

Air quality impacts of the October 2003 Southern California wildfires

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[1] In Southern California, dry summers followed by hot and dry westerly wind conditions contribute to the region's autumn fire season. In late October 2003, 13 large Southern California wildfires burned more than 750,000 acres of land, destroyed over 3500 structures, and displaced approximately 100,000 people. The fire episode was declared the deadliest and most devastating in more than a decade, and local media advised individuals to stay indoors to avoid exposure to excessive levels of PM, CO, VOCs, and ozone caused by the wildfires. This study examines the actual impact of these wildfires on air quality in urban Los Angeles (LA) using "opportunistic" data from other air pollution studies being conducted at the time of the fires. Measurements of pollutant gases (CO, NO_x, and ozone), particulate matter (PM), particle number (PN) concentrations, and particle size distributions at several sampling locations in the LA basin before, during, and after the fire episode are presented. In general, the wildfires caused the greatest increases in PM₁₀ levels (a factor of 3–4) and lesser increases in CO, NO, and PN (a factor of up to 2). NO₂ levels remained essentially unchanged, and ozone concentrations dropped during the fire episode. Particle size distributions of air sampled downwind of the fires showed number modes at diameters between 100 and 200 nm, significantly larger than that of typical urban air. The particles in this size range were shown to effectively penetrate indoors, raising questions about the effectiveness of staying indoors to avoid exposure to wildfire emissions.

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1. Introduction

[2] Wildfires can produce substantial increases in the concentration of gaseous pollutants such as carbon monoxide (CO), nitrogen oxides (NO_x), ozone (O₃), and volatile organic compounds (VOCs) [Cheng *et al.*, 1998; Crutzen and Andreae, 1990] as well as particulate matter (PM) [Dennis *et al.*, 2002; Lighty *et al.*, 2000]. In recent years, there has been much interest in studying the impact of wildfires in elevating the concentrations of pollutants in the atmosphere. For instance, high CO concentrations that occurred episodically in the southeastern United States during the summer of 1995 have been attributed to large forest fires in Canada [Wotawa and Trainer, 2000]. In addition to regional and local impacts [Bravo *et al.*, 2002] wildfires contribute significantly to global emissions of atmospheric trace gases including NO_x, CO, and CO₂ [Crutzen *et al.*, 1979]. Concerns arising from PM emissions from wildfires include acute health effects, direct and

indirect climate forcing, and regional visibility [Bravo *et al.*, 2002; LeCanut *et al.*, 1996].

[3] Emission inventories by the U.S. Environmental Protection Agency (U.S. EPA) estimate that, for the calendar year 2001, wildfires in the U.S. emitted 7.1 million tons of CO, 0.98 million tons of VOCs, 0.60 million tons of PM_{2.5}, and 0.66 million tons of PM₁₀ to the atmosphere (National Emissions Inventory-Air Pollutant Emissions Trends, Current Emission Trends Summaries, August 2003, U.S. EPA), <http://www.epa.gov/ttn/chief/trends/index.html>). These amounts are significant, contributing 6%, 5%, 8% and 3% of the total CO, VOC, PM_{2.5}, and PM₁₀ emissions to the atmosphere in the United States in 2001, respectively. These figures obviously vary from year-to-year with the degree of wildfire activity, and in the severe fire season of 2000, 18% of the total PM_{2.5} emissions in the U.S. were estimated to originate from wildfires. Other emission inventories in specific areas have calculated significant NO_x emissions from wildfires as well [Dennis *et al.*, 2002]. Some systematic studies and source testing have been carried out for prescribed burns and controlled fires in North America [Einfeld *et al.*, 1991; Radke *et al.*, 1991;

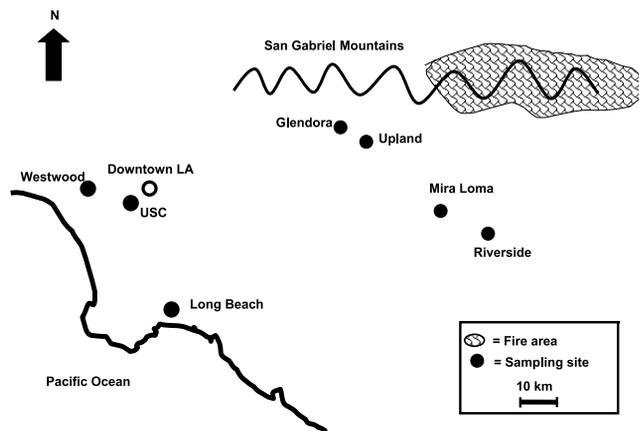


Figure 1. Map showing the fire area and the sampling sites in the Los Angeles basin.

Woods et al., 1991]. Other studies on wildfire emissions have taken advantage of existing pollution monitoring networks and other focused air pollution studies which happen to be sampling when a wildfire event occurs [*Bravo et al.*, 2002; *Brunke et al.*, 2001; *Cheng et al.*, 1998; *Goode et al.*, 2000; *Nance et al.*, 1993]. Such “opportunistic” studies can provide valuable information on wildfire pollutant emission rates and the impacts on air quality levels.

[4] Dry summers, followed by conditions of hot and dry westerly winds (known as Santa Ana winds) contribute to Southern California’s fire season in the autumn months. While the fire season usually starts around the middle of May, the exact date varies from year to year based on weather patterns and the moisture content, distribution, and amount of wild vegetation present. The fire season usually ends when cooler weather and precipitation conditions prevail. This usually occurs toward the end of October, but the fire season is occasionally extended well into January in some Southern California areas (California Department of Forestry and Fire Protection, Fire Statistics, <http://www.fire.ca.gov/MiscDocuments/FAQs.asp#13>). The presence of thick and dry foliage and bushy chaparral adds to the fire danger in the fire season in Southern California. In general, pollution levels are observed to be high during fire events [*Bravo et al.*, 2002]. The Los Angeles basin is surrounded by high mountains on three sides, opening to the Pacific Ocean to the west and southwest. The topography and frequent temperature inversions lead to the accumulation of airborne pollutants, particularly in the eastern portion of the basin, due to the prevailing westerly sea breeze [*Lu and Turco*, 1996].

[5] In late October 2003, 13 large Southern California wildfires, ranging from Simi Valley in the North to San Diego 150 miles to the south, burned more than 750,000 acres of land, destroyed over 3,500 structures, including 2,700 homes, and displaced 100,000 people. Twenty human deaths were attributed to the wildfires. The cost of the damage has been estimated to be \$2 billion. The fires having the greatest effect on the air quality of the Los Angeles (LA) Basin included the Grand Prix and Old fires in San Bernardino County and the adjacent Padua fire in Los Angeles County. These fires were located to the northeast of central Los Angeles, with Santa Ana wind

conditions, blowing toward the southwest, transporting emissions to the western portions of the Basin. The fuel was predominantly mixed chaparral, California sagebrush, annual grass and canyon live oak. Pine, perennial grass and other urban vegetation were also burned. The fires started around 23 October and had significant impacts on the air quality of the LA basin until 29 October, when the winds reversed direction and resumed their normal onshore pattern (National Interagency Coordination Centre, 2003, Statistics and Summary, http://www.nifc.gov/news/2003_statsumm/intro_summary.pdf). This fire episode was declared the deadliest and most devastating in more than a decade, and there was a significant level of worldwide press coverage. Local media advised individuals to stay indoors to avoid exposure to excessive levels of PM, CO, VOCs, and ozone caused by the wildfires. This motivated the following analysis that examines the actual impact of these wildfires on air quality and measured pollutant concentrations in urban Los Angeles. This paper presents measurements of pollutant gases (CO, NO_x, and ozone) as well as PM concentrations and characteristics at different sampling locations in the LA basin before, during, and after the October 2003 fire episode. In addition, the effect of fire on indoor particle concentrations and size distributions was also investigated. Since the fire episode could not be predicted, the current study took advantage of several preexisting air pollution studies that were being conducted at the time of the wildfires. Given the “opportunistic” nature of these samples, the measurement techniques were not necessarily targeted for fire emissions, and not all of the data is complete in all sampling sites.

2. Methods

[6] As part of the routine sampling of an ongoing study associated with the University of Southern California (USC) Children’s Health Study (CHS), supported by the South Coast Air Quality Management District and the California Air Resources Board, concentrations of carbon monoxide (CO), ozone (O₃), nitrogen oxide (NO), nitrogen dioxide (NO₂), particulate matter with aerodynamic diameters less than 10 μm (PM₁₀) and particle number (PN) are continuously measured in several locations in Southern California. Continuous data were collected concurrently throughout the calendar year 2003, and five sites within the LA Basin impacted by the wildfires were examined in this study: Long Beach, Glendora, Mira Loma, Upland and Riverside (see Figure 1). The choice of these sampling sites was based on their location within the Los Angeles Basin, the availability of the data for the desired period, and the observed impacts of the Grand Prix, Old and Padua fires. Generally, these urban sites are the most polluted among the monitoring sites of the CHS.

[7] Located near a busy surface street, the Long Beach station is about 1 km northeast of a major freeway. The Glendora station is located in a residential area nestled in the foothills of the San Gabriel Mountains. It is at least 1 km away from major roadways and 3 km from the nearest freeway. The Upland site is also located in a residential area about 6 km downwind of the Glendora site, but is located within 1 km of the I-210 freeway. The Mira Loma site is located in a building on the Jurupa Valley High School

campus. It is directly east of a major freeway interchange, is surrounded by several major warehouse facilities, and is located about 80 km east of downtown Los Angeles. The sampling location at Riverside is within the Citrus Research Center and Agricultural Experiment Station (CRCAES), a part of the University of California, Riverside. It is about 10 km southeast of the Mira Loma site and is situated upwind of surrounding freeways and major roads.

[8] The concentrations of CO were measured near-continuously by means of a Thermo Environmental Inc. Model 48C trace level CO monitor. Concentrations of NO and NO₂ were measured with a Continuous Chemiluminescence Analyzer (Monitor Labs Model 8840), and O₃ concentrations were monitored using a UV photometer (Dasibi Model 1003 AH). Total particle number concentrations (greater than about 10 nm in diameter) were measured continuously by a Condensation Particle Counter (CPC, Model 3022/A, TSI Incorporated, St. Paul, MN) set at a flow rate of 1.5 L min⁻¹. At the Upland site, the CPC was connected to a Scanning Mobility Particle Sizer (SMPS, Model 3936, TSI Incorporated, St. Paul, MN), to measure the size distribution of submicrometer aerosols (15–750 nm) using an electrical mobility detection technique. In this configuration, the CPC flow rate was maintained at 0.3 L min⁻¹ (with the sheath flow of the SMPS set at 3 L min⁻¹), and particle number counts were calculated from the SMPS size distributions. Unfortunately, due to a brief power outage and limited site access resulting from the nearby fires, SMPS data were lost from the morning of 24 October to noon of the 29 October (the peak of the fire impact). However, the other monitors at this site continued to function properly in this time window. Continuous particle number and gaseous copollutant concentrations were averaged to form 1-hour and 24-hour average values for the subsequent analysis.

[9] Hourly PM₁₀ mass concentrations in each site were measured by a low temperature Differential Tapered Element Oscillating Microbalance monitor (low temperature TEOM 1400A, R&P Inc., Albany, NY). The design and performance evaluation of this monitor is described in greater detail by *Jaques et al.* [2004]. Briefly, the system consists of a size-selective PM₁₀ inlet, followed by a Nafion[®] dryer that reduces the relative humidity of the sample aerosol to 50% or less. Downstream from the Nafion dryer and ahead of the TEOM sensor is an electrostatic precipitator (ESP) allowing for the removal of particles from the sample stream. The ESP is alternately switched on and off, for equal time periods of about 10 min. This dual sampling channel design makes it possible to account for effects such as volatilization of labile species, adsorption of organic vapors and changes in relative humidity and temperature, all of which affect the TEOM signal. The study by *Jaques et al.* [2004] showed that the time averaged TEOM PM₁₀ mass concentrations agreed within ±10% with those of collocated Federal Reference Methods (FRM).

[10] In addition to the data collected at the CHS sites, semicontinuous PM_{2.5} (fine) and ultrafine PM mass concentrations were measured at the Southern California Super-site located near downtown Los Angeles at the University of Southern California (USC). Two-hour PM mass concentration data were collected with a Beta Attenuation Monitor (BAM, Model 1020, Met One instruments, Inc., OR)

[*Chung et al.*, 2001]. The BAM consisted of a size-selective inlet (2.5 μm for fine and 0.15 μm for ultrafine) [*Chakrabarti et al.*, 2004], a filter tape, a beta radiation source, and a beta radiation detector. The difference in the transmission of beta radiation through the filter tape before and after a particulate sample has been collected, is measured and used to determine the mass of collected particulate matter. Continuous operation is achieved by automatic advancement of the filter tape between sampling periods.

[11] Finally, in a concurrent but unrelated study, particle size distributions were measured indoors and outdoors of a two-bedroom apartment in the Westwood Village area near the University of California, Los Angeles. The residence is located about 100 m mostly downwind (east) of the I-405 freeway, a very busy traffic source. A Scanning Mobility Particle Sizer (SMPS 3936, TSI Inc., St. Paul, MN) was set up in a bedroom and sampled alternating indoor and outdoor size distributions on a 24-hour basis. The aerosol sampling flow rate of the SMPS was set to 1.5 L min⁻¹ in order to measure particles as low as 6 nm as well as to minimize the diffusion losses of ultrafine particles during sampling. The maximum size detectable at these settings was 220 nm, and a scan time of 180 s was used. The sampling lines were kept the same length and as short as possible (1.5 m) for both indoor and outdoors samples. Measurements were made through a switching manifold that alternately sampled indoor and outdoor air, each for 9-min periods, in which three size distributions were taken in sequence. There were no known major indoor sources of aerosols in the residence for the period from 1000 to 1900 LT, when the residents were at work and from 2300 to 0700 LT when the residents were asleep in the other bedroom. The door of the sampling bedroom was always kept closed to minimize the influence of any other possible indoor activity. The residence was under natural ventilation with windows closed at all times during the sampling period. This study provided a unique opportunity to monitor infiltration of PM of outdoor origin into the indoor environment, and to estimate indoor exposures to PM from the wildfires.

3. Results and Discussion

[12] Figures 2a–2e present the 24-hour average concentrations of CO, NO, NO₂, O₃, PM₁₀ and particle number (PN) before, during and after the October fire period in Southern California at the five CHS sampling sites examined in this study. A summary of the average concentrations of the pollutants before, during and after the fire is given in Table 1. As surmised from the news reports and the data, the period of fire influence was from 23–29 October. Figure 2 clearly shows that the concentrations of all the pollutants drastically decreased on 30 October and then increased back to more typical levels by 4 or 5 November. The rapid decline is associated with the wind reversal on the afternoon of 29 October when an onshore wind pattern replaced the Santa Ana conditions, followed by rainfall on 30 and 31 October. Figure 3a displays a satellite photo from NASA Earth Observatory on 28 October 2003 showing the extent of the fires and the prevailing wind direction during the peak of the fire episode. On 29 October, the winds shifted to an onshore pattern (Figure 3b) blowing fresh fire emissions

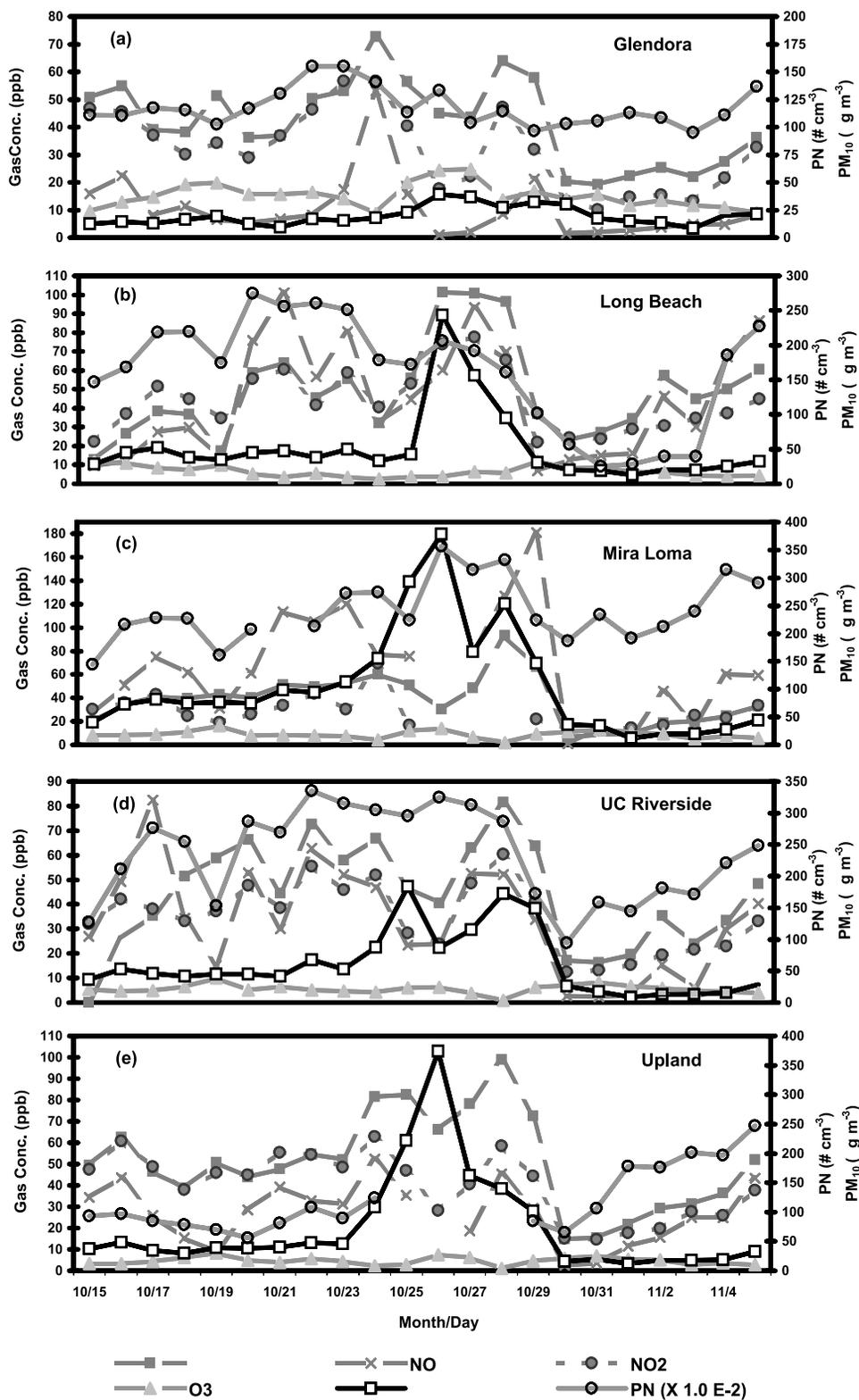


Figure 2. The 24-hour averaged PM and gaseous pollutant concentrations during the study at (a) Glendora, (b) Long Beach, (c) Mira Loma, (d) UC Riverside and (e) Upland. For comparison purposes, CO concentrations (in ppb) have been divided by 20, and PN concentrations (in cm^{-3}) have been divided by 100, as indicated in the legend.

Table 1. Average Hourly Concentrations of Pollutants With the Standard Deviation at the Five CHS Sites Before, During, and After the Fire^a

	CO, ppm	NO, ppb	Average Concentration, \pm SD			PN, particles cm^{-3}
			NO ₂ , ppb	O ₃ , ppb	PM ₁₀ , $\mu\text{g m}^{-3}$	
<i>Prefire</i>						
Glendora	9 \pm 3	11 \pm 16	37 \pm 16	37 \pm 21	12 \pm 14	10,400 \pm 5500
Long Beach	6 \pm 6	23 \pm 49	47 \pm 19	29 \pm 18	33 \pm 16	19,300 \pm 12,400
Mira Loma	6 \pm 4	45 \pm 54	29 \pm 14	25 \pm 26	61 \pm 35	16,200 \pm 8200
UC Riverside	8 \pm 6	40 \pm 29	33 \pm 19	29 \pm 29	47 \pm 23	16,200 \pm 12,100
Upland	10 \pm 4	24 \pm 28	44 \pm 16	21 \pm 23	39 \pm 18	9000 \pm 3700
<i>During Fire</i>						
Glendora	11 \pm 5	25 \pm 30	39 \pm 28	44 \pm 23	27 \pm 25	12,200 \pm 6200
Long Beach	14 \pm 9	55 \pm 68	56 \pm 24	15 \pm 16	93 \pm 92	18,000 \pm 8500
Mira Loma	12 \pm 8	105 \pm 85	39 \pm 26	17 \pm 18	215 \pm 171	28,500 \pm 14,600
UC Riverside	12 \pm 7	46 \pm 36	42 \pm 22	18 \pm 21	121 \pm 112	28,800 \pm 16,100
Upland	15 \pm 7	43 \pm 34	47 \pm 24	15 \pm 16	165 \pm 138	data not available
<i>Postfire</i>						
Glendora	5 \pm 2	5 \pm 5	17 \pm 11	31 \pm 11	18 \pm 29	11,000 \pm 6300
Long Beach	8 \pm 6	39 \pm 49	32 \pm 11	16 \pm 12	21 \pm 10	8600 \pm 9700
Mira Loma	4 \pm 3	57 \pm 45	20 \pm 11	19 \pm 15	28 \pm 16	23,900 \pm 10,700
UC Riverside	6 \pm 4	14 \pm 25	20 \pm 10	23 \pm 15	18 \pm 10	1,7400 \pm 11,000
Upland	6 \pm 4	21 \pm 25	23 \pm 12	17 \pm 13	19 \pm 10	16,700 \pm 8600

^aData in bold indicate statistically significant differences between the prefire and during-fire concentrations at $p = 0.05$.

toward the east away from the LA Basin. The fires continued to burn for many days after, but the cooler and wetter weather helped the firefighting effort and the fires were under control within another week.

[13] The data summary in Table 1 indicates that with the exceptions of NO₂ and O₃, the concentrations of CO, NO, PM₁₀ and PN during the fire event were significantly higher (at the $p = 0.05$ level) than their respective values preceding the fire event. Statistical comparisons between during and postfire concentrations was not conducted, because, as evident from the data in Table 1 and Figure 2, the unstable and wet weather conditions during the week of 30 October to 5 November resulted in lower than average air pollutant concentrations. It is of particular note, however, that the most dramatic increase in the concentrations of any pollutant during the fire events was observed for the PM₁₀ concentrations, which, with the exception of one site (Glendora), rose by almost three to four-fold in all sites during this period. While typical PM₁₀ concentrations in Los Angeles are on the order of 50 $\mu\text{g m}^{-3}$ or less [Christoforou *et al.*, 2000], levels rose to near or above 200 $\mu\text{g m}^{-3}$ at some sites during the fires. PM₁₀ levels at Glendora did not rise to the same degree, possibly due to the site's location at the base of a canyon in the San Gabriel Mountains. The Santa Ana winds tend to blow down the mountain canyons, and there was little or no fire activity in or upwind of this particular canyon. Upland, on the other hand, was within 2–3 km and directly downwind of extreme wildfire activity. The other three sites were all further downwind from the wildfires, but all sites experienced atypical PM₁₀ levels. It is possible that the higher wind speeds during Santa Ana conditions increased re-suspended dust emissions that contributed to the elevated PM₁₀ levels. This effect, if dominant, should be observed at all sites. However, the fact that Glendora PM₁₀ levels remained within the “typical” range indicates that the impact of fire smoke plumes is the main cause of the elevated PM₁₀ levels. Previously reported data during

Santa Ana events without fires also demonstrate that such high levels of PM₁₀ are not typically observed on a 24-hour basis [Geller *et al.*, 2004].

[14] By contrast, the total particle number concentrations, also shown in Figure 2, did not exhibit the same extreme concentration increases during the fires. PN levels increased significantly only in Mira Loma and perhaps Riverside, and only by an approximate factor of two. Even these higher levels of PN have been observed on occasion under typical, nonfire influenced, conditions in the LA Basin [Kim *et al.*, 2002]. No significant increase in PN was observed at Long Beach, and Glendora, the latter being minimally affected by the fires as discussed above. Owing to the aforementioned power outage, PN data were not available at the closest site to the fires, Upland, during the wildfire period. Emissions testing of foliar fuels demonstrate that high particle number levels are emitted from these sources. However, given the observed high PM mass levels, and thus the increased PM surface area in the fire smoke plumes, it is conceivable that emitted smaller particles are scavenged by coagulation with larger particles in the smoke plume [Formenti *et al.*, 2003]. This process may occur over the few hours that it takes for the fire particles to reach our sampling sites. Many of the smaller particles, which make up the majority of particle number concentrations, may no longer exist as individual particles. Thus PM mass levels remain high while PN levels are diminished. This hypothesis may explain why the largest PN increase was seen at Mira Loma and Riverside, both of which are much closer to the fire areas than the sites further downwind such as Long Beach.

[15] Similar to particle number, CO concentrations at these sites were only modestly affected by the fires. With the exception of Glendora, the observed increases were statistically significant at the $p = 0.05$ level, but the degree of increase was much less than that observed for PM₁₀. Mira Loma, Upland and Long Beach experienced CO around twice normal levels during the fire. As in the case of PN, CO concentrations in the area of Glendora appear to be

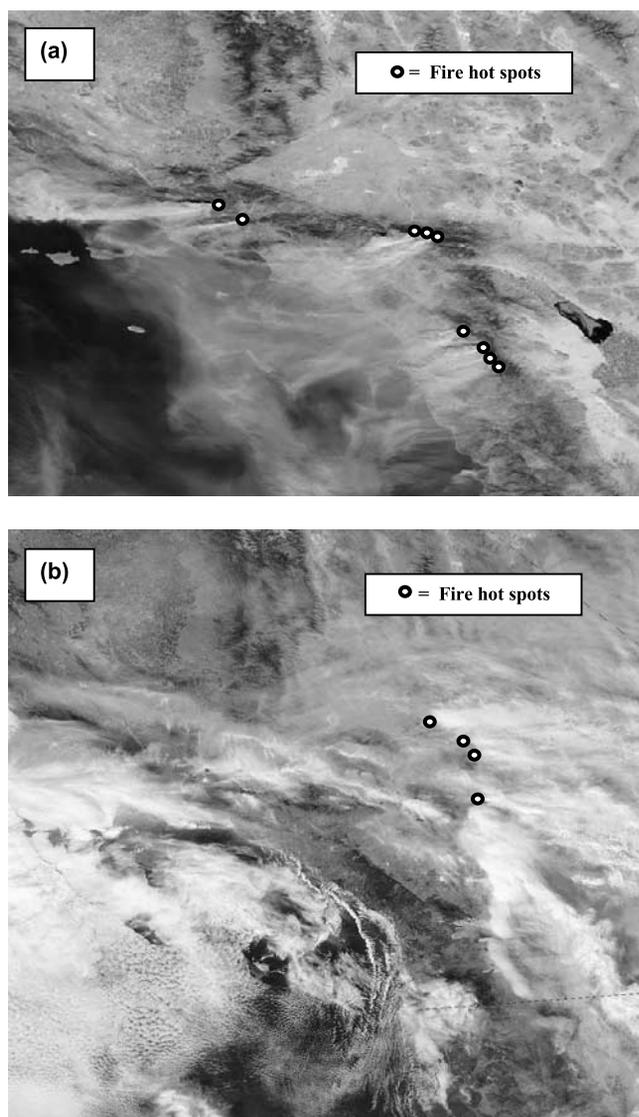


Figure 3. Satellite images from NASA earth observatory showing (a) Southern California during the peak of the fire episode on 28 October 2003, with the smoke plumes blowing west, and (b) the same area after the wind reversal with a visible marine layer and blowing the smoke plumes toward the northeast on the afternoon of 29 October 2003.

unaffected by the fire events. The relatively low increase in CO due to the fires can be explained by other, more significant sources of CO in Los Angeles. Emission factors from the U.S. EPA (AP-42, Fifth Edition, Volume I-Chapter 13.1: Wildfires and Prescribed Burning, U.S. EPA, October 1996) and other studies [Barbosa *et al.*, 1999; Pereira *et al.*, 1999; Scholes *et al.*, 1996] show that the ratio of CO mass to PM₁₀ mass in wildfire emissions lies typically between 8 and 16. The same ratio for various motor vehicles under varying driving conditions is much higher, ranging from about 200 to over 2000 [Cadle *et al.*, 2001; Chase *et al.*, 2000]. In urban areas dominated by vehicular sources, wildfires will thus affect ambient levels of CO to a lesser degree than the ambient levels of PM₁₀. A review of historical pollutant data during Santa Ana conditions without fire activity (9 February 2002 and 6 January 2003)

shows that CO levels can diminish due to fewer CO sources upwind and increased basin ventilation. However, this effect is inconsistent, and varies greatly with sampling site and from event to event. Thus no true “Santa Ana baseline” can be established for comparison purposes. For this reason, comparisons are limited to the “typical” conditions before the fire episode.

[16] NO concentrations follow similar trends with those for CO and PN (i.e., they increase significantly in every location during the fire) but this increase is on the order of two-fold or less, hence smaller than the increase observed for PM₁₀. While the increase in NO concentrations during the fire event seems to be minor at the Riverside location, the nearby Mira Loma site shows more than double the NO levels relative to levels before the fire events. It is possible that Mira Loma may have been more directly downwind of fire areas than Riverside, which would explain this discrepancy. This is supported by the observed PM₁₀ levels at these two sites, which also increased more dramatically in Mira Loma than in Riverside. Relative to NO, PN, and PM₁₀, the effect of fires was negligible for NO₂ as the concentrations did not change significantly in any of the five sampling sites during the fire events. While some NO₂ is emitted directly from combustion processes, most of the NO₂ in urban air is formed in the atmosphere by the reaction of NO with ozone. Under normal conditions in Los Angeles, NO, and thus NO₂ levels are dominated by diesel vehicle emissions [Fujita *et al.*, 2003]. However, the NO increases observed during the fires were not accompanied by corresponding increases in NO₂ concentrations. Although no conclusive explanation can be determined from the current data, it is possible that the PM in the fire smoke blanketing the LA basin blocked incoming solar radiation and thus reduced photochemical activity in the atmosphere. This would result in lower ozone levels and thus lower observed levels of NO₂. Increased concentrations of organic gases (VOCs) emitted by the fires may also play a role in the complex atmospheric chemistry of NO, NO₂, and ozone [Cheng *et al.*, 1998]. Interestingly, with the exception of Glendora, which experienced marginally (but not significant) increased O₃ concentrations during the fire episode, the concentrations of O₃ decreased by about 25–50% at all the other sites during the fire period. As mentioned above, the fire smoke covering the basin and the corresponding reduction in photochemical activity may be a possible explanation for this decrease in concentration.

[17] The effect of the wind direction change can also be seen in the hourly concentrations of the measured pollutants in Upland as shown in Figures 4a and 4b. The high concentrations of PM₁₀ at Upland can be clearly seen during the entire fire period, with the highest hourly concentration measured at 769 $\mu\text{g m}^{-3}$. On 29 October, at 12:00 P.M., the PM₁₀ level was 153 $\mu\text{g m}^{-3}$ and within one hour it dropped to 65 $\mu\text{g m}^{-3}$. Within four hours, PM₁₀ concentrations dropped to below 20 $\mu\text{g m}^{-3}$. This marks the time of the wind reversal mentioned above. Unfortunately, hourly data of particle number concentrations in this time frame are not available due to the power outage. Similar to the 24-hour data, the hourly gaseous pollutant levels did not increase as much as the PM₁₀ levels during the period of wildfire influence. However, with the exception of ozone, concentrations of all the gaseous pollutants dropped precipitously when the wind reversal occurred.

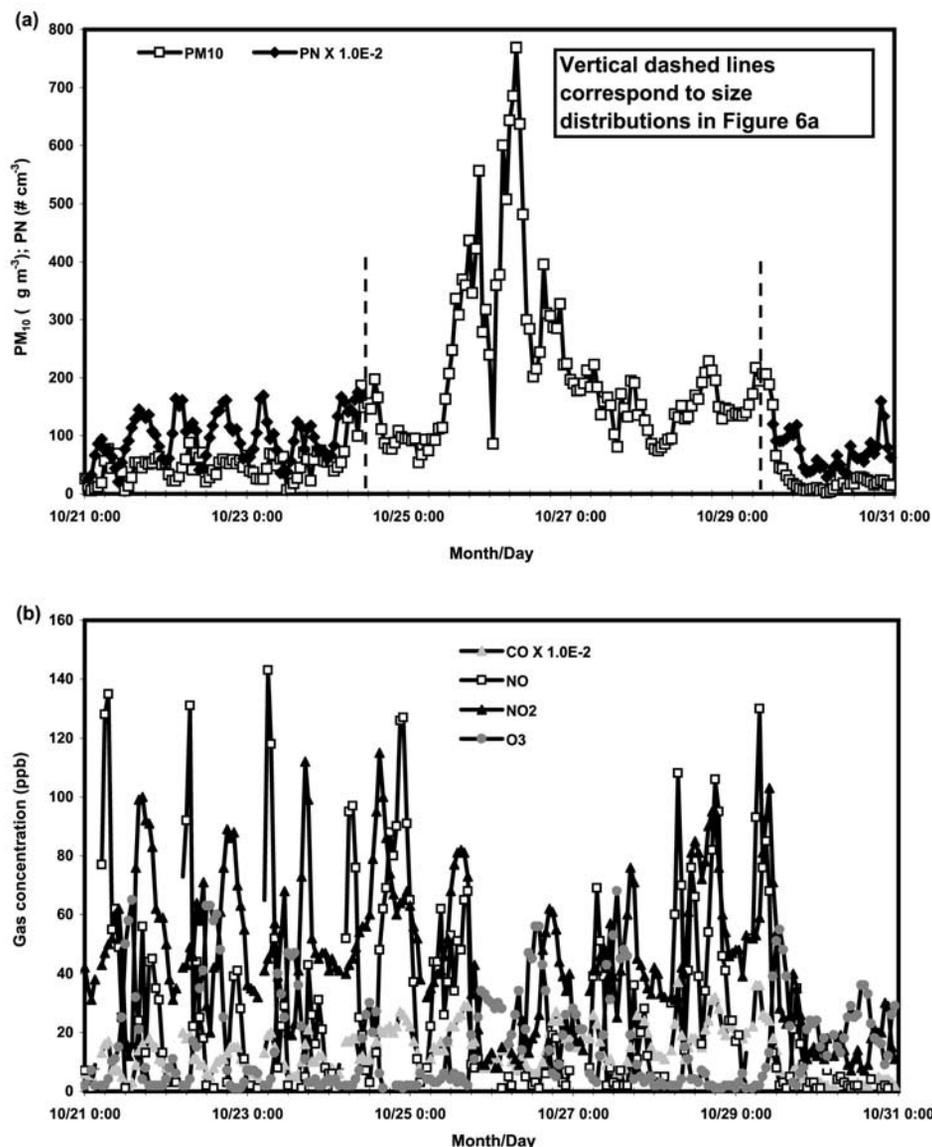


Figure 4. Hourly (a) PM and (b) gaseous pollutant concentrations at Upland.

[18] Semicontinuous ultrafine and fine (PM_{2.5}) particle mass concentration data support the argument that the atmospheric concentrations of smaller particles (measured above as PN), increased to a lesser extent than the larger particles. Figure 5 displays the 2-hour ultrafine and fine PM mass obtained from the BAM measurements at the USC site. The average ultrafine particle mass concentrations increased from an average value of 5.4 (± 2.3) to 6.9 (± 2.7) $\mu\text{g m}^{-3}$. While this increase is statistically significant ($p < 0.01$), it is still less dramatic than the obvious increase in PM_{2.5} during the fire events. The average concentration of PM_{2.5} more than doubled, from 19.1 (± 5.2) to 51.3 (± 26.1) $\mu\text{g m}^{-3}$. The highest fine particle mass measured during the fire episode at USC was 115 $\mu\text{g m}^{-3}$. The wind reversal was marked by a steep reduction in fine particle mass midday on 29 October when the fine PM dropped from 105 $\mu\text{g m}^{-3}$ in the morning to 25 $\mu\text{g m}^{-3}$ by 2:00 P.M.

[19] Figures 6a and 6b show the one-hour averaged particle size distribution at Upland corresponding to the

times marked by vertical lines in Figure 4a. Because of the loss of SMPS data for almost entire fire period, we have selected times just before (Figure 6a) and just after (Figure 6b) the power outage. The particle size distribution at a given hour (1000 and 1200 LT) was averaged for different days before and after the fire, and compared to the same hour during the influence of the fires. It can be seen that the size distribution corresponding to the periods of fire influence significantly differs from those without the fire influence. The mode in the number-based particle size distribution spans from 100 to 300 nm and is indicative of the wildfire smoke. Previous emissions testing have shown similarly large number modes in the particle size distributions from the burning of foliar fuels [Hays *et al.*, 2002]. Such large diameter number modes are not normally seen in urban locations [Kim *et al.*, 2002] where particle number concentrations are dominated either by primary vehicular emissions or by nucleation processes [Woo *et al.*, 2001]. Since particle volume is proportional to the cube of the

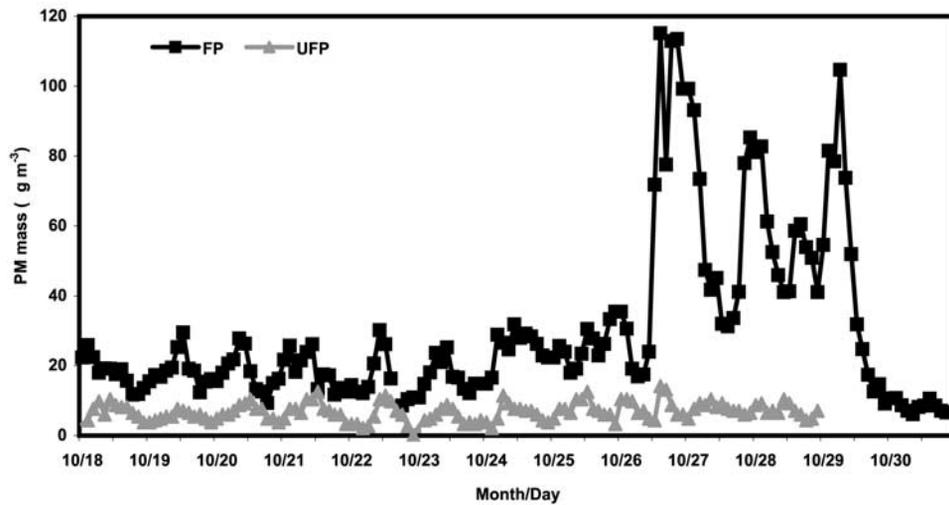


Figure 5. Two-hour averaged fine (FP) and ultrafine (UFP) particle mass concentrations at USC.

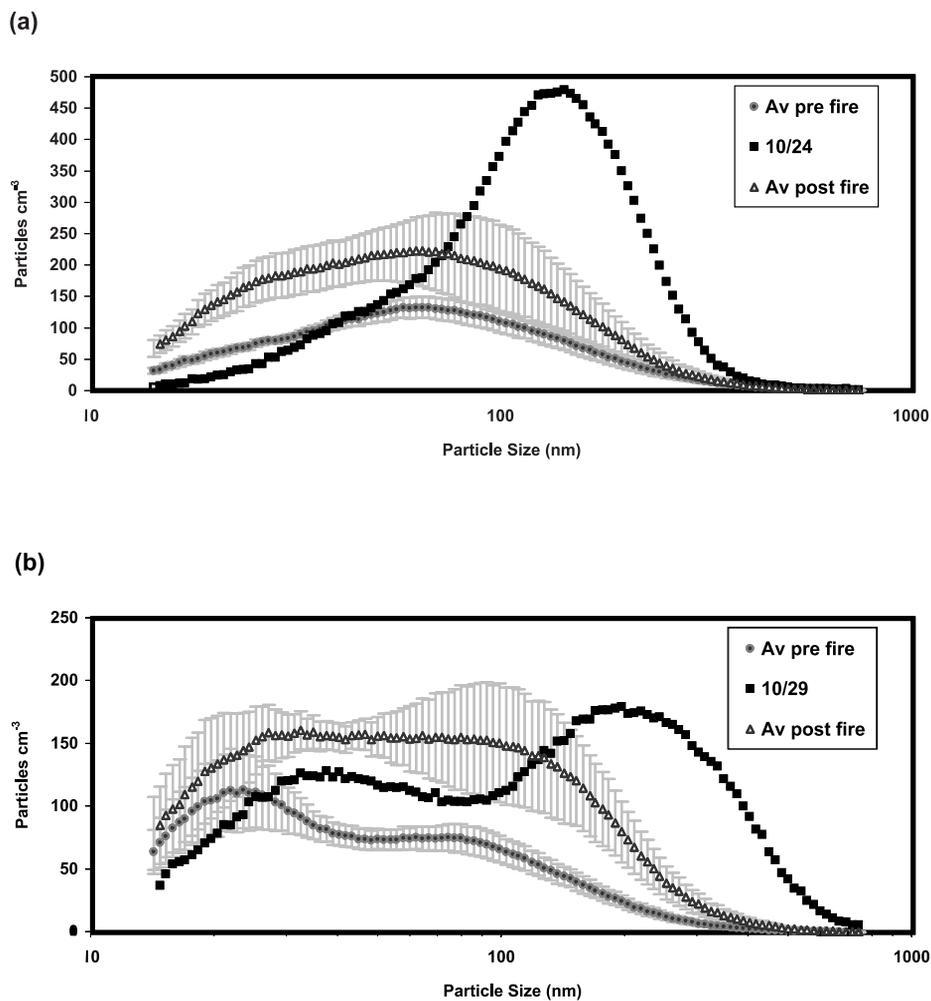


Figure 6. Particle size distributions at Upland (a) at 1000 LT before (24 October 2003) and after the fires and b) at 1200 LT before (29 October 2003) and after the fires.

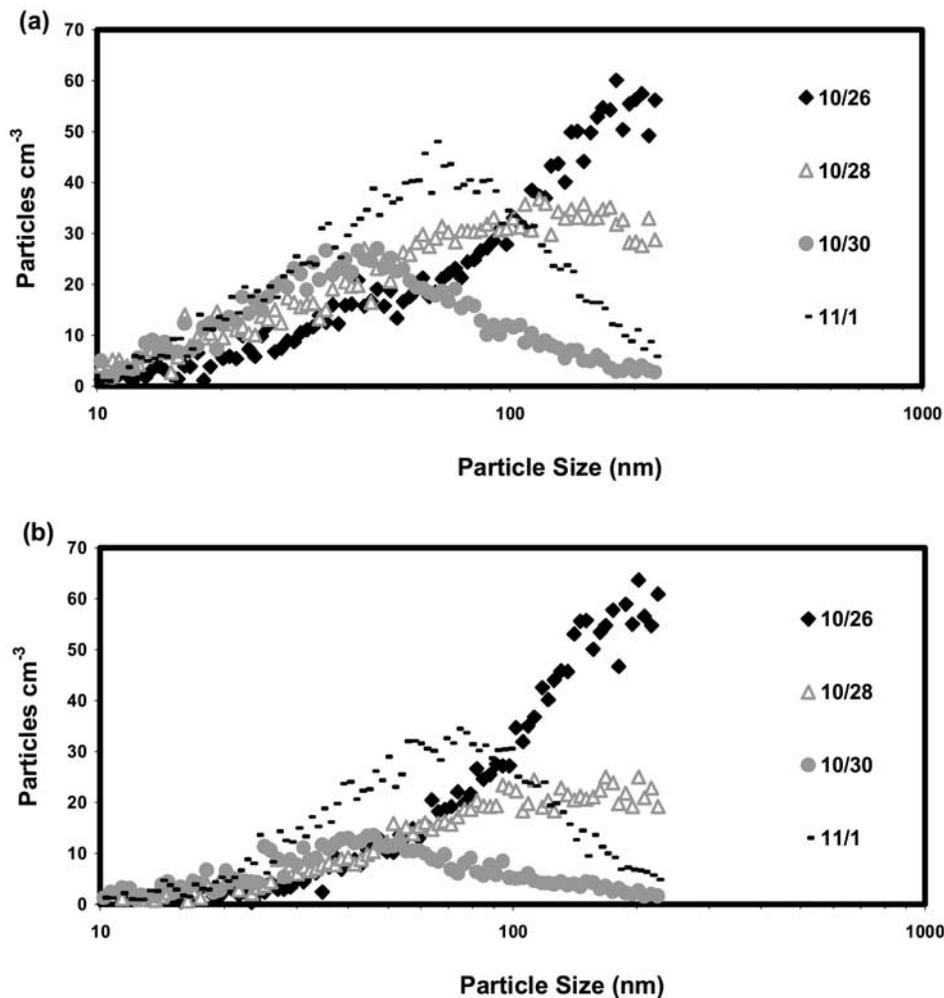


Figure 7. Particle size distributions on different days at 2300 LT in Westwood Village: (a) outdoor and (b) indoor.

diameter, a modest increase in particle number concentrations in these larger size modes is sufficient to account for the larger increases observed for PM mass.

[20] Indoor and outdoor particle number size distributions were also available from a concurrent study near UCLA in the western portion of the Los Angeles Basin. Figures 7a and 7b display ambient and corresponding indoor particle size distributions from 2300 to 2400 LT for different days during and after the fire events. This period was selected to minimize the influence of any possible indoor sources (i.e., cooking, cleaning) and outdoor traffic from the nearby freeway. The effect of the fires on indoor concentration is evident, with an aerosol mode diameter at about 200 nm on 26 and 28 October, and then a shift to a lower size range (between 50 to 70 nm) on 30 October and 1 November, respectively. Number concentrations both indoors and outdoors also decrease as we move away from the fire period. It is of interest to note that on 26 October (i.e., in the middle of the wildfire period), the indoor and outdoor size distributions are virtually identical in both number concentration and mode, which suggests that the majority of the outdoor aerosol infiltrated indoors with a penetration value close to 1. This is not a surprising result, considering that based

on our measurements, the majority of the particles emitted from the fire are in the 100–300 nm range. This is also the range of maximum indoor penetration of outdoor aerosols and minimum indoor deposition rate [Allen *et al.*, 2003; Long *et al.*, 2001]. As the mode in aerosol size distributions shifts to smaller sizes, the indoor concentrations are approximately 50–75% lower than outdoors, which is also consistent with the penetration values determined by Long *et al.* [2001] and Wallace and Howard-Reed [2002] for the particles in the 40–80 nm range.

[21] To put the above results in perspective, Figures 8a and 8b show the measured indoor and outdoor particle size distributions during the morning traffic commute period, from 0600 to 0700 LT, while the wildfires were still active (27 October) and after the fire event (4 November). The outdoor size distribution on 27 October is characterized by one dominant mode at about 25 nm, which is associated with vehicular emissions [Zhu *et al.*, 2002a, 2002b], followed by a second mode at about 200 nm, which reflects the influence of the wildfires. The indoor size distribution for that date (Figure 8a) shows that the super-100 nm particles are virtually at identical concentrations with their corresponding outdoor levels, whereas the concentrations

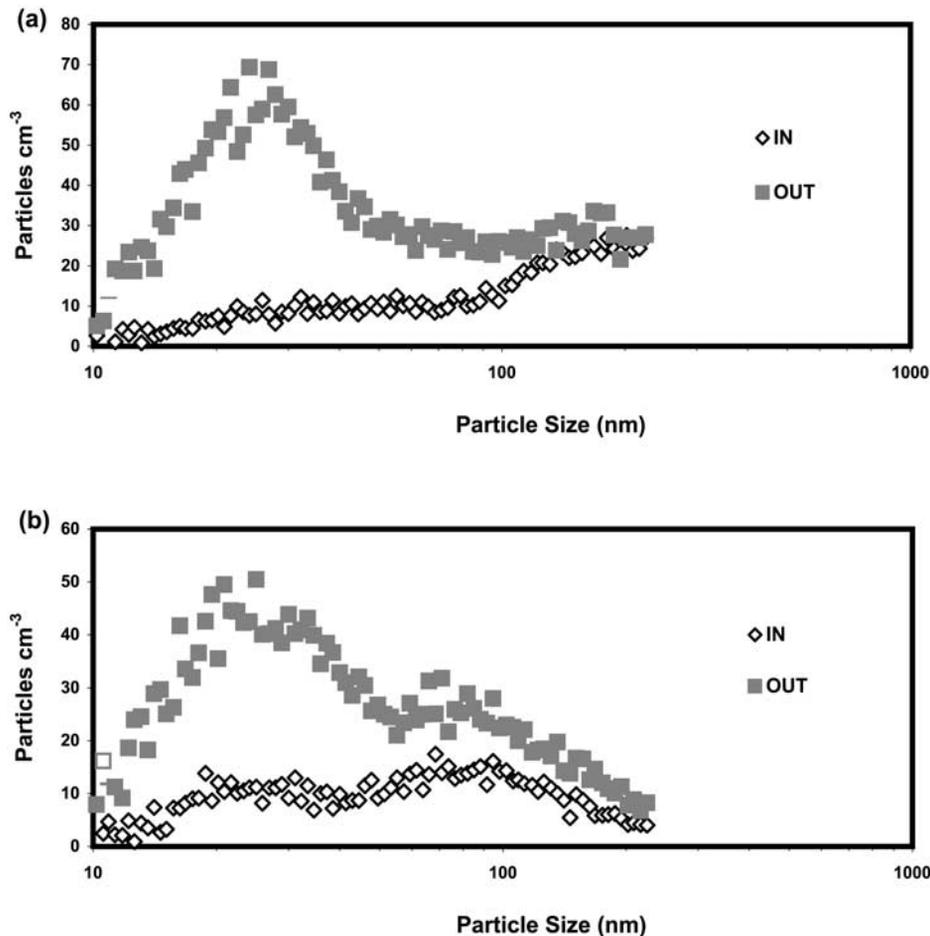


Figure 8. Indoor/outdoor particle size distributions at 0600 LT in Westwood Village on (a) 27 October 2003 and (b) 4 November 2003.

of smaller particles indoors are substantially lower than those outdoors. Similar trends are also shown in Figure 8b, with the exception that the second mode in the 200 nm range observed during the fire period no longer exists in either the indoor or outdoor environment.

[22] The data plotted in Figures 8a and 8b indicate an average outdoor-to-indoor penetration ratio of about 0.15 to 0.20 for particles in the 20–50 nm range, which, as stated above, originate from nearby traffic sources. This value is somewhat lower than the indoor penetration ratios reported by Long *et al.* [2001] and Wallace and Howard-Reed [2002] for that size range, which normally range between 0.3–0.7, depending on home characteristics and air exchange rates. One possible explanation for the lower values observed in our study may be that, as shown in recent reports in the literature [Sakurai *et al.*, 2003; Tobias *et al.*, 2001], sub-50 nm particles from vehicular emissions consist of semivolatile material, compared to the mostly nonvolatile particles in the 50–100 nm range. Thus, after penetrating indoors, they may have completely evaporated or shrunk to sizes below about 6 nm, the lower size detection limit of the SMPS. It is unknown what source, size or composition of ambient PM is responsible for the observed health effects. However, our results show that the prevailing advice during the fire episode for people to stay indoors may not be

effective in reducing exposure to most of the particles emitted from wildfires.

4. Summary and Conclusions

[23] Coincidental air pollution sampling campaigns proved valuable in determining the impacts of the October 2003 wildfire episode on pollutant levels in the Los Angeles Basin. The greatest impact was observed on PM₁₀ concentrations which increased by factors of three or four depending on location. CO and NO levels increased to a lesser extent (a factor of approximately two), most likely due to the different relative emission rates of these pollutants from wildfires compared to typical urban sources such as traffic. Particle number concentrations and NO₂ were essentially unchanged, except at the sites nearest the fires where PN levels almost doubled. Ozone levels during the fires were observed to be lower during the fires at some sites, a possible result of light scattering by the smoke plume reducing photochemical activity levels. Particle number distributions downwind of the fires displayed number modes with diameters between 100 and 200 nm, larger than typical urban aerosol and explaining the larger increases in PM₁₀ and PM_{2.5} mass concentrations than that for ultrafine particle mass and particle number. These particles were also

shown to penetrate effectively indoors, calling into question the prevailing advice to the public to remain inside to avoid exposure to harmful wildfire emissions.

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