaching 50 ppb in the late afternoon and lowest levels ~5–11 ppb just before dawn. During the initial stage of the fires (Phase II), when the Santa Ana winds were present, the daytime peaks were reduced to <47 ppb, the nighttime concentrations were elevated and remained >24 ppb, and the 3-day average was 36.8 ppb. During the second period of the intensive fires (Phase III), when the Santa Ana winds slackened, the smoke plume drifted over the SMER area and daytime O_3 concentration gradually increased. On the third day of Phase III (October 26), O_3 reached an hourly maximum of 95.2 ppb, and exceeded the federal 8-h standard of 75 ppb at 78.3 ppb. During that phase the nighttime values decreased to 3–7 ppb. After October 26, when the areal growth of the fires was much diminished and wind patterns returned to normal (Phase IV), diurnal changes of O_3 concentrations were similar to those of the pre-fire Phase I, with daytime maxima of 46–54 ppb, and nighttime values dropping to 4–11 ppb. While generally the diurnal characteristics of O_3 concentration in October 2007 were similar to those observed in 2005, 2006, and 2008, at the first period of the fires (Phase II) on October 21–24, 2007 during the Santa Ana winds, these patters were quite different (Fig. 4).

Ozone concentrations during the fires were also analyzed for the selected air quality monitoring stations in the San Diego and Riverside counties. At the Banning, Perris and Lake Elsinore sites (located 55 km northeast, 42 km north, and 32 km northwest of the SMER, respectively), the O_3 diurnal distribution patterns were similar (Fig. 5a–c) to those observed at the SMER site (Fig. 5d). At all these sites the O_3 concentrations stayed at ~40 ppb during the Santa Ana winds, and increased to ~75–85 ppb on October 26 after the Santa Ana winds stopped and wind direction changed to the southwest. At Camp Pendleton, located about 30 km southwest of the SMER site, no significant changes in O_3 concentrations were seen, although the October 21–27 values were slightly elevated compared with the preceding period (Fig. 5e). At Del Mar, just 10 km west of the Witch fire, highly elevated O_3 concentrations reached 110 ppb on the first day of the fires, October 21, and then gradually diminished to <60 ppb after October 26 (Fig. 5g). In Escondido, 35 km southeast of the SMER, O_3 concentrations reached daily maxima of >60 ppb on 5 of the 8 days after fires ignited; nighttime levels over that period were essentially 0 ppb with the exception of October 21 and 22, when 12 h average never dropped <32 ppb (Fig. 5f). Farther south, at El Cajon, elevated O_3 concentrations started on October 21 and lasted for about a week. At that location nighttime O_3 concentrations were near 0 ppb (Fig. 5i). The highest O_3 value of 108 ppb was recorded at Alpine on October 22, and elevated concentrations lasted until October 28. At that site the nighttime O_3 values were elevated during the fires and stayed >40 ppb (Fig. 5h). Among these sites, only at Alpine, on October 23, the federal 8-h standard was exceeded and reached 84.6 ppb.

Backward trajectories ending at the receptor sites at 1600 PST, October 22 (Fig. 6) show that over a period of 24 h air masses moved into southern California from central Idaho through the remote areas of Utah and Arizona (Great Basin, vicinity of the Rockies and Wasatch ranges, Mojave Desert). For the SMER, Banning, and Lake Elsinore (Fig. 6a) and Perris (Fig. 6b), air masses entered California near the Nevada & Arizona border and continued moving through the remote areas of the eastern Riverside and San Bernardino counties. For the Camp Pendleton, Del Mar and Escondido sites (Fig. 6c), and Alpine and El Cajon (Fig. 6d) the back trajectories were generally similar to the previous sites, however, were positioned more to the south when air masses entered into southern California across the Arizona border. Average 24-h O_3 concentrations near those trajectories were during that period: Yellowstone NP – 33. 5 ppb; Great Basin NP – 36.6 ppb;

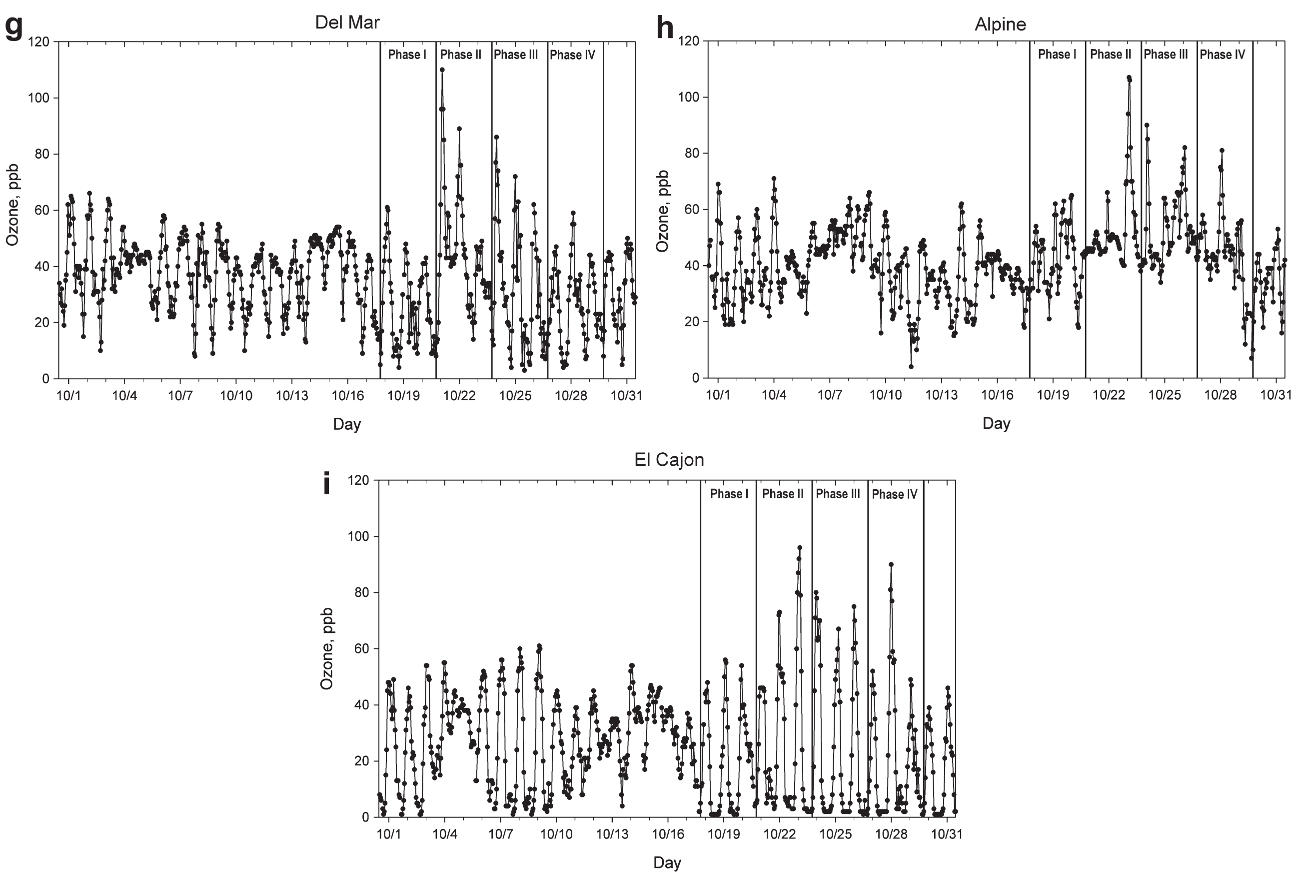

Fig. 5. (*continued*).

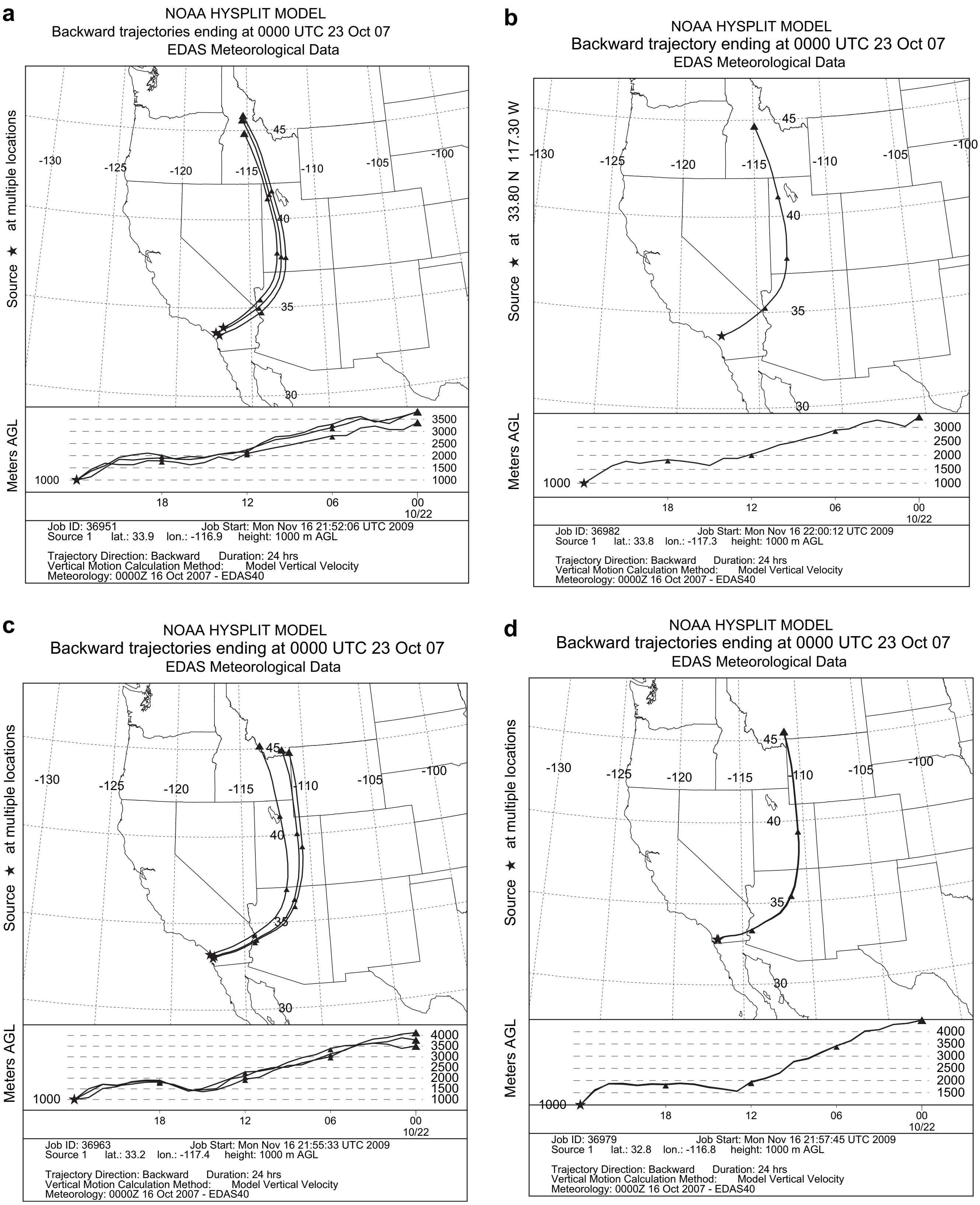

Fig. 6. Backward trajectories for the selected San Diego and Riverside counties monitoring stations during the fires and Santa Ana winds ending at 1600 PST, October 22, 2007 (0000 UTC, October 23, 2007) and run back until 1600 PST, October 21, 2007: (a) Banning, Lake Elsinore and SMER; (b) Perris; (c) Camp Pendleton, Del Mar and Escondido; (d) Alpine and El Cajon.

Grand Canyon NP– 44.0 ppb, and Joshua Tree NP – 38.8 ppb (http://www.epa.gov/castnet/data.html). Thus the concentrations at the two latter sites were very similar to those recorded during that time in Banning (44.5 ppb), SMER (40.1 ppb), Lake Elsinore (40.3 ppb) and Perris (41.3 ppb).

4. Discussion

Santa Ana winds, which commonly occur in southern California in autumn and winter (Carle, 2006; http://www.atmos.ucla.edu/~fovell/ASother/mm5/SantaAna/winds.html), set in on October 21 and lasted until October 24, 2007. The winds caused a rapid spread of some of the most devastating fires in the recent history of the United States and caused 9 deaths, 85 injuries, >1500 houses burned, and evacuation of ~1 000 000 people (http://en.wikipedia.org/wiki/California_wildfires_of_October_2007). While effects of the fires on air quality were reported, those were mostly regarding increased concentrations of particulate matter reaching unhealthy levels at various monitoring stations of the San Diego Air Pollution Control District (SDAPCD). The highest value for PM_{10} reached 450 $\mu g\ m^{-3}$ on October 22 and the 24-h $PM_{2.5}$ was >125 $\mu g\ m^{-3}$ on October 22–23 in Escondido, significantly exceeding the federal standard of 35 $\mu g\ m^{-3}$ (http://www.epa.gov/particles/standards.html). After October 27, the PM_{10} concentrations dropped to <100 $\mu g\ m^{-3}$ at all SDAPCD monitoring locations (http://www.sdapcd.org/air/reports/smog.pdf).

The SMER pre-fire O_3 concentrations had a clearly defined diurnal pattern that also occurred in other analyzed sites of this study. Such patterns are typical for the suburban areas of southern California where maximum concentrations in autumn are much lower than in summer and where NO emitted from nighttime traffic reduces O_3 concentrations (Seinfeld and Pandis, 1998). During the first phase of the fires, O_3 concentrations at the SMER site stayed relatively stable with modest diurnal changes. Similar low levels and diurnal patterns were also seen in the upwind sites of Banning, Perris and Lake Elsinore as the result of strong Santa Ana winds moving clean air with low, background continental O_3 concentrations into southern California. All these sites showed O_3 concentrations similar to such upwind distant sites as the Grand Canyon NP and Joshua Tree NP.

During the second phase of the fires, the daytime O_3 concentrations at the SMER site gradually increased. Several factors may have contributed to that phenomenon: slackening of the Santa Ana winds; return of the southerly and southeasterly daytime winds moving O_3-enriched air masses and O_3 precursors from large fires near San Diego and the reopened I15 freeway and other highways; daytime northwesterly and northerly winds bringing similarly polluted air masses from the Santiago Fire in Orange County. These increases of O_3 concentrations were taking place despite the diminished solar radiation and lower potential for photochemical reactions caused by smoke aerosols emitted from regional fires. The nighttime O_3 concentrations again were dropping to ~0 ppb because there was enough NO from traffic emissions and fires for O_3 titration. After the fires typical O_3 patterns for this area and time of the year returned. Comparison of the October 2007 data with those of 2005, 2006, and 2008 at the SMER site showed that in general the O_3 diurnal patterns were similar for all these years, only with those during the October 21–24, 2007 period being exceptional due to the combination of the Santa Ana winds and fire effects. Similar O_3 distribution patterns were also seen in other sites affected by the fires. However, the Alpine site differed from all other sites – it is further away from the highly populated areas and therefore its O_3 diurnal changes are characteristic of the location where there is not enough NO for the nighttime O_3 titration.

Our results indicate that the October 2007 fire events caused a significant, although only short-lasting, increase of O_3 concentrations at the SMER receptor site. The maximum value of 95 ppb measured on October 26 was about 40–45 ppb higher than the highest values measured before and after the fire. On the same day the 8-h average O_3 concentration reached 78.3 ppb and thereby exceeded the federal air quality standard of 75 ppb. During the fires, ambient O_3 at the SMER and other monitoring sites were elevated and highly variable, but the federal 8-h standard for O_3 was exceeded only on single days at the SMER and Alpine sites. Pfister et al. (2008) reported that the September and October 2007 California wildland fires significantly increased O_3 ambient concentrations and frequency of exceedances of the 75 ppb 8-h federal standard in rural areas downwind of the fires even though the fires occurred during autumn, which is a period of low photochemical activity. All these finding should be carefully evaluated from a perspective of potential effects of prescribed burning as a possible management tool for fuel reduction. Deterioration of air quality and exceedance of federal standards for criteria pollutants are the key issues that land and air resource managers have to consider before using prescribed fires. However, it can be expected that spatially limited and less intense prescribed fires, when carefully applied during periods of low photochemical activity (spring, autumn or winter), should not cause major problems from the perspective of the O_3 air quality standard. It should be emphasized, however, that the 2007 wildfires caused much more serious violations of the particulate matter standards (PM_{10} and $PM_{2.5}$) (http://www.sdapcd.org/air/reports/smog.pdf) and thus posed a threat to human health. Consequently, compliance with these standards may be a more serious issue than potential exceedances of the O_3 standard. The issue of compliance with air quality standards during prescribed burning is critical because it could restrict that practice enough to limit its potential to mitigate potential impacts of catastrophic wildfires.

Continuous air quality observations, such as the described ones, can provide crucial information needed for a better understanding of potential implications of wildland fire emissions on human health and ecosystem responses. Detailed analyses of the effects of fires on temporal changes of O_3 concentrations at the receptor sites that take into account changing meteorological conditions are essential for understanding potential changes in air quality caused by fires and for development of air pollution dispersion models of improved accuracy.

Long-term monitoring of the criteria pollutants (such as O_3, NO_2 or PM) as well as of the pollutants with strong potential for ecological effects (NH_3 and HNO_3) should be developed for remote areas. Such efforts can help in understanding background patterns of these pollutants and their changes in the presence of wildfires or various anthropogenic activities. This may be especially important for national parks, nature reserves or other areas of high ecological value.

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