Analysis of the effects of combustion emissions, freeway closure and Santa Ana winds on air quality during the October 2007 southern California wildfires

Andrzej Bytnerowicz[1], Dan Cayan[2], Phil Riggan[1], Susan Schilling[1], Philip Dawson[3], Diane Alexander[1], Mary Tyree[2], Lynn Wolden[1] and Robert Tissell[1]

[1]US Forest Service, Pacific Southwest Research Station, 4955 Canyon Crest Drive, Riverside, CA 92507
[2]Climate Research Division, Scripps Institution of Oceanography, University of California, San Diego, 9500 Gilman Drive, La Jolla, CA 92093
[2]30358 Stephanie Street, Hemet, CA 92543

Abstract

Combustion emissions, closure of a major interstate freeway and other local transportation routes, as well as the 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 for a rural receptor site, the Santa Margarita Ecological Reserve, located among several large fires in San Diego and Orange counties. During the initial phase of the fires (strong, dry northerly and northeasterly Santa Ana winds and greatly reduced traffic), daytime O_3 concentrations were reduced, and nighttime O_3 levels were abnormally elevated. During the second phase of the fires (no Santa Ana winds present, resumed traffic, and air filled with smoke), daytime O_3 concentrations steadily increased and reached 95.2 ppb. During that period the 8-h daytime average O_3 concentration reached 78.3 ppb, which exceeded the federal standard of 75 ppb. Nighttime levels returned to ~0 ppb during this episode. After six days of the fires, O_3 diurnal concentrations returned to the pre-fire patterns and levels. Average 2-week concentrations of nitrogen dioxide (NO_2), ammonia (NH_3) and nitric acid (HNO_3) during the fires were elevated compared to the pre-fire conditions.

Key Words: air pollution, ozone, nitrogen dioxide, ammonia, nitric acid, meteorology.

Introduction

Tropospheric ozone (O_3) is a naturally occurring greenhouse gas formed during photochemical reactions between nitrogen oxides (NOx), methane (CH_4), 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). Western regions of the United States are subject to those levels (CASTNET, 2007).

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 down-wind of the pollution source areas (Bytnerowicz et al., 2008). 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 aimed at reduction of O_3 formation, 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).

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). Ozone concentrations greater than 40 ppb are considered potentially phytotoxic (Fowler et al, 1999). Ozone injury to plants depends on their exposure dose, peak O_3 concentrations, various environmental factors, as well as the gas exchange and metabolic defense capacity of plants (Wellburn, 1988).

Biomass burning results in elevated O_3 concentrations in adjacent downwind areas as photochemical reactions are fed by the NOx, CO and VOCs emissions. There are many examples of elevated concentrations related to wildland fires in many parts of the World (Goldammer et al., 2009). Increases of O_3 concentrations in areas downwind from the October 2007 southern California wildfires demonstrate that intense wildfire periods can significantly increase frequency of violations of the U.S. federal air quality standard for O_3 even during the photochemically less active seasons (Pfister et al., 2008).

Elevated concentrations of nitrogen dioxide (NO_2), nitric acid (HNO_3) and ammonia (NH_3) have been reported in the vicinity of wildland fires (Goldammer et al., 2009). While NO_2 is a criteria pollutant regulated from a point of view of human health effects (http://www.epa.gov/ttnnaaqs/standards/nox/s_nox_index.html), HNO_3 and NH_3 are not regulated. However, due to their high deposition velocity and reactivity, these pollutants are the major drivers of nitrogen (N) dry deposition. N deposition is elevated in areas of semi-arid southern California, primarily those subject to exposures to high concentrations of HNO_3 and NH_3 resulting from traffic, agricultural and industrial activities, and waste management. Heightened N levels may effect changes in plant community species composition, increased biomass production and subsequent fire intensity, or contamination of water (Fenn et al., 2003).

Information gained from a detailed analysis of the wildland fire effects on ambient O_3 and N pollutants concentrations is important for understanding potential air quality problems from a perspective of the predicted more frequent and intense wildfires in the southwestern United States as the climate warms up (Westerling et al., 2006). Such information 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).

Material and Methods

The Santa Margarita Ecological Reserve (SMER), established in 1962, 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 4344-acre reserve encompasses a 5-mile reach of the Santa Margarita River, the longest protected coastal river in southern California, and a variety of upland chaparral and agricultural habitats.

Interstate freeway I15 is the major north-to-south inland route between San Diego and Riverside County with 241,000 cars a day counted at the Rancho Bernardo Center Drive intersection and 195,000 cars at the intersection with California state highway 78 in Escondido (http://webpawner.com/users/beachbuminda650/). The SMER is located to the west and in the vicinity of I15 (Figure 1).

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-meter 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 approximate elevation of 350 m. The Devils Creek Overlook lies 4 km west of I15 and is within 400 m of the North Station where air quality measurements were performed.

Air Quality measurements

Air quality measurements have been performed at the SMER North Station site since May 2005. Ozone concentrations were measured with a 2B Technologies UV-absorption monitor (Bognar and Birks, 1996), and the results were presented as one-hour averages. Ogawa passive samplers (Pompano Beach, FL) were used to measure concentrations of ammonia (NH_3) and nitrogen dioxide (NO_2). Ammonia was absorbed on two replicate cellulose pads coated with citric acid forming ammonium citrate. Ammonium concentrations in filter extracts were determined colorimetrically on a Technicon Autoanalyzer, and ambient NH_3 concentrations were calculated based on a comparison of passive samplers with collocated annular denuder systems (Koutrakis et al., 1993). Nitrogen dioxide was collected on two replicate cellulose filters coated with triethylene amine (TEA), nitrite concentrations were determined with ion chromatography (Dionex, Model 4000i), and NO_2 concentrations were calculated based on a comparison with the collocated NO_2 active monitors (Bytnerowicz et al., 2007). Passive samplers developed by the US Forest Service were used for monitoring nitric acid (HNO_3) concentrations (Bytnerowicz et al., 2005). In that sampler, ambient air passes through a Teflon membrane and gaseous HNO_3 is absorbed on a Nylasorb nylon filter as NO_3^-. Nitrate concentrations in sample extracts were analyzed by ion chromatography, and concentrations of HNO_3 were calculated using the calibration curves. Three replicate HNO_3 samplers were used at each monitoring site. Since May 2005, passive samplers were exposed for 2-week long periods, then collected and replaced. The results from a set of measurements taken during October 2006 and October 2007 are discussed in this paper.

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 from the USDA Foreign Agriculture Service (FAS) subset in the MODIS Rapid Response System Subsets collection and trimmed to show ~250 km area centered on the reserve.

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. Because of the fires, several roads and highways closed, some of them remaining closed until November, 2007. The I15 freeway closed for three days with traffic resuming on October 24. 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 (Figure 1). The perimeters of fires are presented on Figure 1 and their characteristics are summarized in Table 1. MODIS images help with better visualization of wildfire smoke dispersion. Compared to the pre-fire conditions when the air was clear (Figure 2 a), smoke from several Southern California fires was moved by strong Santa Ana winds in the southwestern direction into the Pacific Ocean, however, it did not affect the SMER (Figure 2 b). 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 (Figure 2 c).

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 (Figure 3). Before the fires, winds were mostly southwesterly and occasionally northwesterly. The early phase of the fires, through October 24th, was marked by persistent northeasterly winds with average speed of 15 m/s and relative humidity which dropped markedly and held to very low levels near 10%. Thereafter, in the second phase of the fires, 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 afternoon relative humidity increased to ~20 - 40% as an onshore flow developed. It is noteworthy that during the fires, and particularly during this latter period, 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 2007 southern California fires (Table 2, Figure 4). The “pre-fire” phase 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, when the Santa Ana winds were present and freeways closed, the daytime peaks were reduced to values less than 47 ppb and the nighttime concentrations were elevated and remained greater than 24 ppb. During the second phase of the intensive fires, when the Santa Ana winds slackened, the smoke plume drifted over the SMER area, and after freeway I15 was re-opened, daytime O_3 concentration gradually increased. On the third day of this second phase (October 26) O_3 reached an hourly maximum of 95.2 ppb, and exceeded the federal 8-hour standard at 78.3 ppb. During that stage 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, diurnal changes of O_3 concentrations were similar to those of the pre-fire phase, with daytime maxima of 46 - 54 ppb, and nighttime values dropping to 4 – 11 ppb. In general, the October 2007 pattern of O_3 concentration and its diurnal characteristics at the SMER, with the exception of the six days of most rapid fire spread, were very similar to those observed in October 2006 (data not shown).

Ozone concentrations during the fires were also monitored at other air quality monitoring stations in San Diego County. 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 (Figure 5 a). 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 less than 60 ppb after October 26 (Figure 5 c). In Escondido, 35 km southeast of the SMER, O_3 concentrations reached daily maxima greater than 60 ppb on 5 of 8 days after fires ignited; nighttime levels over that period were essentially 0 ppb with the exception of October 22. 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 apparently due to reaction of O_3 with NO emitted by the nighttime traffic (Figure 5 e). The highest among all of the San Diego County measurements 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

(Figure 5 f). The Alpine site is further away from the highly populated areas and therefore its O_3 diurnal patterns are characteristic of the location where there is not enough NO for the O_3 titration. Among these 5 sites, only at Alpine, on October 23, the federal 8-hour standard was exceeded and reached 84.6 ppb.

Two-week long average concentrations of ammonia (NH_3), nitric acid (HNO_3) and nitrogen dioxide (NO_2) were about 2-fold higher during the 2-week long period encompassing the fires compared to the pre-fire period. After the fires, concentrations of NH_3 and HNO_3 were lower than during the fires, however much higher than before the fires. After the fires, concentrations of NO_2 additionally increased (Figure 5).

Discussion

Santa Ana winds, which commonly occur in southern California in fall and winter (Carle, 2006), 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, >1,500 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 500 $\mu g\ m^{-3}$ on October 22 and the 24-hour $PM_{2.5}$ was >125 $\mu g\ m^{-3}$ on October 22-23 in Escondido. After October 27, the PM_{10} concentrations dropped <100 $\mu g\ m^{-3}$ at all SDAPCD monitoring locations (http://www.sdapcd.org/air/reports/smog.pdf).

Effects of forest fires on air quality, including increased O_3 concentrations downwind from the fires caused by the emissions of its precursors (NOx, VOCs and CO) and photochemical reactions, are well-known (Goldammer et al., 2009). However, additional effects on O_3 diurnal patterns have not previously been well described, such as greatly reduced vehicle combustion emissions due to closures of major highways and changing wind patterns aided by the detailed analysis of meteorological conditions during the fires. The presented analysis of the southern California fires from a perspective of changes of ambient O_3 as well as of N air pollutants could lead to a better understanding of the effects of wildland fires on air quality and to develop air pollution dispersion models of improved accuracy.

Pre-fire O_3 concentrations followed a clearly defined diurnal pattern with peak concentrations in late afternoon of ~50 ppb, and daily minima occurring just before dawn. The pattern was typical for suburban areas of southern California where maximum concentrations in autumn are much lower than in summer and where NO emitted from nighttime traffic reacts with O_3 to reduce its concentrations (Seinfeld and Pandis, 1998). During the first phase of the fires, a combination of various factors affected the observed O_3 changes. Although solar radiation was similar to the pre-fire phase, maximum O_3 concentrations were reduced mainly because the air masses brought by northerly and northeasterly Santa Ana winds neither O_3-rich nor O_3 precursor-rich as they traversed the

Mojave Desert and passed over relatively sparsely populated rural and suburban areas of Riverside County. During that period, the nighttime O_3 remained elevated, because the closed I15 freeway and many other local highways did not supply sufficient amounts of NO for O_3 titration. Such a scenario is similar to that in remote mountain locations where O_3 remains high at night because NO oxidation to NO_2 reduces concentrations of NO to low levels during air mass transport (Burley and Ray, 2006; Bytnerowicz et al., 2008). During the second phase of the observed fires, daytime O_3 concentrations gradually increased. Several factors may have contributed to that phenomenon: slackening of the Santa Ana winds that had brought relatively clean air masses from the north and northeast; 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 from the southern portion of the re-opened I15 freeway; daytime northwesterly and northerly winds bringing similarly polluted air masses from the Santiago Fire in Orange County and from the traffic emissions on more northern portions of the I 5 freeway; and emissions from other re-opened local highways. 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 for O_3 titration. Our results show that these 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-hour average O_3 concentration reached 78.3 ppb and thereby exceeded the federal air quality standard of 75 ppb. Ambient O_3 at the SMER and other monitoring sites were elevated and highly variable. Comparison with other sites indicates that the SMER site was unique in regard to changes in diurnal patterns caused by the combined effects of fires, Santa Ana winds and traffic closures. We note that during the fires the federal 8-hour standard for O_3 was exceeded only 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-hour federal standard in rural areas downwind of the fires even though the fires occurred during fall, which is a period of low photochemical activity. Our results confirmed that the O_3 federal standard was exceeded in the vicinity of the fires, but only over a single day. After several days of burning in the area, the O_3 concentrations in the SMER receptor site and other sites returned to normal. This 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 key issues that land and air resource managers have to consider before using prescribed fires. Despite the large size of the southern California fires, the federal O_3 standard was exceeded only on single days at two sites. Therefore it can be expected that spatially limited and less intense prescribed fires, when carefully applied during periods of low photochemical activity (spring, fall or winter), should not cause major problems from a perspective of the O_3 air quality standard. However, 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 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 since it could so restrict that practice as to cripple its potential to mitigate subsequent impacts of catastrophic wildfires.

From the perspective of potential effects on vegetation, no serious damage could be expected since coastal sage and chaparral communities are relatively tolerant of O_3 exposures (Stolte et al., 1982), especially during autumn when plants have low physiological activity (Grulke, 2009).

Concentrations of NH_3, NO_2 and HNO_3 measured during the 2-week long period including the fires (October 18 – November 1) were higher than in the pre-fire period, consistent with other observations (Goldammer et al., 2009). Equipment designed for shorter-term measurements, such as real-time instruments for NO_2 monitoring, or denuder systems for NH_3 and HNO_3 (Koutrakis et al., 1993), could provide a better understanding of the effects of fires on these pollutants. However, such equipment is expensive and requires air conditioning (NO_2 monitors), or is labor-intensive (denuders), and as such is not practical for routine air quality monitoring. Observed NH_3 and HNO_3 concentrations during the 2007 fires were well above their background levels (Bytnerowicz and Fenn, 1996), although similar to the October 2006 values when no fires were present in the vicinity of the SMER (data not shown). Concentrations determined during the fire period were not high enough to cause any phytotoxic effects, but might lead to elevated deposition of N to ecosystems with possible ecological implications (Fenn et al., 2003).

Wildland fires, Santa Ana winds and freeway closures all had pronounced impact on concentrations and the diurnal characteristics of O_3 at the remote rural receptor site. The O_3 diurnal changes observed at this remote site were unique compared with those from other air pollution monitoring sites in the affected area. Increased levels of NO_2, HNO_3 and NH_3 during the fires were also observed at the studied site. Continuous air quality observations such as these, at a modest network of stations can provide crucial information needed for a better understanding of potential implications of wildland fire emissions on human health and ecosystem responses. Long-term monitoring of the criteria pollutants (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.

Acknowledgements

We thank management of the Santa Margarita Ecological Reserve staff, including Mark Van Scoy and Pablo Bryant for assistance during air quality monitoring. Spencer Kawamoto and Douglas Alden at Scripps Institution of Oceanography, U. C. San Diego, helped with meteorological observations and processing.

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Captions

Figure 1. October 2007 wildland fires and locations of the Santa Margarita Ecological Reserve (SMER) and selected air quality monitoring stations in San Diego County: (a) Camp Pendleton; (b) SMER; (c) Del Mar; (d) Escondido; (e) El Cajon; (f) Alpine.

Figure 2. MODIS images of smoke plumes during four stages of the southern California fires; (a) normal, pre-fire conditions - October 20, 2007; (b) Santa Ana winds, wildland fires and freeway closure – October 23, 2007; (c) 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 1km-resolution pixels containing the thermal anomaly detected by MODIS. Area of the SMER is shown as a cyan color polygon.

Figure 3. Meteorological conditions, including vector wind [m/s] (upper panel), wind speed [m/s] and air temperature [°C] (middle panel), and relative humidity [%] and downward solar radiation [W/m2] (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 one minute; wind vectors are plotted every 20 minutes. Horizontal axis marks days in GMT.

Figure 4. Changes of O_3 concentrations at SMER during: (a) normal, pre-fire; (b) Santa Ana winds, wildland fires and freeway closure; (c) wildland fires after Santa Ana winds and freeway re-opening; (d) normal, post-fire.

Figure 5. October 2007 O_3 concentrations at SMER and selected San Diego County monitoring stations: (a) Camp Pendleton; (b) SMER; (c) Del Mar; (d) Escondido; (e) El Cajon; (f) Alpine.

Figure 6. Concentrations of NH_3, HNO_3 and NO_2 at SMER before, during and after fires.

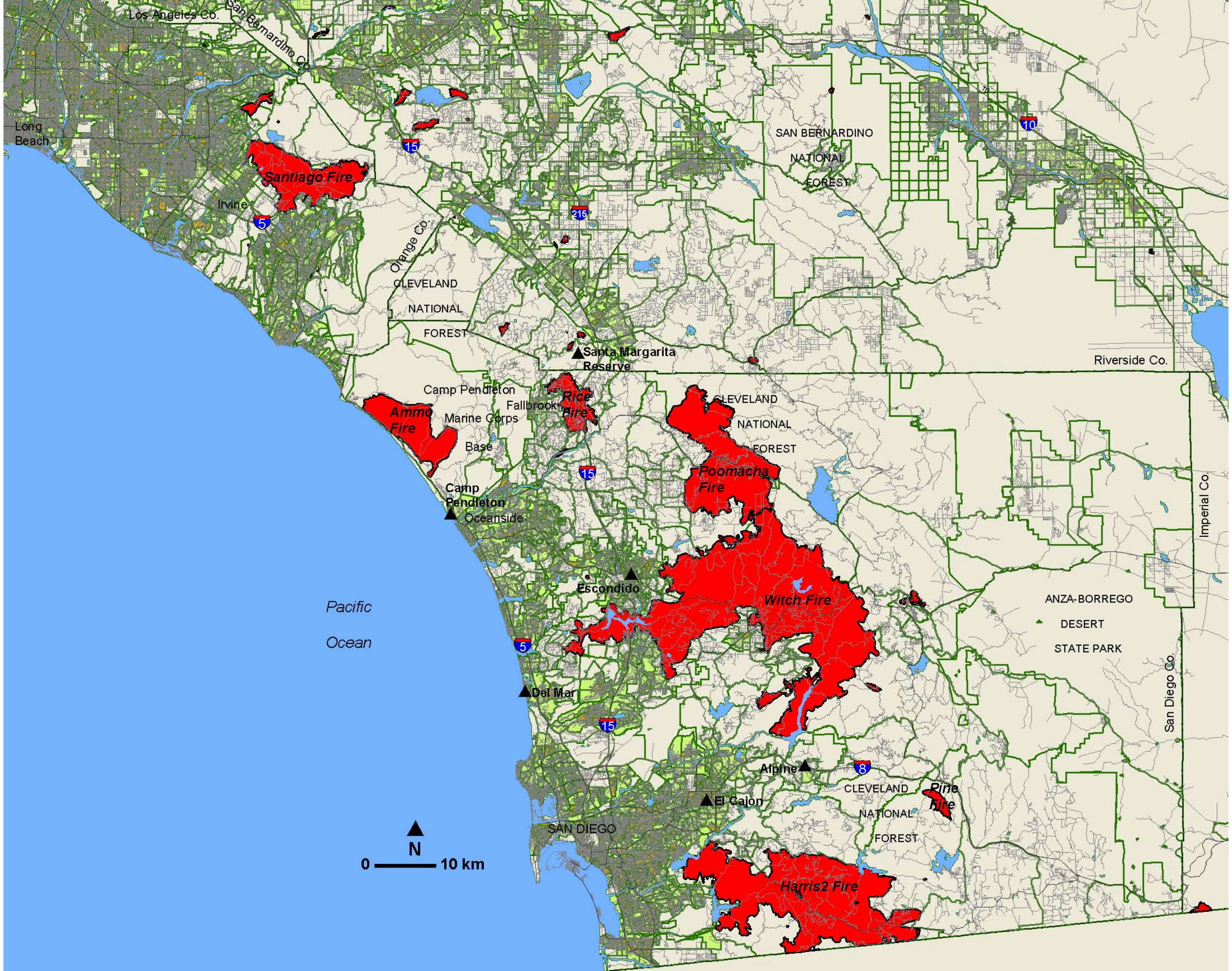
Table 1. Southern California fires of October 2007.

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	8,500	22-35	W
Witch	October 21	October 31	80,125	37-67	S & SE
Rice	October 22	October 28	3,830	2-11	S
Rosa	October 22	October 24	165	1	N
Pine	September 12	September 13	855	87-93	SE

Table 2. Summary of O_3 concentration changes in various phases of the described event.

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

***Remark - dominant winds in bold**


Los Angeles Co.
San Bernardino Co.
Long Beach
Santiago Fire
Irvine
Orange Co.
CLEVELAND
NATIONAL
FOREST
SAN BERNARDINO
NATIONAL
FOREST
Riverside Co.
Santa Margarita Reserve
Camp Pendleton
Fallbrook
Marine Corps
Base
Ammo Fire
Rice Fire
CLEVELAND
NATIONAL
FOREST
Poomacha Fire
Camp Pendleton
Oceanside
Imperial Co.
Escondido
Witch Fire
ANZA-BORREGO
DESERT
STATE PARK
Pacific
Ocean
Del Mar
San Diego Co.
Alpine
El Cajon
CLEVELAND
NATIONAL
FOREST
Pine Fire
SAN DIEGO
N
0
10 km
Harris2 Fire
5
15
215
10
8

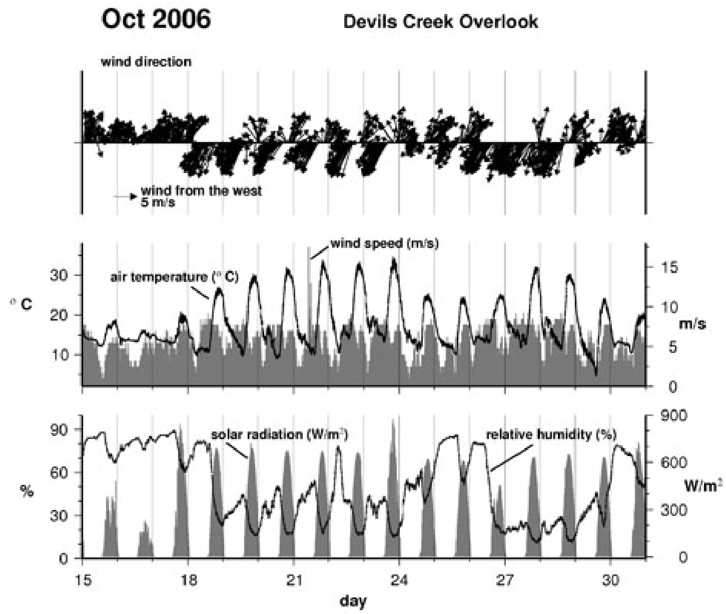


Figure 3 a.

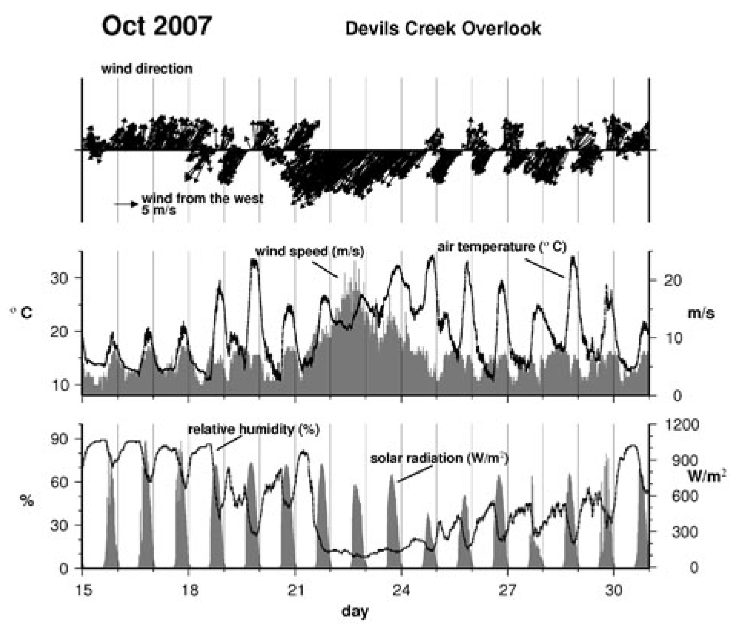


Figure 3 b.

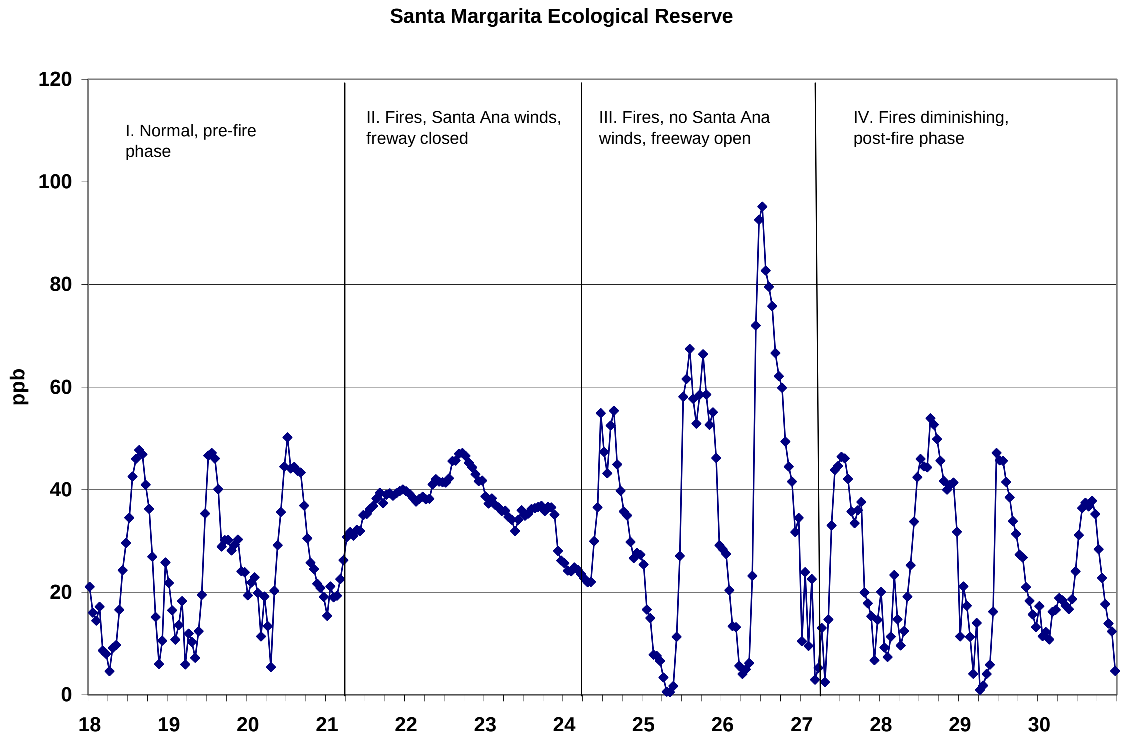

Santa Margarita Ecological Reserve
120
100
80
60
40
20
0
ppb
18
19
20
21
22
23
24
25
26
27
28
29
30
I. Normal, pre-fire phase
II. Fires, Santa Ana winds, freway closed
III. Fires, no Santa Ana winds, freeway open
IV. Fires diminishing, post-fire phase

Figure 4.

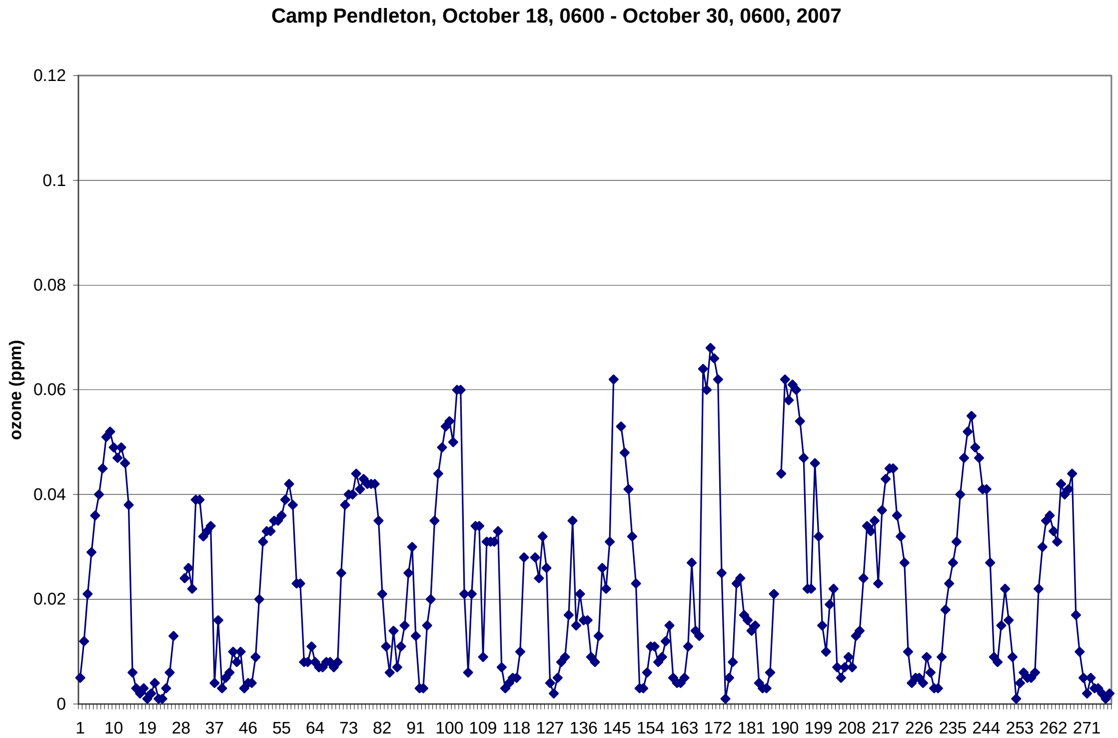

Camp Pendleton, October 18, 0600 - October 30, 0600, 2007
ozone (ppm)
0.12
0.1
0.08
0.06
0.04
0.02
0
1 10 19 28 37 46 55 64 73 82 91 100 109 118 127 136 145 154 163 172 181 190 199 208 217 226 235 244 253 262 271

Figure 5 a.

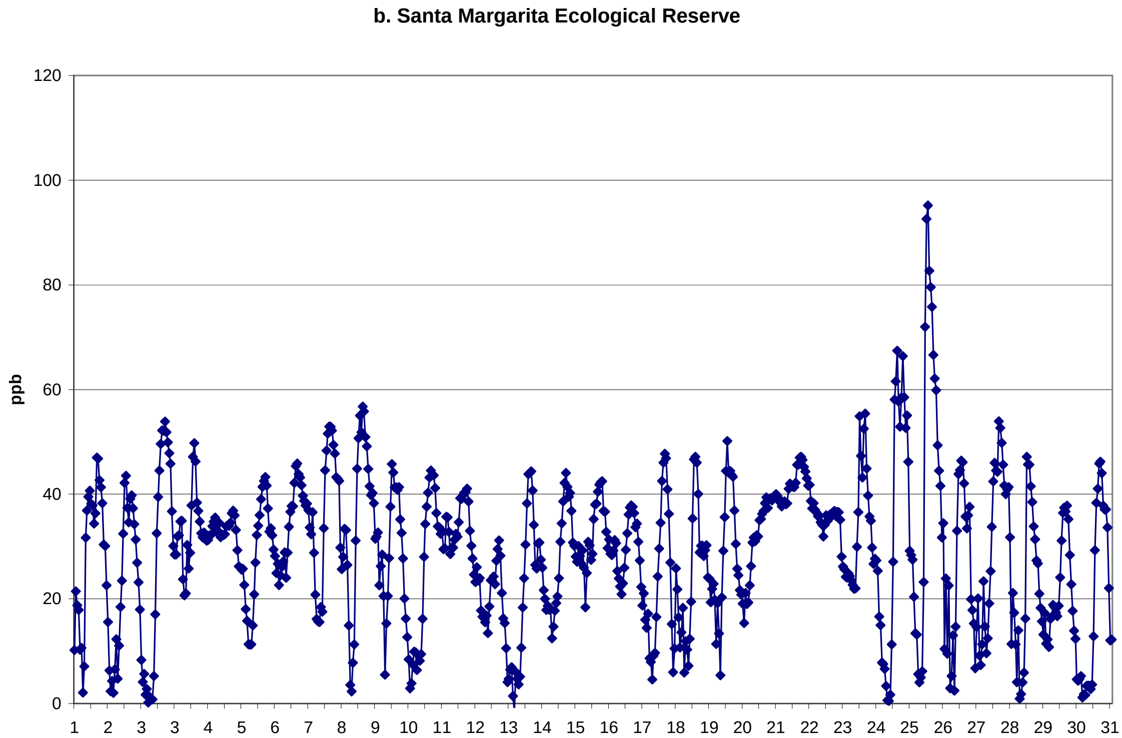

b. Santa Margarita Ecological Reserve
ppb
120
100
80
60
40
20
0
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 26 27 28 29 30 31

Figure 5 b.

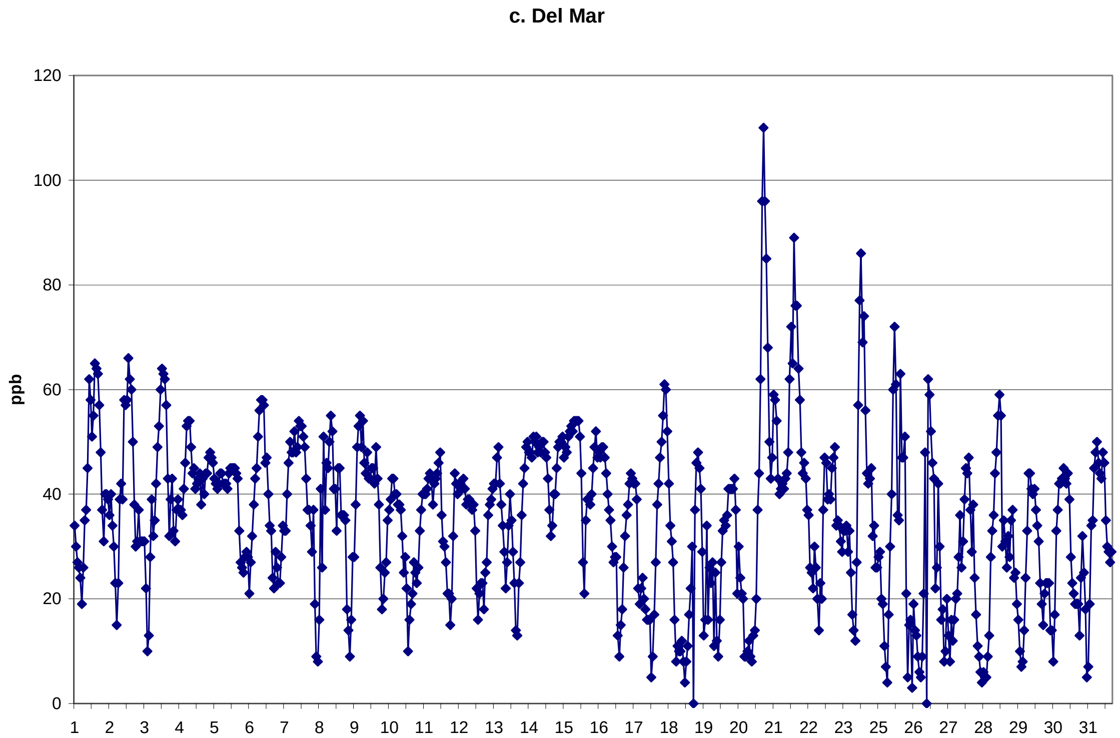

c. Del Mar
ppb
120
100
80
60
40
20
0
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 25 26 27 28 29 30 31

Figure 5 c.

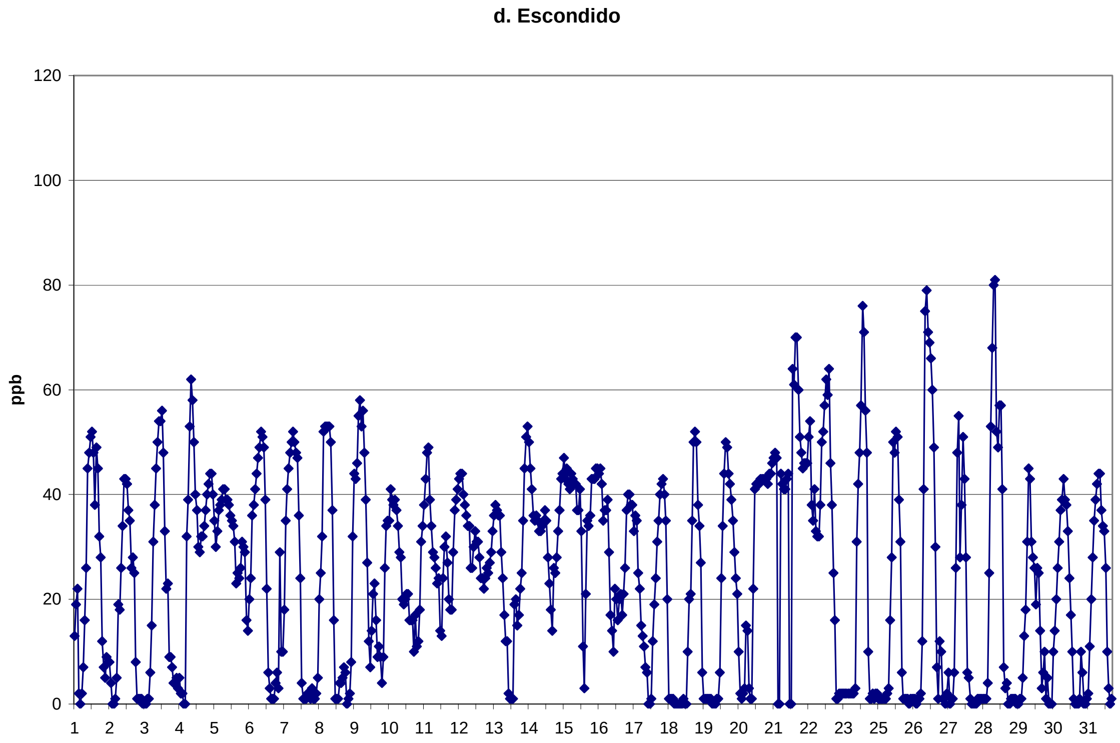

d. Escondido
ppb
120
100
80
60
40
20
0
1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
20
21
22
23
25
26
27
28
29
30
31

Figure 5 d.

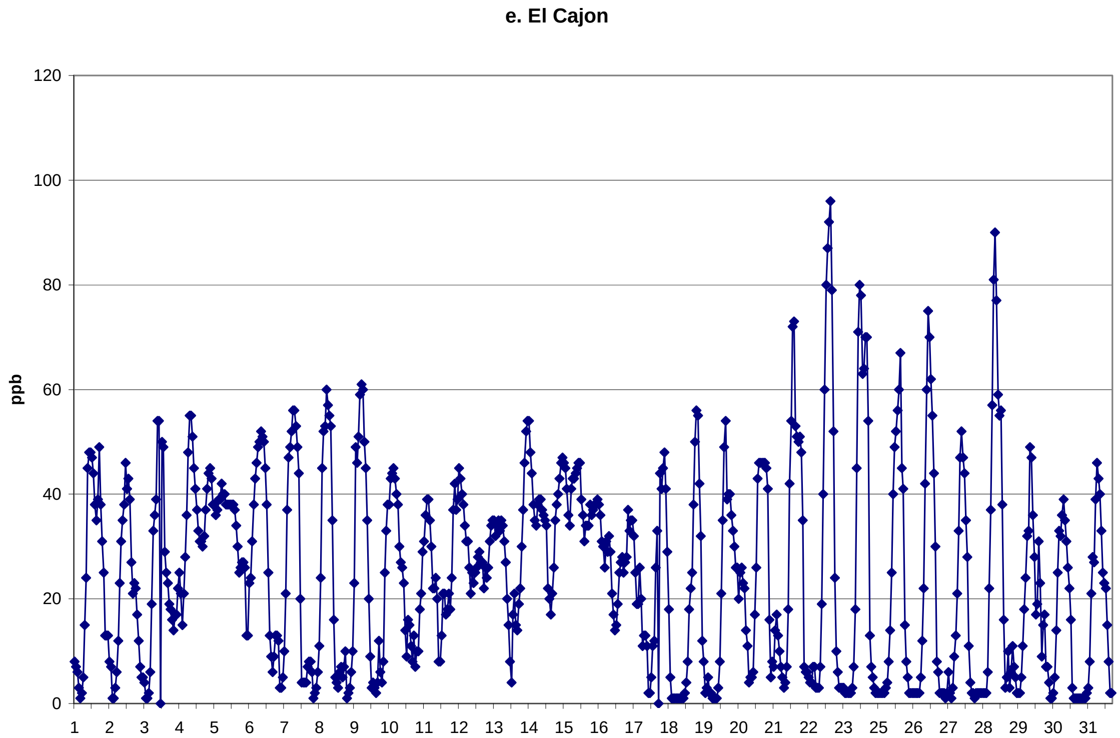

e. El Cajon
ppb
120
100
80
60
40
20
0
1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
20
21
22
23
25
26
27
28
29
30
31

Figure 5 e.

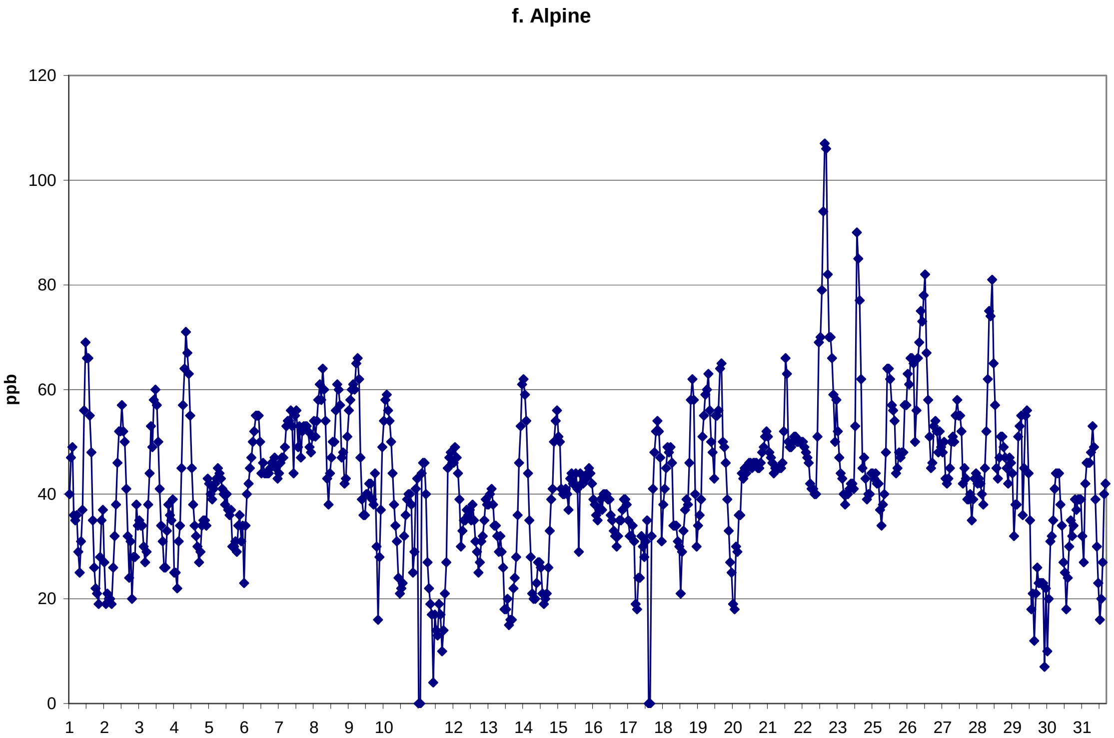

f. Alpine
ppb
120
100
80
60
40
20
0
1
2
3
4
5
6
7
8
9
10
12
13
14
15
16
17
18
19
20
21
22
23
25
26
27
28
29
30
31

Figure 5 f.

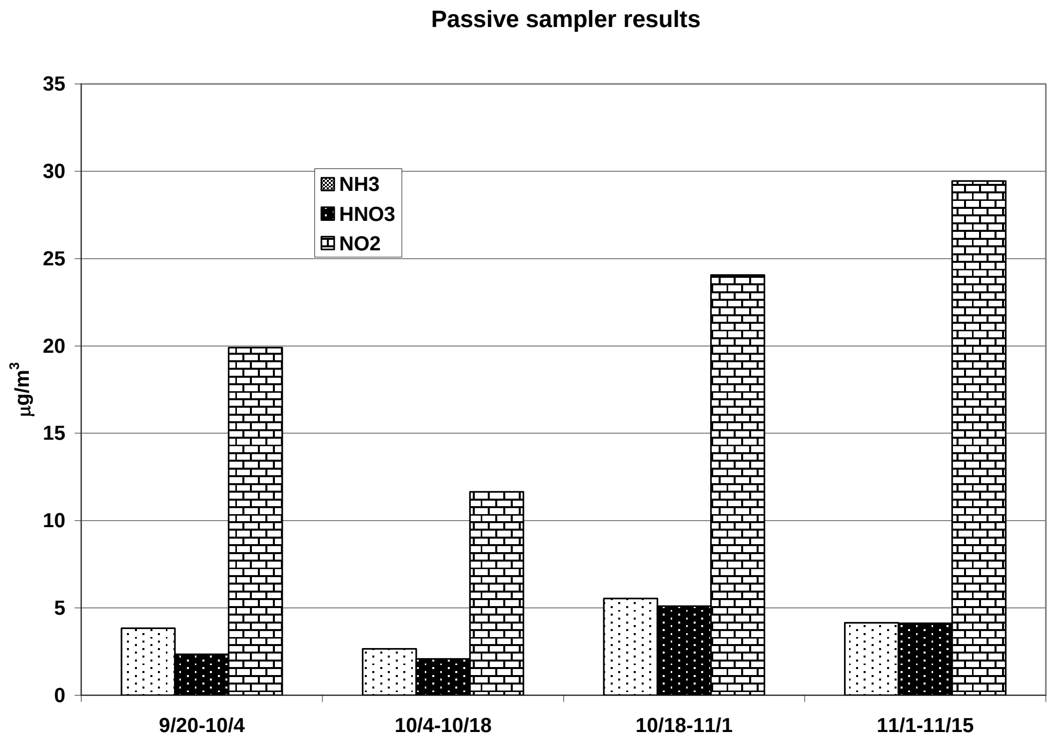

Passive sampler results
35
30
25
20
15
10
5
0
μg/m3
NH3
HNO3
NO2
9/20-10/4
10/4-10/18
10/18-11/1
11/1-11/15

Figure 6.