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Measuring Individual Ozone Exposure in Los Angeles Urban Parks

Pamela E. Padgett, Patricia L. Winter, Lee-Anne Milburn, and Weimin Li

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Cover photo: Angeles National Forest

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Abstract

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Exposure to ozone pollution has serious health risks. Damage to lungs and impairment of cardiovascular health are of particular concern for vulnerable populations, including children, the elderly, and individuals from "disadvantaged" communities whose health may already be compromised. This report describes three successive studies that led to the development of techniques for directly monitoring human exposure to ozone. Experiment 1 was a proof of concept and exploration of potential systematic errors. Experiment 2 was a more extensive use of personal monitoring to evaluate differences in ozone exposure based on differing urban structure, which contrasted "disadvantaged" and "affluent" communities. These communities varied across multiple environmental (e.g., percentage of canopy cover), social (e.g., percentage of communities of color), and economic (e.g., median household income) criteria. Experiment 3 built on earlier studies with modifications to address the codependency of ozone concentration and geography as well as the high daily variability of ambient ozone concentrations, which requires that comparisons between different locations be conducted on the same day.

Passive sampling techniques were used to monitor human exposure. Typically these devices are used to monitor ambient conditions in remote areas. When deployed for those types of studies, the devices are protected from the direct effects of wind by baffles and shields; thus ambient conditions are calculated by diffusion of ozone across still air rather than uninterrupted impacts of ozone on the detection surface. In these studies, we amended the normal deployment protocols so that the passive samplers were worn by individuals in a way to mimic the direct exposure of the respiratory system to ozone as a result of wind, body movement, and position.

The current regulatory standards for ozone exposure were established to protect human health; however, direct monitoring of individuals is seldom implemented. Our data suggest that humans engaged in outdoor activities are frequently exposed to several times the 8-hour standard of 65 parts per billion set for vulnerable populations.

Keywords: Ozone, air quality, passive ozone monitor, urban parks, Los Angeles, health, recreation.

Summary

This report reflects a progression of work across three studies inspired by a paired social-ecological systems approach to consider the benefits and risks of outdoor recreation participation. Work demonstrating numerous benefits of outdoor recreation support the value to participants. While some work has been done on possible risks of outdoor recreation, and some literature has asserted caution for leisure and physical activities, we were intrigued by the implications of degraded air quality in outdoor settings. Earlier work identified the probable lack of awareness of poor air quality among recreationists, although more recent work suggests public warnings about degraded air quality results in shifts in outdoor activities. With the probable increase in ozone associated with climate change effects, and the already reported degraded air quality in some prime forested areas to which recreationists travel to enjoy clean air, we hoped to develop a reasonable onsite approach to measuring ozone exposure.

For our first experiment, we took advantage of a planned recreation survey on the Angeles National Forest by adding an assessment of ozone exposures. Our aim was to test the viability of using a passive ozone monitoring approach that was portable and non-intrusive. As this report indicates, the first experiment was somewhat successful; however, several adjustments to the approach were deemed important for its continued application.

For the second experiment, we combined social (recreation observations) and ecological (ozone monitoring) investigations in both disadvantaged and affluent communities in Los Angeles, focusing on urban parks as a nexus for potential resilience. Studies have asserted that urban parks may serve as an important buffer to climate change effects in cities. Knowing that ambient ozone concentrations in Los Angeles are frequently in the unhealthy range, we used personal ozone monitoring to explore the idea that outdoor recreationists may be exposed to higher health risks because air pollution is part of their recreation experience. We also explored the potential for subsequent impacts on these climate change buffers. The second experiment showed that passive monitoring seemed to detect elevated exposure in parks in disadvantaged communities. However, a confound with geographic placement and the inability to contrast park data on the same days were clear limitations to our ability to draw conclusions from these data.

In the third and final experiment described in this report, we outline findings from a systematic comparison of ozone exposure experienced by recreationists in urban parks in the city of Los Angeles in both affluent and disadvantaged communities across inland and coastal zones. Findings suggest that recreationists are exposed to ozone levels that exceed recommended health thresholds. Our

methodological approach offers a guide to others seeking to understand the possible levels of air pollution in outdoor recreation and leisure settings. Additionally, we believe our findings suggest concern for future resilience owing to projected effects from climate change suggesting an increase in high ozone days for the Los Angeles area. Support for in-situ monitoring efforts is evident in our findings.

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Chapter 1: Introduction and Background

Poor air quality is a human health issue in nearly every major city in the United States (American Lung Association 2019). Emissions from motor vehicles, manufacturing, energy production, and even such sources as restaurants, dry cleaners, and landscaping equipment contribute to an atmospheric soup of aerosols, particulates, and gases that affect health and well-being. Rural areas do not escape air quality problems. Agricultural chemicals, animal production emissions, and tillage contribute to reduced air quality. Furthermore, air pollution is not a stationary phenomenon. It is routinely transported long distances, affecting communities far removed from the sources (Curtis et al. 2006). In fact, ozone, the focus of this report, is usually much more concentrated in areas upwind of sources because the atmospheric synthesis of ozone requires time to mix precursors and react with sunlight (Sharma et al. 2017).

Many air pollutants have serious effects on human health. For example, data collected beginning in the late 20th century have shown that fine particulates damage lung tissue and lead to elevated rates of asthma and other pulmonary diseases, as well as heart disease (Heinzerling et al. 2015, Shaughnessy et al. 2015). Volatile organic compounds (VOC) include chemicals such as benzene, formaldehyde, and acetaldehyde that are produced by many industrial and combustion processes (Stroud et al. 2018). They are also emitted from less expected sources such as barbeques, dry cleaning establishments, and restaurant grills. In health science, the VOCs of most concern are indoor pollutants emitted by paints, carpets, and upholstery; their health effects run the gamut from minor eye and skin irritation to serious internal organ damage (Sarigiannis et al. 2011). Unlike most air pollutants of concern, only about 23 percent of VOCs are synthetically produced. The remaining 77 percent are emitted by plants (mostly as isoprenes), soil microbes, and vegetation fires (US EPA 2015a). VOCs are also a critical component in the atmospheric synthesis of ozone. The other air pollutants that play a critical role in ozone synthesis are nitrogen oxides (nitric oxide [NO], nitrogen dioxide [NO_2], and nitrous oxide [N_2O]) frequently grouped and abbreviated as NO_x. Small amounts of N_2O are emitted by soil organisms in a process called denitrification, and lightning is responsible for producing about 10 percent of the total NO_x globally (Schumann and Huntrieser 2007), but the majority of NO_x is produced by fossil fuel combustion. Nitrogen oxides by themselves are not generally harmful to human health at ambient levels except for people with preexisting asthma (WHO 2000).

In this report, we focus on ozone for several reasons. Ozone is a direct threat to human health. It is often indicative of the presence of other air pollutants; it is one of the few air pollutants that are routinely monitored at local, regional, and continental scales; and it can be measured using relatively inexpensive passive monitoring techniques.

Ozone is not emitted from any source at ground level. It is a secondary pollutant synthesized in the atmosphere in the presence of sunlight from precursors emitted by both human activity and natural processes. Whereas stratospheric ozone (between 15 and 48 km [6 and 30 mi] above the Earth's surface) is critical to protecting life on Earth from ultraviolet radiation, ozone in the troposphere (the atmospheric layer closest to the Earth's surface) is deleterious to all organisms (US EPA 2017).

The synthesis of ozone requires three components: nitrogen oxides, which come primarily from fossil fuel combustion and therefore are nearly all anthropogenic in origin; volatile organic carbon compounds, which may be natural or anthropogenic; and energy from the sun (fig. 1.1). Ozone synthesis is not a static or stationary process. The chemical reactions that produce it occur in a dynamic atmosphere; both the precursors and the resulting ozone move with prevailing winds. These winds frequently move air pollution away from its sources into more pristine areas, resulting in the well-described phenomenon of source areas being lower in ozone concentrations than outlying regions (Bytnerowicz 2005, Musselman and Korfmacher 2014).

After sunset, the synthesis cycle halts, and atmospheric concentrations of ozone begin to decrease through chemical and physical degradation. There are several pathways for ozone degradation. In urban areas where NO_x emissions tend to occur continuously, the presence of NO_x reverses the synthesis cycle, reducing ozone to elemental oxygen (O_2) (see fig. 1.1). Secondly, ozone is highly reactive. When it comes in contact with surfaces, including dust particles, it can chemically attack compounds on the surface; through reduction/oxidation reactions, ozone is reduced to O_2 while the reactive substrate is oxidized. Nighttime concentrations of ozone in polluted, low-elevation urban areas usually decrease to background levels of approximately 35 parts per billion (ppb) (Vingarzan 2004). In contrast, in surrounding mountains where the ambient air is cleaner with fewer dust particles and lower NO_x concentrations, ozone concentrations often remain at higher than background levels throughout the night (Musselman and Korfmacher 2014).

Modeling and predicting ozone concentrations at specific locations for specific periods are challenging. Precursor concentrations and weather conditions largely control synthesis rates, but wind patterns, local topography, and urban structure affect distribution, degradation, and deposition at any given location over time. Although the general pathway for ozone synthesis is known (fig. 1.1), the details of its atmospheric chemistry are still not completely understood, causing discrepancies between modeled and measured concentrations. Therefore, although modeled predictions of ozone concentrations are critical to planning for future impacts, onsite monitoring of ambient conditions is essential to understanding the effects of ozone on human health.

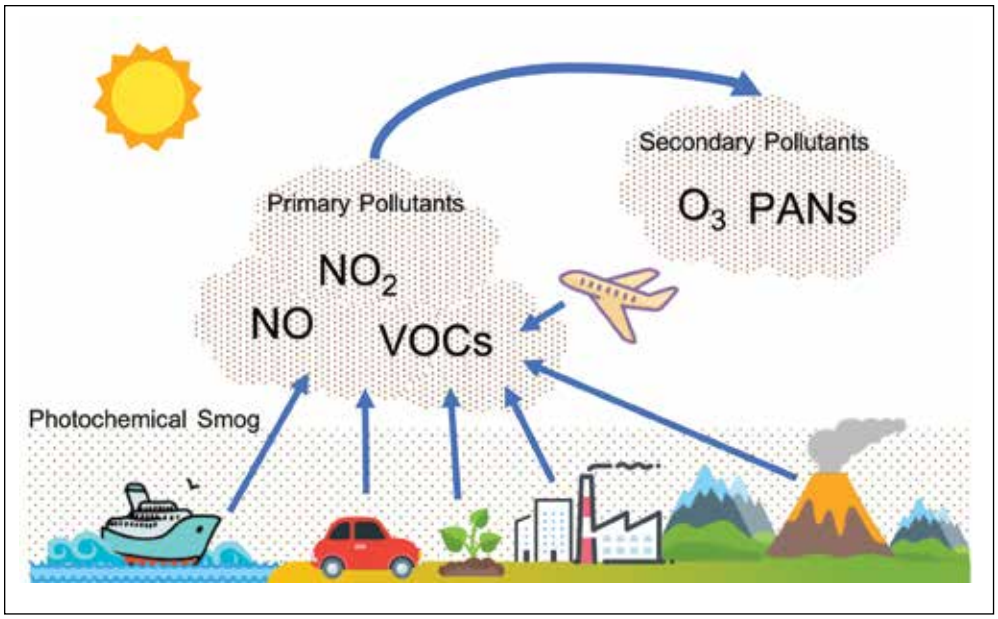

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Figure 1.1—Atmospheric synthesis of ozone and related air pollutants. Volatile organic carbon and nitrogen oxides are the main precursors of ozone. Volatile organic carbon refers to a suite of compounds that are produced naturally by plants and from human activity such as combustion. Nitrogen oxides are mostly the product of fossil fuel combustion. Once in the atmosphere, the energy from sunlight drives a complicated series of chemical reactions, forming ozone.

Effects of Ozone

Ozone is a powerful oxidant and is known to cause injury to organisms by attacking cell membranes. The exact mechanics of cell membrane damage are not well understood. In plants, ozone is particularly harmful to chloroplasts, the organelle responsible for photosynthesis in leaves thereby reducing net primary productivity and often causing premature abscission of leaves (Flowers et al. 2007). This loss reduces foliar cover and is responsible for up to 30 percent yield losses from staple crops such as wheat and rice (Debaje 2014). Damage to crops from ozone is a serious global concern as the world becomes both more urbanized and more populated, requiring an ever increasing supply of food (Avnery et al. 2013). The effects of ozone on trees in natural ecosystems have been well studied. Particularly in the Western United States, ozone can exacerbate drought conditions, making trees more susceptible to insect and pathogen attack (Karnosky et al. 2007).

In humans, ozone attacks lung tissue, reducing respiratory capacity, which can aggravate preexisting pulmonary diseases, and can lead to premature death (fig. 1.2). Increases in numbers of hospital visits and admissions in urban centers for asthma and other pulmonary ailments are well correlated with elevated ozone levels, particularly in sensitive populations such as young children and the elderly (Curtis et al. 2006, Pride et al. 2015). Chronic ozone exposure is also correlated

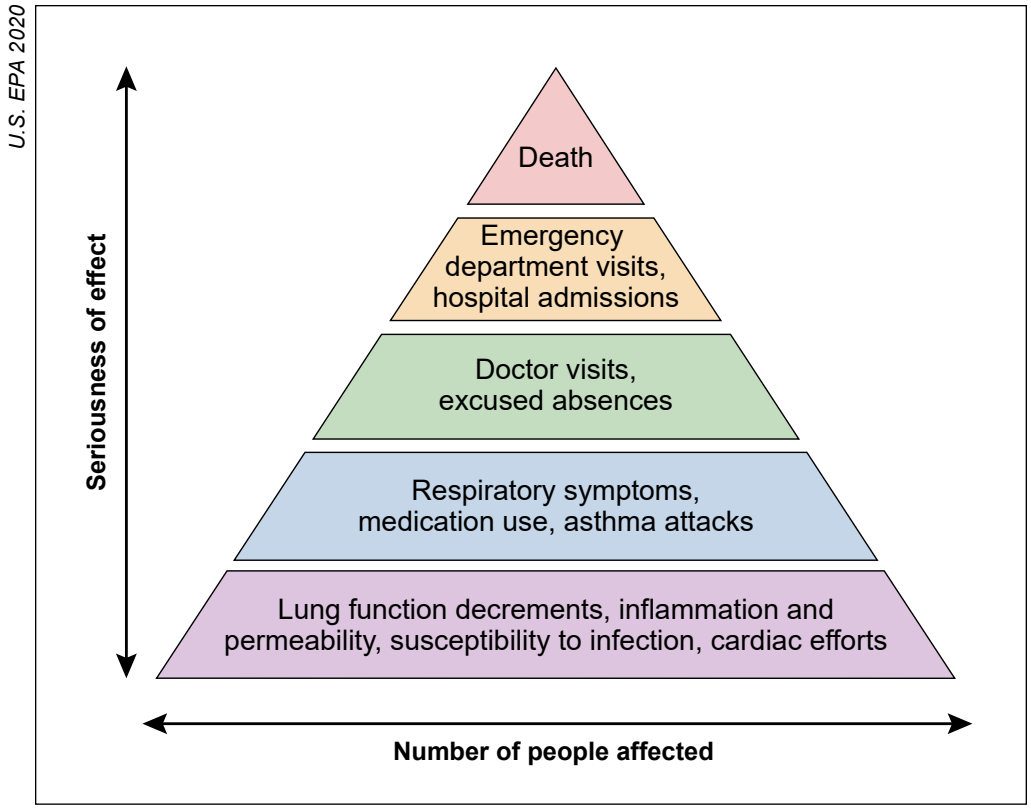

Figure 1.2—Severity of human health effects resulting from ozone exposure.

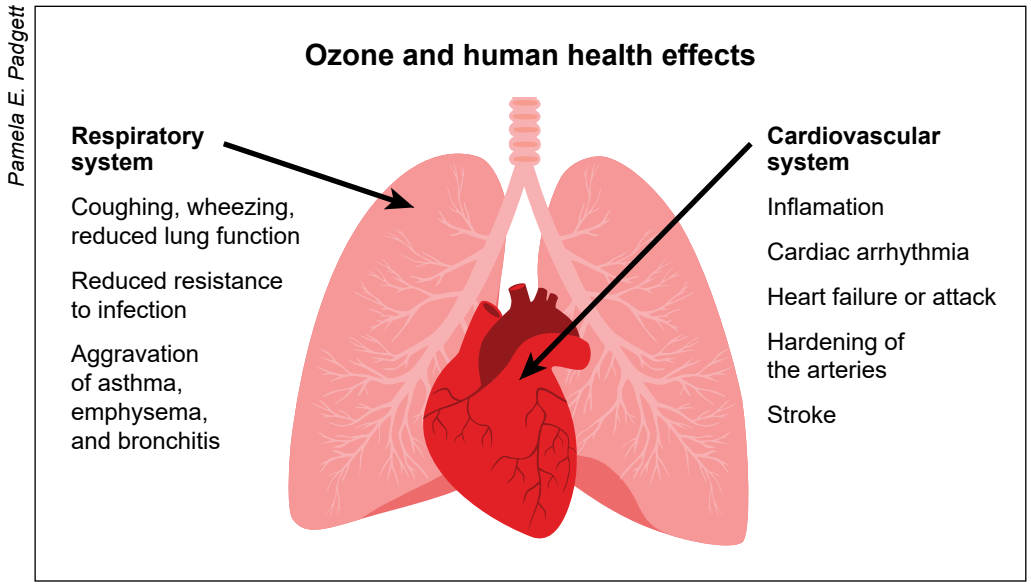

Figure 1.3—Ozone primarily attacks the respiratory system, leading to diseases of the lungs and heart.

with shorter lifespan and reduced quality of life (Curtis et al. 2006). Direct medical evidence has demonstrated that chronic exposure contributes to serious reductions in lung function and high occurrence of incapacitating lung and cardiac disease, among other health impacts (fig. 1.3). A report by the Union of Concerned Scientists (Perera and Sanford 2011) calculated the health and economic effects of increased ozone resulting from increased temperatures. It predicted an average of 2.8 million more occurrences of acute respiratory symptoms such as asthma attacks, shortness of breath, coughing, wheezing, and chest tightness at treatment and loss of productivity costs of $5.4 billion by 2020 (Perera and Sanford 2011).

Measuring Ozone

Many urban areas have set up monitoring networks to track ozone concentrations over time and provide daily predictions for health advisories. These efforts have largely been in response to U.S. Environmental Protection Agency (U.S. EPA) rules for reduction of ambient ozone pollution in compliance with national standards (US EPA 2015b). Data from these monitoring stations are available at the U.S. EPA website. In California, emissions from mobile sources (automobiles) are regulated by the California Air Resources Board, a department within the California Environmental Protection Agency, while emissions from stationary sources such as industry, oil refineries, and power plants are monitored and regulated by local air pollution control districts and air quality management districts. Local air pollution control districts are independent entities funded through fees, fines, and grants. Los Angeles (county and city) falls in the South Coast Air Quality Management District (AQMD), which covers Orange County, San Diego County, and the non-desert areas of Riverside and San Bernardino Counties. The South Coast AQMD operates about 40 air quality monitoring stations; it uses the data for air quality predictions and alerts published daily, and to determine long-term trends in air pollution.

Overall, the trend in ozone concentrations for southern California has been downward (fig. 1.4). Future trends, however, are uncertain, particularly without changes to emission standards (Fujita et al. 2016, Pollack et al. 2013, Rasmussen et al. 2013). Modeling efforts have attempted to predict the effects of climate change on various air pollutants. Horne and Dabdub (2016) argued that, in the Los Angeles Basin, areas with poor air quality will see more serious declines in air quality as temperatures increase, while locations with better air quality will see more modest increases in key pollutants, including ozone. With increased temperatures overall, the seasonal patterns of ozone are expected to shift. Ozone pollution is typically a summertime problem in southern California; projections suggest that elevated

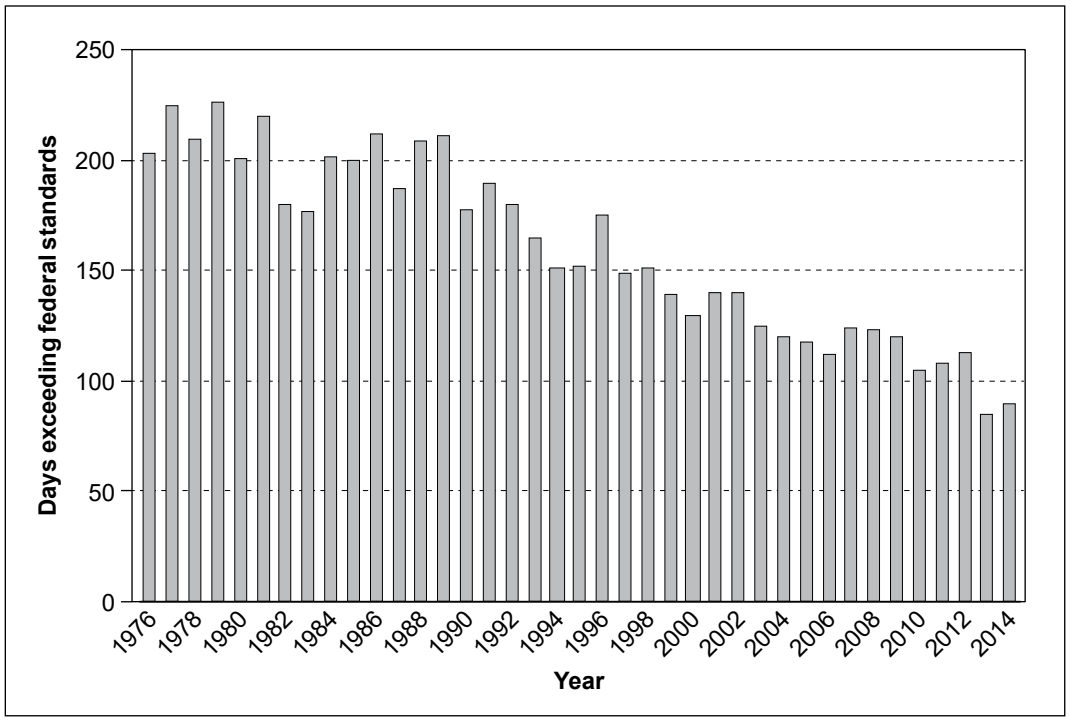

Figure 1.4—Ambient ozone concentration in southern California showing a 30-year decline; ppm = parts per million. Illustration adapted from South Coast Air Quality Management District (https://www.aqmd.gov/home/research/publications/50-years-of-progress).

temperatures associated with summer will begin earlier in the year, increasing the number of days associated with elevated ozone loads and increasing extremes in daily concentrations (Schnell et al. 2016).

Electronic instruments that provide real-time measurement of ozone levels are the "gold standard" for determining pollution levels; however, they are costly to purchase and maintain and they require electricity, temperature control, and secure locations. Therefore, it is typical for only one ozone monitoring instrument to represent a relatively broad area. Because microclimate has an important effect on ozone concentrations, measurements of ozone at specific locations away from active monitors can be substantially different (see chapter 1 in Makar et al. 2017).

"Passive" monitoring systems are small devices that require no electricity and are often deployed to monitor in areas in which real-time monitoring instruments are unavailable. Passive monitoring has been widely used in ecological studies to determine ozone distribution and landscape-level concentrations in forests and shrublands around the world (e.g., Panek et al. 2013). However, they have not been widely used to determine direct exposure of individuals to ambient ozone concentrations. A few studies have evaluated personal exposure of individuals to ozone, comparing findings across community types (urban versus rural). Two

studies in Turkey compared indoor, outdoor, and personal exposure to ozone, among other pollutants. In the first study, Demirel et al. (2014) used passive samplers to measure benzene, toluene, ethyl benzene, xylenes, NO_2, and ozone levels during a 24-hour period. The study compared an urban area to its outlying rural area; the goal was to investigate exposure of school children to these compounds at home and at school. Although the methods indicated that the children were wearing the samplers at the same time the ambient conditions were being monitored, no details were provided as to how the personal samplers were deployed. The results, however, showed that rural children were at greater risk from ozone exposure than urban children. The second study in Turkey (Bozkurt et al. 2015) evaluated exposure to NO_2, sulfur dioxide (SO_2), and ozone in a more extensive investigation using both passive samplers and active monitors. As in the study by Demirel et al. (2014), Bozkurt et al. (2015) provided no details regarding the personal monitoring techniques used. Their data also showed that rural populations were exposed to higher ozone levels than groups in urban and industrial centers.

A study published in 2000 (Geyh et al. 2000) most closely resembles the approach taken in the series of studies we described in this report. The study was conducted in three locations: the city of Upland, a highly urban area on the eastern edge of the Los Angeles Basin; and two rural mountain communities, Crestline and Running Springs, which respectively are 37 km (23 mi) and 50 km (31 mi) northeast of Upland and 1036 m (3,400 ft) and 1494 m (4,900 ft) higher in elevation. Geyh et al. (2000) used the same model of Ogawa passive sampler used in our project and followed a similar protocol for their deployment and extraction. The primary difference in their study was that samplers were worn almost continuously for 6 days (adjustments were made for sleeping, bathing, etc.), with sampling periods distributed over a full year. As in the studies conducted in Turkey, both rural mountain communities experienced higher ozone exposures. Their ozone levels were 20 percent higher than the urban community in summer months, and 60 percent higher in winter months (Geyh et al. 2000).

Ozone Concentrations in the Los Angeles Basin

Smog and ozone have been long-standing environmental problems in Los Angeles (Fujita et al. 2016). Historically, Los Angeles and the greater South Coast Air District have experienced the most severe ozone pollution in the Nation. In the 1960s, summer peak ozone concentrations exceeded 500 ppb relatively frequently. Regulatory controls of VOC and NO_x emissions have dramatically reduced ambient ozone concentrations and the number of days when the current US EPA standard of 70 ppb (average exposure over an 8-hour period) is exceeded has declined.

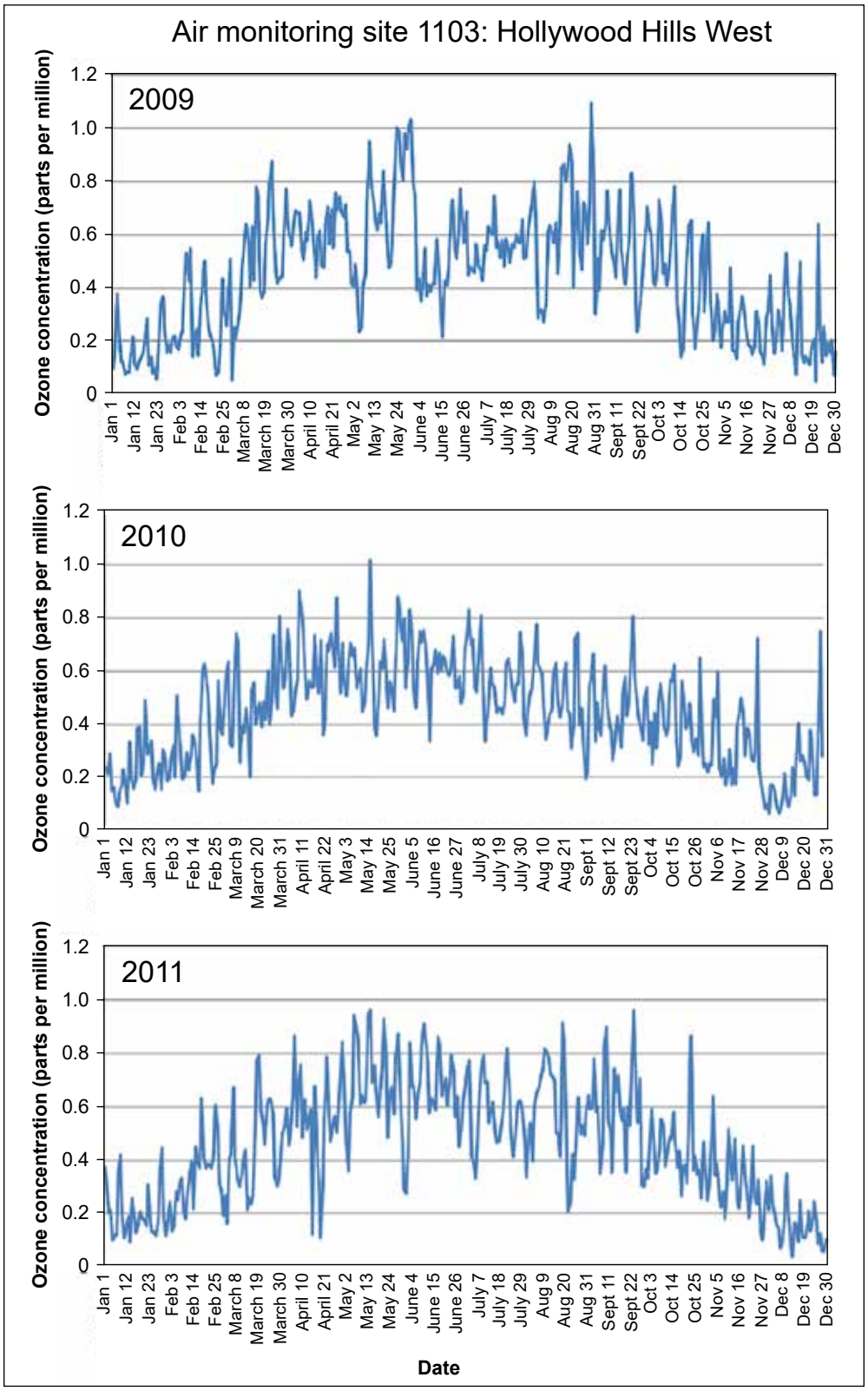

Figure 1.5—Annual variation in ozone concentrations measured at the Hollywood Hills West monitoring station. Each chart shows ozone concentrations starting January 1 and ending December 31 for the years 2009 through 2013. Note the interannual variation; for example, in June and July 2009, ozone levels were comparatively low, and in 2013, several high ozone events occurred in January and February.

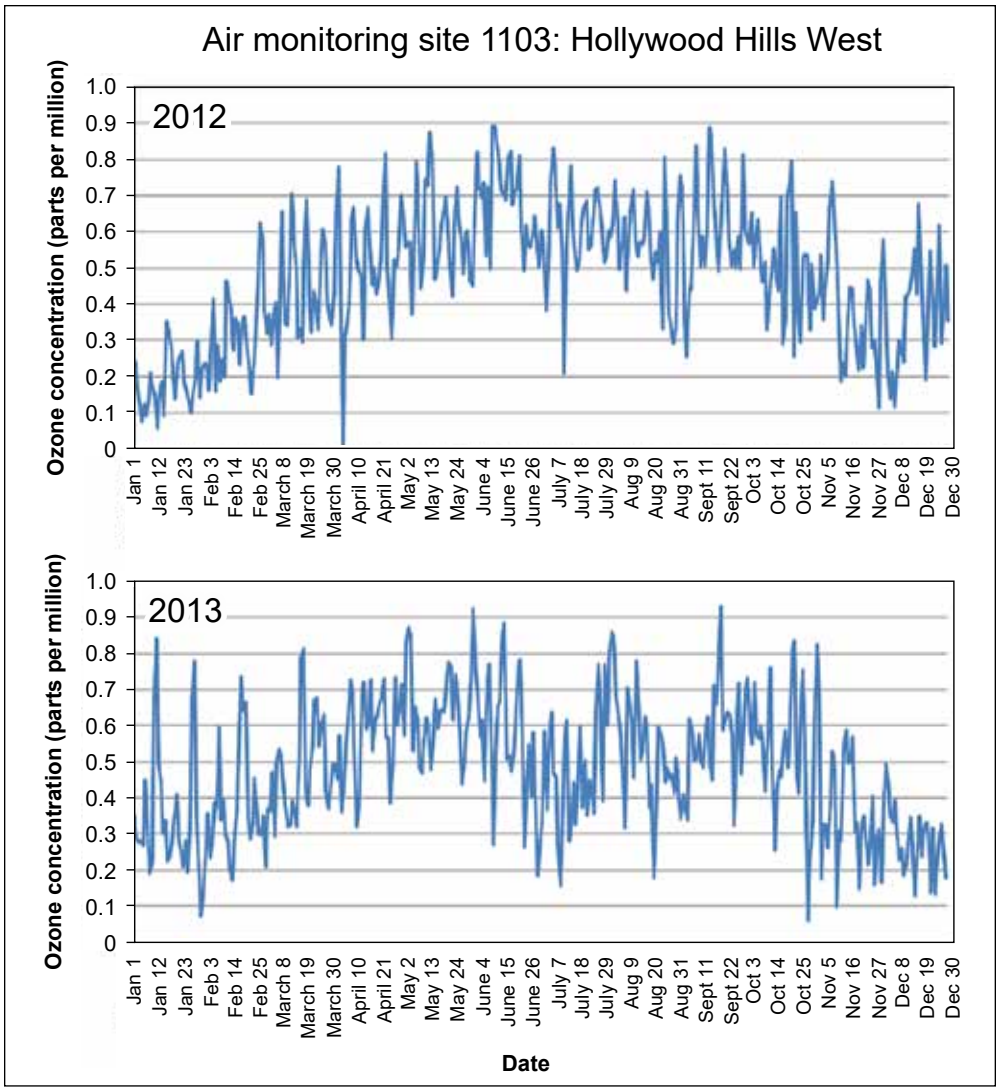

Figure 1.5—continued

However, attaining long-term reductions in emissions, and thus ozone, is expected to be particularly challenging in light of projected increases in ozone associated with climate change (Fujita et al. 2016).

Ozone concentrations, as described earlier, vary widely from day to day and from year to year. Prior to initiating experiments 1 and 2 (see chapter 2), we evaluated annual patterns in ozone using electronic monitoring data from permanent air quality monitoring stations located around the Los Angeles Basin. Figures 1.5 and 1.6 display ozone data from the Hollywood Hills West and Sun Valley monitoring stations, respectively, for 2009 through 2013. In each of the 5 years shown (figs. 1.5 and 1.6), ozone concentrations in summer months, May through September, are generally higher than in winter months. However, in some years (such as 2009) there were periods of lower than expected ozone concentrations during summer; 2013 also had a low ozone period from mid-June

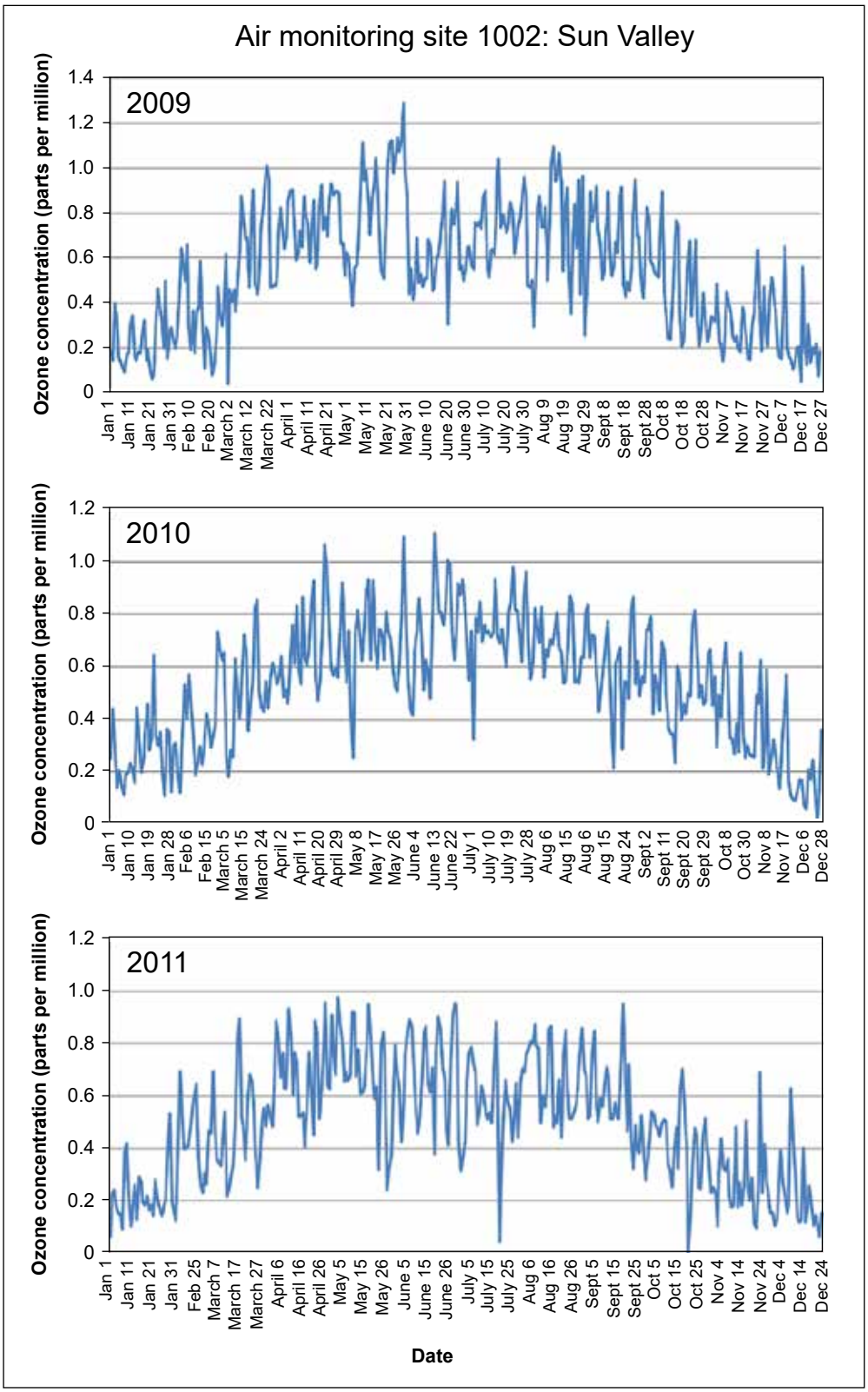

Figure 1.6—Annual variation in ozone concentrations measured at the Sun Valley monitoring station. Each chart shows ozone concentrations starting January 1 and ending December 31 for the years 2009 through 2013. Note the series of low ozone days in June and July 2009, similar to those recorded in Hollywood Hills West, although the Sun Valley monitoring stations did not record the relatively high concentrations measured in January and February 2013 by the Hollywood Hills West monitor.

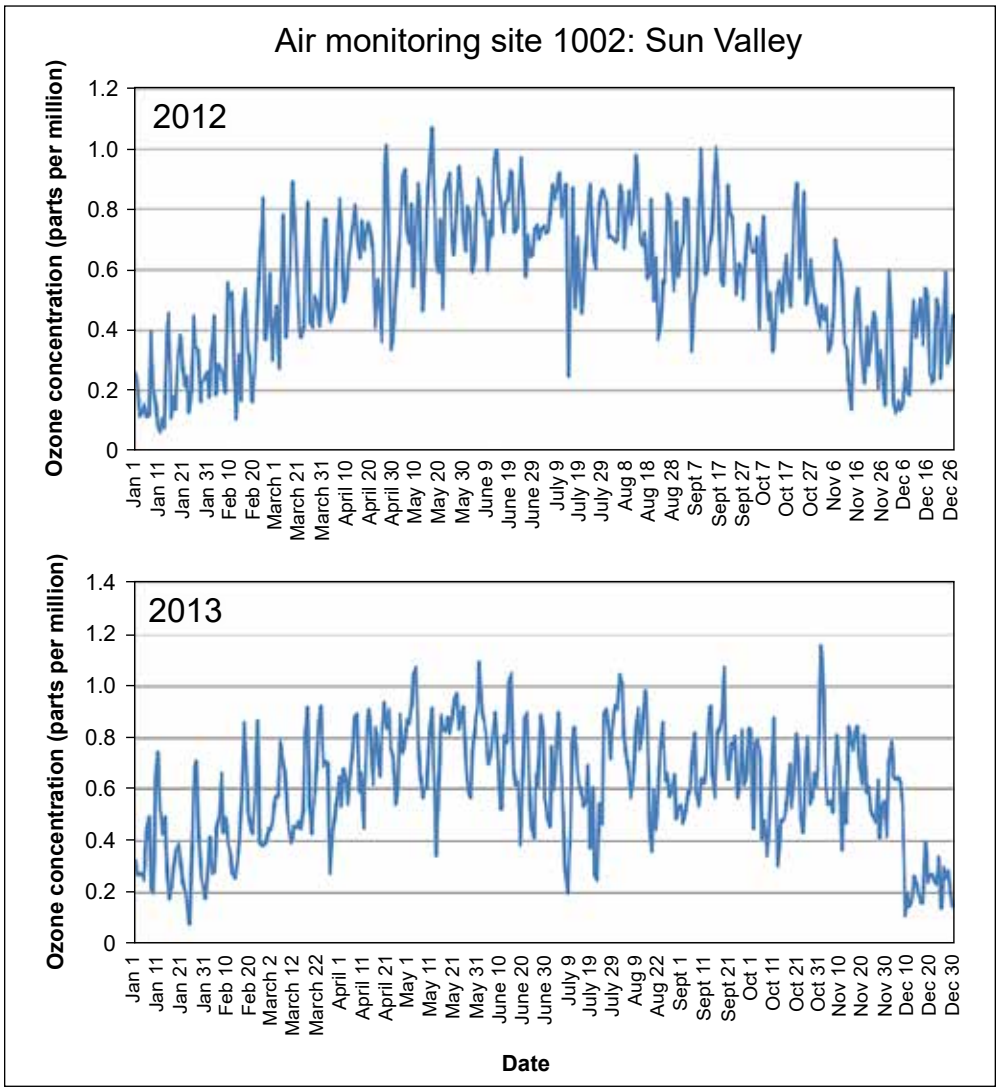

Figure 1.6—continued

until the end of July. Both 2010 and 2013 had episodes of high ozone concentrations in December and January. In most years, August and September had consistently high ozone concentrations. We used these data to plan our field study of ozone levels in a number of Los Angeles parks.

Passive Sampler Techniques and Application

Passive samplers are devices that use a chemical reaction to a pollutant to determine ambient concentrations. In an ozone sampler, that reaction is the conversion of nitrite (NO_2^-) to nitrate (NO_3^-). The Ogawa™ passive ozone sampler[1] is one of the

[1] The use of trade or firm names in this publication is for reader information and does not imply endorsement by the U.S. Department of Agriculture of any product or service.

most commonly used devices for air quality sampling (Ogawa USA, 1230 SE 7th Avenue, Pompano Beach, FL 33060).

Ogawa passive samplers are inserted into a "badge" (fig. 1.7) that can be pinned or clipped to clothing (Ogawa and Co. 2001). Each sampler contains two paper filters (collection pads) impregnated with HNO_2. The filters are placed between two stainless steel mesh screens and inserted into a Teflon body capped with a Teflon diffuser. The mesh screens and the Teflon diffuser function as a barrier, allowing ambient air to slowly diffuse across the barrier to the paper filters, rather than blowing the ambient air directly on the filters. When ozone contacts the filters, the HNO_2 is oxidized to HNO_3. The conversion of HNO_2 to HNO_3 is directly proportional to the amount of ozone contacting the filters. After exposure, the sampler is disassembled and the filters are transferred to a small plastic bottle for extraction of HNO_3. To extract the filters, 5 ml of ultrapure water are added to the bottle, which is shaken for 30 minutes to dissolve the accumulated HNO_3. The two filters can either be extracted together, or separately. In the experiments described in this report, each filter was extracted separately. The quantity of dissolved HNO_3 is measured using a Dionex™ ion chromatograph. The quantity of ozone that

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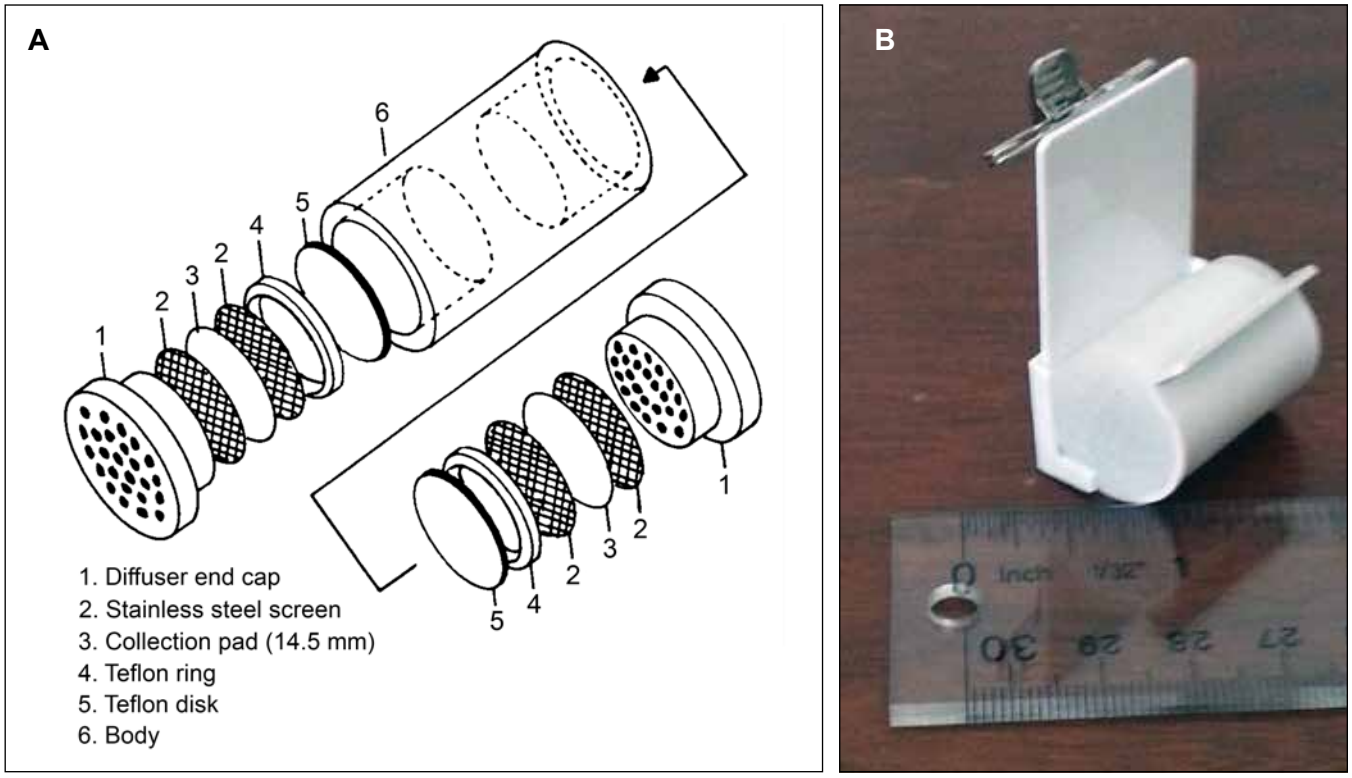


Figure 1.7—Ogawa ozone passive monitor. (A) Detailed view of the passive monitoring device adapted from Ogawa USA. (B) Passive device inserted into the badge, which is then attached to the clothing of the observer.

contacted the filters can be calculated by standard equations using the HNO_3 concentration measured and the amount of time the samplers were deployed (equation 1).

$$[O_3]\ ppm = \frac{[NO_3]\ \mu g ml^{-1}] \times extract\ vol\ (ml)}{21.8\ ml\ O_3 / time\ (min)} \times \frac{1\,\mu mol\ NO_3}{62\ \mu g\ NO_3} \times \frac{1\,\mu mol\ O_3}{1\,\mu mol\ NO_3}$$

$$\times \frac{24.45\ \mu L\ O_3}{1\,\mu mol\ O_3} \times \frac{10^{-6}\ m^3\ O_3}{1000\,\mu L\ O_3}$$

In equation 1, [NO_3 µgml^{-1}] is the concentration of nitrate in the extraction solution in micrograms per milliliter, and the extraction volume is the total amount of water used to extract the filters; time is the total length of time the samplers were exposed, in minutes; 62 µg is the molecular weight of nitrate, and one molecule of nitrate is produced by one molecule of ozone. One micromole of ozone occupies 24.45 µl in volume. The final conversion is the number of ozone molecules in a cubic meter, which is given as parts per million.

All chemical analysis was conducted in the chemistry laboratories at the Pacific Southwest Research Station facility in Riverside, California. However, a number of commercial laboratories are capable of extracting and analyzing passive samplers. Ogawa USA maintains a contract laboratory that will provide analytical results and final ozone concentration calculations.

Note that passive samplers measure cumulative ozone, representing the total amount of ozone encountered by the filters over time. When exposure time is factored in, this calculation yields average hourly ozone concentration but does not reveal diurnal changes in concentrations that are typical over the course of a day. Electronic instruments generally yield hourly data, which enable visualization of the typical pattern of ozone concentration: from low levels of ozone during night and early morning hours to increasing levels during daytime hours as sunlight and temperatures rise, and returning to decreasing concentrations during evening hours. To compare the two methods, hourly concentrations from active monitors can be averaged over the period of interest and compared to an hourly average calculated from the cumulative exposure in the passive monitors.

Chapter 2: Development of Techniques for Monitoring Ozone Exposure in Humans

This report summarizes the testing and evaluation that went into using passive samplers to estimate human exposure to ozone during recreational activities. Several variables specific to the methods and techniques were evaluated in experiments 1 and 2, and the potential for differentiating ozone exposure to individuals in different areas of Los Angeles was explored in experiments 2 and 3 (table 2.1).

Table 2.1—Summary of changes made between experiments 1, 2, and 3

Experiment	Purpose	Location	Modifications to protocols
1	Proof of concept	San Antonio Canyon, Angeles National Forest	
2	Measure O_3 exposures to visitors in four Los Angeles parks with different socioeconomic structures	Sun Valley, Brentwood, City of Los Angeles	1. Added second monitoring badge per observer 2. Created more detailed deployment procedures 3. Trained the observers directly 4. Added a travel blank to each field day
3	Refine the study to eliminate geographic bias and allow direct comparisons of daily O_3 exposures across the socioeconomic strata	Sun Valley, Brentwood, Wilmington, Hollywood Hills West, City of Los Angeles	1. Moved Ogawa badges from the torso to the head 2. Paired one park in an affluent community with one park in a disadvantaged community on every observation day during the same period 3. Added two additional parks in two new communities to balance inland and coastal effects. 4. Changed exposure times from 5 to 6 hours 5. Changed to two observation time bands—one in the morning and one in the afternoon

Experiment 1

In experiment 1, the goal was to determine if samplers were sufficiently sensitive to measure ozone exposure in terms of hours rather than days, as was customary in most prior studies including the Geyh et al. (2000) study. Further, variability across samplers deployed in the same area at the same time was evaluated. Low variability between the two filters in a single sampler is desirable, and variability of less than 20 percent across individual badges exposed to similar environments was viewed as a realistic goal for this study.

Results from the passive samplers were compared to data from the electronic active monitors in the region. High levels of ozone recorded by the electronic instruments would be expected to align with higher ozone measurements as determined by the passive monitors. Field trials of passive systems have been deployed immediately adjacent to active monitoring stations. In those studies,

the Ogawa passive ozone systems very closely reflected the cumulative exposures calculated from the active monitoring systems (Alonso et al. 2002).

Criteria to assess applicability and reliability were mostly met in experiment 1. However, higher than desired variability across individual badges was discovered. This led to stricter protocols for handling the badges before, during, and after deployment, including the addition of a second badge per observer and modification of badge placement location on each member of the field team in subsequent experiments.

Experiment 2

Experiment 2 was focused on discerning measurable differences between two communities in Los Angeles. One community was in an affluent area near the Pacific coast and the other was in a disadvantaged area farther inland. The disadvantaged inland community, compared to the affluent community, had more than four times the population density per square mile, less than half of the percentage of tree canopy cover, residents had less than half the median household income, and the community was majority Latino. This effort was part of a program of research examining paired social and ecological vulnerability and resilience of urban communities using public parks as the nexus for exploring climate change effects (Winter et al. 2019).

Within each area, two urban public parks were chosen as study sites. Although the parks were in the same overall airshed, physiographic and local pollution sources created differences in diurnal patterns of ozone concentrations as determined by electronic monitors. Pairs of observers wore two badges each for a 5-hour period falling between 9 a.m. and 4 p.m. while walking a predetermined path through the parks at specified intervals. Between these intervals, the observers remained in the parks, out doors, with the badges exposed to the ambient environment. Pairs of observers visited the two parks in the two communities on different days.

At first, there was no apparent pattern of differences in ozone concentrations between the two communities. However, active data from nearby electronic monitors showed that ambient concentrations differed widely from day to day. We concluded that to accurately compare ozone exposure concentrations between the two communities, observers needed to be deployed in each community on the same days at the same time. However, the passive data, when compared to data from electronic active monitors, were remarkably close to values reported by the active monitors. There were a few exceptions, which suggested modification of badge placement on the observers. Placement on the observer's torso, front and back,

may have influenced exposure by blocking air exposure during the rest period or while the observer was taking notes. It is likely that badges were then shielded from ambient conditions, or were exposed to contamination from contact with the backs of benches or other surfaces while observers were sitting.

Experiment 3

Four protocol modifications were made for experiment 3 (table 2.1): (1) Badges were placed on a hat rather than on the observers torso, which improved uniformity of exposure. (2) Communities being compared (an affluent and a disadvantaged community) were visited by a pair of observers on the same day, at the same time. (3) To separate the effects of topography from socioeconomic variations, four additional parks were added to the study: two parks in an affluent inland community and two parks from a disadvantaged community on the coast. As a result, experiment 3 included two parks in an affluent coastal community, two parks in an affluent inland community, two parks in a disadvantaged coastal community, and two parks in a disadvantaged inland community. These affluent and disadvantaged communities varied markedly on environmental, social, and economic criteria similar to experiment 2 (see Winter et al. 2019). (4) Each observation period was extended from 5 hours to 6 hours. (5) Instead of having a single observation period in the middle of the day, a morning observation period (7 a.m. to 1 p.m.) and an afternoon observation period (1 p.m. to 7 p.m.) were established to capture a greater range of park use and environmental conditions.

Chapters 3 through 5 present the details of each of the three experiments, along with the resulting findings.

Chapter 3: Experiment 1—Proof of Concept

Introduction

Ozone exposure is a ubiquitous problem in southern California, including in its surrounding rural areas and wildlands. Although the negative consequences of ozone exposure are often considered within an urban context, the environmental and ecological effects of ozone were first discovered in the San Bernardino Mountains east of the greater Los Angeles Basin. The highest ever recorded atmospheric ozone concentrations in the country were in the mountain community of Crestline, in San Bernardino County, a community of approximately 11,000 people at an elevation of 1400 m (4,600 ft) (Lee et al. 2003).

Concern about exposure to high levels of ambient ozone in rural areas and wildlands is widespread. For several years, the National Park Service (USDI NPS 2018) has maintained Web-based information on air quality for park visitors. Some parks such as Rocky Mountain National Park and Great Smoky Mountains National Park have webcams that enable virtual observation of air quality. Rural and wilderness areas provide many recreational opportunities and draw millions of visitors every year. As part of the outdoor nature-based experience, visitors expect to enjoy clean air, an unimpaired view, and health benefits from spending time in an unpolluted environment. For public land managers, there is a need to know how air quality affects recreationists.

The use of electronic instruments by the U.S. EPA, California's Air Quality Monitoring Department, and others are a helpful guide; however, they do not necessarily provide data showing the exposure level of individuals engaged in outdoor activities. Local geography and vegetation, physical exertion, and health status, as well as ambient weather conditions, can contribute to the concentration and impact of air pollutants on outdoor recreationists (Geyh et al. 2000, Sharma et al. 2017). Most people spend most of their time indoors at home, work, or school. In many modern buildings, ozone concentrations are near zero because heating and air conditioning equipment is very effective in removing ozone from ambient air (Weschler 2000), although some office equipment and ionizing air cleaners (which may produce ozone as a byproduct or a sanitation device) can increase indoor concentrations. Yet, exposure to outdoor air pollution is inequitably distributed, and the interaction with additional factors elevates concerns for disadvantaged communities (O'Neill et al. 2003). Recreationists, people without fixed residences, or people without air conditioning systems in their homes are exposed to ambient conditions for lengthy periods of time, which may have deleterious effects on their health. Earlier studies focused on prolonged exposures in terms of weeks or months. This work was intended to focus on exposures during a recreational outing during a single day.

In this study we were testing the feasibility of measuring personal ozone exposure experienced by individuals engaged in outdoor activities over a short period of time.

Approach and Methods

Ogawa passive monitors have been used in field applications to monitor ozone (and other air pollutant) concentrations in remote areas for more than a decade (Alonso et al. 2002, Bytnerowicz 2005). The results of these studies have revealed poor air quality in wilderness areas, national forests, and national parks where air quality was once thought to be excellent. Ogawa monitors have been used to establish gradients in air pollution: these gradients enable researchers to evaluate the effects of air pollution on vegetation and unique habitat types through a gradual increase in pollution loads. These studies deploy passive samplers for 2 or 3 weeks at a time. This study evaluated the sensitivity and accuracy of the Ogawa passive monitors for detecting ozone concentrations after only 5 hours.

To test the practicality of these particular passive samplers for short exposures, we partnered with an ongoing study of recreation use in the Angeles National Forest in southern California. Two individuals were stationed at the trailhead of San Antonio Canyon in the Angeles National Forest (fig. 3.1) over 9 days in July and August to survey forest visitors about recreational and fire management preferences. Each survey team member wore one Ogawa badge during the approximately 5-hour

Figure 3.1—San Antonio Canyon, Angeles National Forest, California.

survey period (n = 2 per individual, n = 4 per day). The survey team was trained in deployment of the passive monitors. Each team member was to remove a badge from the sealed container and internal plastic bag and attach the badge to their clothing while still in their vehicle. The exposure starting time was noted on the container. After the shift, each badge was returned to the plastic bag and resealed in the plastic jar, and the exposure ending time was noted. The investigators were paired so that two people wearing one badge each were involved in the collection of data from approximately 10 a.m. to 3 p.m. each day. Field teams avoided spurious contamination of the field samplers by following the established protocol (i.e., not smoking or approaching campfires, staying away from running automobiles, and not touching the collection ends of the badges).

On one of the field days, two badges were sent out with the survey team; these "field blanks" were carried by the team but not opened or exposed to the elements. Two unused "lab blank" filters that did not leave the laboratory were analyzed along with samples from the field. This approach provided a lab blank to test for background contamination inherent to the filters, and a field blank to test for contamination occurring during travel to and from the field location.

In sum, four filters (two per badge), totaling 36 collector pads, were exposed to the environment for a 5-hour duration on 9 days. Two lab blanks and four field blanks provided information on background concentrations owing to travel or storage. Shortly after field data collection, the sealed plastic containers housing the samples were returned to the laboratory, where they were held for analysis until all samples were onsite. The filters were extracted as described in chapter 1.

To compare onsite measurements from the passive samplers with local "official" ozone concentration measurements from electronic instruments, monitoring data for each of the 9 days of air quality monitoring conducted by the survey field team were acquired from the nearest air quality monitoring stations, Lake Gregory and Pomona, both are operated by the South Coast Air Quality Management District (AQMD). These data can be accessed from the South Coast AQMD website (http://www.aqmd.gov/home/air-quality/air-quality-studies/air-quality-monitoring-studies). The active monitoring sites and the passive monitoring study site are shown in figure 3.2, and the geographic details are in table 3.1.

Data and Results

To determine if 5 hours were sufficient to detect a signal under an expected moderate level of ambient ozone, the samples collected by the survey team were compared to both the field blanks and lab blanks. The results indicated that the Ogawa passive monitors worn by the survey team were adequately sensitive to

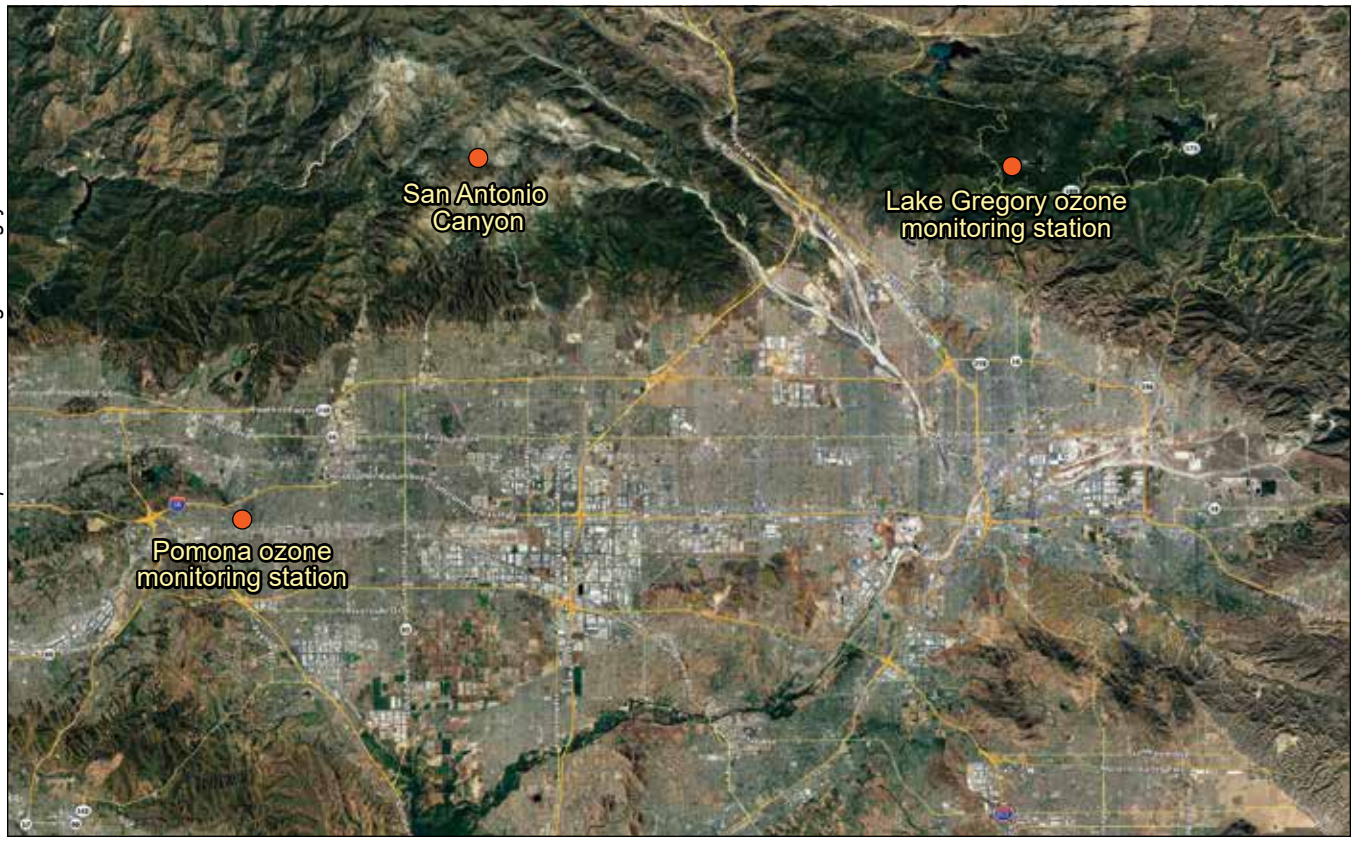


Figure 3.2—Locations of ozone monitoring sites relative to the study site in San Antonio Canyon. The Lake Gregory and Pomona ozone monitoring stations house electronic ozone monitoring instruments used for air pollution forecasting and regulatory compliance.

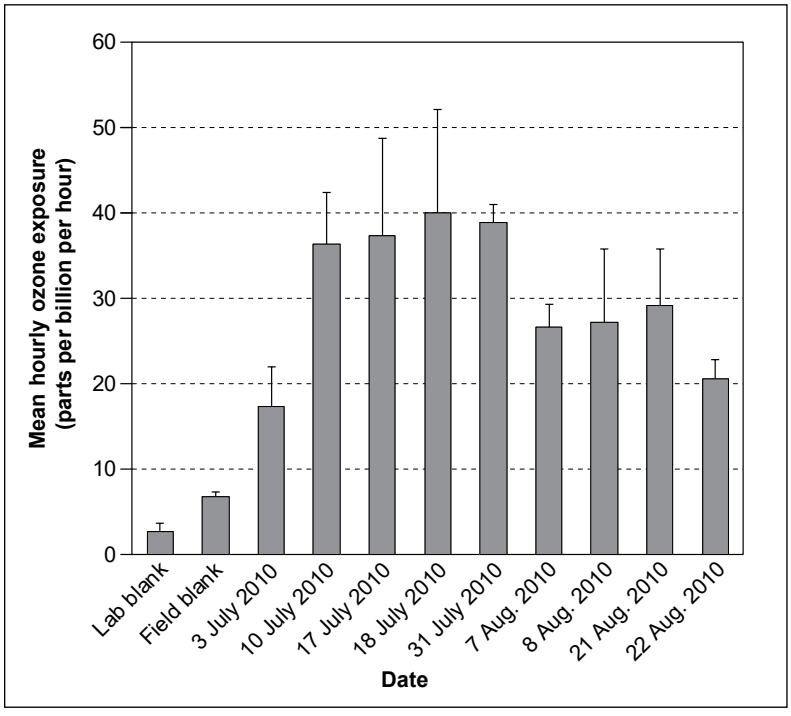


Figure 3.3—The average exposure to ozone experienced by two observers (n=4) on each of the nine days of the study. Lab blanks (n=2) and field blanks (n=4) were compared to the filters from onsite to evaluate the badges' level of sensitivity to ozone. Error bars = standard error of the mean; ppb/hr = parts per billion per hour.

Table 3.1—Locations of electronic ozone monitoring stations and the passive sampler test site

Site location	Latitude	Longitude	Elevation
			Meters
924 N Garey Avenue, Pomona, CA 91767	34.06698	-117.75138	273
Lake Gregory, 24171 Lake Drive, Crestline, CA 92625	34.24313	-117.27230	1390
San Antonio Canyon Trailhead	34.24167	-117.61167	1371

ambient ozone and were promising, but unacceptably high variability across sampling units required modifications in the method for reliable application. Single-factor ANOVA indicated that when both samplers worn by the two observers on each day (n = 4 total sample filters) were pooled as a single sample and compared to the four travel blank filters and the two laboratory blank filters, all of the exposed filters were significantly higher ($p = 0.03$) than the unexposed filters. However, when the blank filters were removed from the ANOVA, none of the calculated ozone concentrations differed from one another, statistically, during the field trials ($p = 0.29$) (fig. 3.3). In comparing these data to data from past environmental studies, it seemed that ozone concentrations calculated from badges worn by human observers had unusually high variability, suggesting that they were subject to some sort of systematic errors in deployment.

Each individual wore one passive monitoring sampler, and in each sampler two filters sitting at opposite ends of the badge—approximately 5 cm (2 inches) apart—were exposed to the ambient air. Each filter was extracted and analyzed separately in the laboratory. The results showed that there were frequently large variations between two filters worn by the same individual, as seen on 18 July 2010 (fig. 3.4), and between paired individuals on the same day, as seen on 8 August 2010. Environmental monitoring has shown that relying on a single passive monitor is problematic for this reason, because there is no way to statistically reconcile highly variable data (this was the lead author's experience; also see Bytnerowicz 2005). Furthermore, because we cannot rule out the possibility that the two individuals were, in fact, exposed to different ozone concentrations because of microsite conditions, both data points were accepted for each day for comparison with the active monitor data.

In these cases, we concluded that improper handling or contamination of the badges accounted for the wide variation on any one individual. For example, improper handling may have been as simple as blocked airflow to one end of the badge by a piece of clothing, or one end of the badge may have been inadvertently touched sometime during the day. From an interpretive standpoint, we cannot be certain which of the filters (if either) provides accurate data.

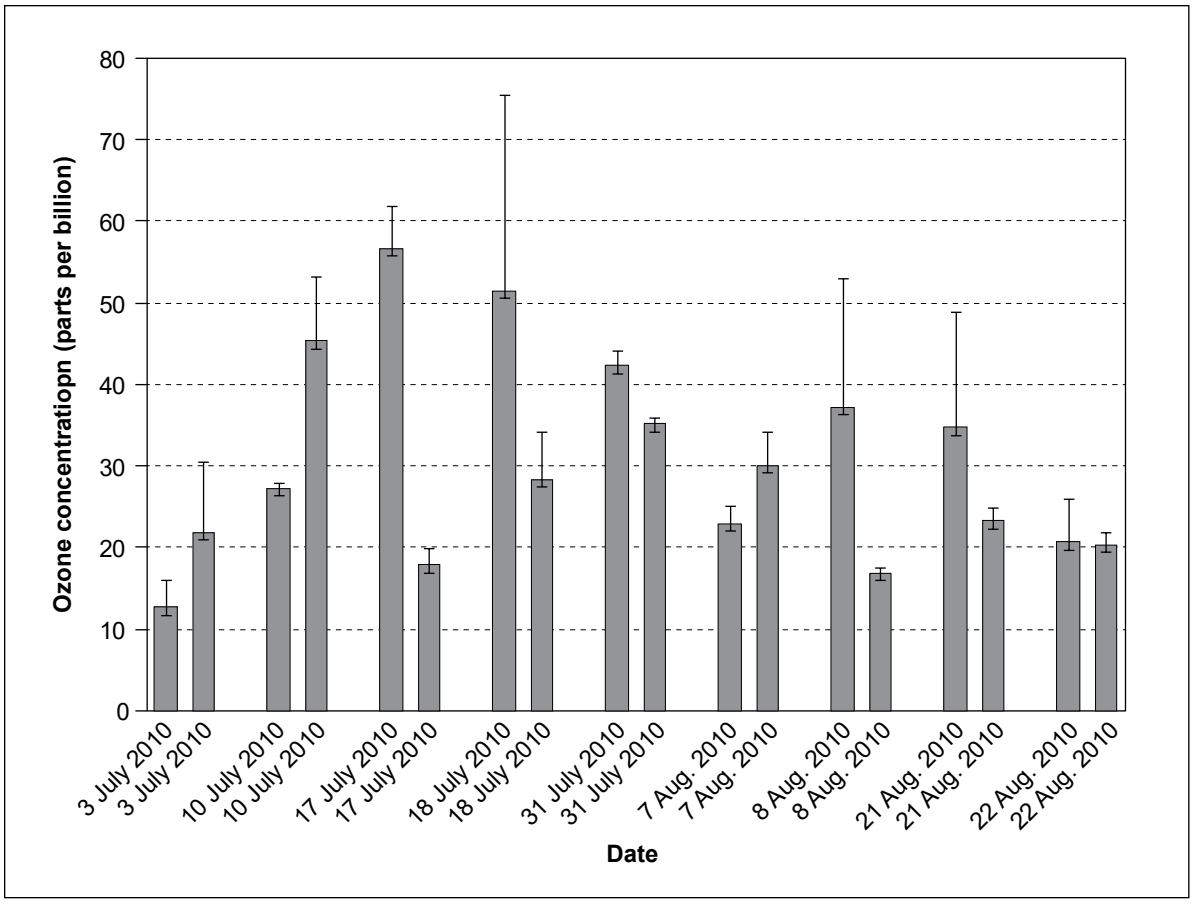


Figure 3.4—Average ozone exposure by individual (n=2). Error bars = standard error of the mean. ppb = parts per billion.

To compare the passive monitoring systems to the electronic real-time monitors, data from the two nearest monitoring stations were acquired for each day of the field trial. The monitoring stations differed from the study site in elevation and airshed, but when the data from both stations were compared (fig. 3.5), the diurnal trends and magnitude of the exposures were similar between the two stations. We concluded that both stations were acceptable for our needs.

Even with caveats and less than ideal circumstances, a close relationship was evident between ambient ozone concentrations measured by the stations and the amount of ozone indicated by the Ogawa passive monitors (fig. 3.6). The exposure values from the Ogawa samplers would be expected to be lower than ambient concentrations owing to reduced and restricted airflow around a solid object. In general, ambient monitors are placed in areas with open air fetch not impeded by manmade or natural objects (US EPA 2007), unlike a badge worn on a human body.

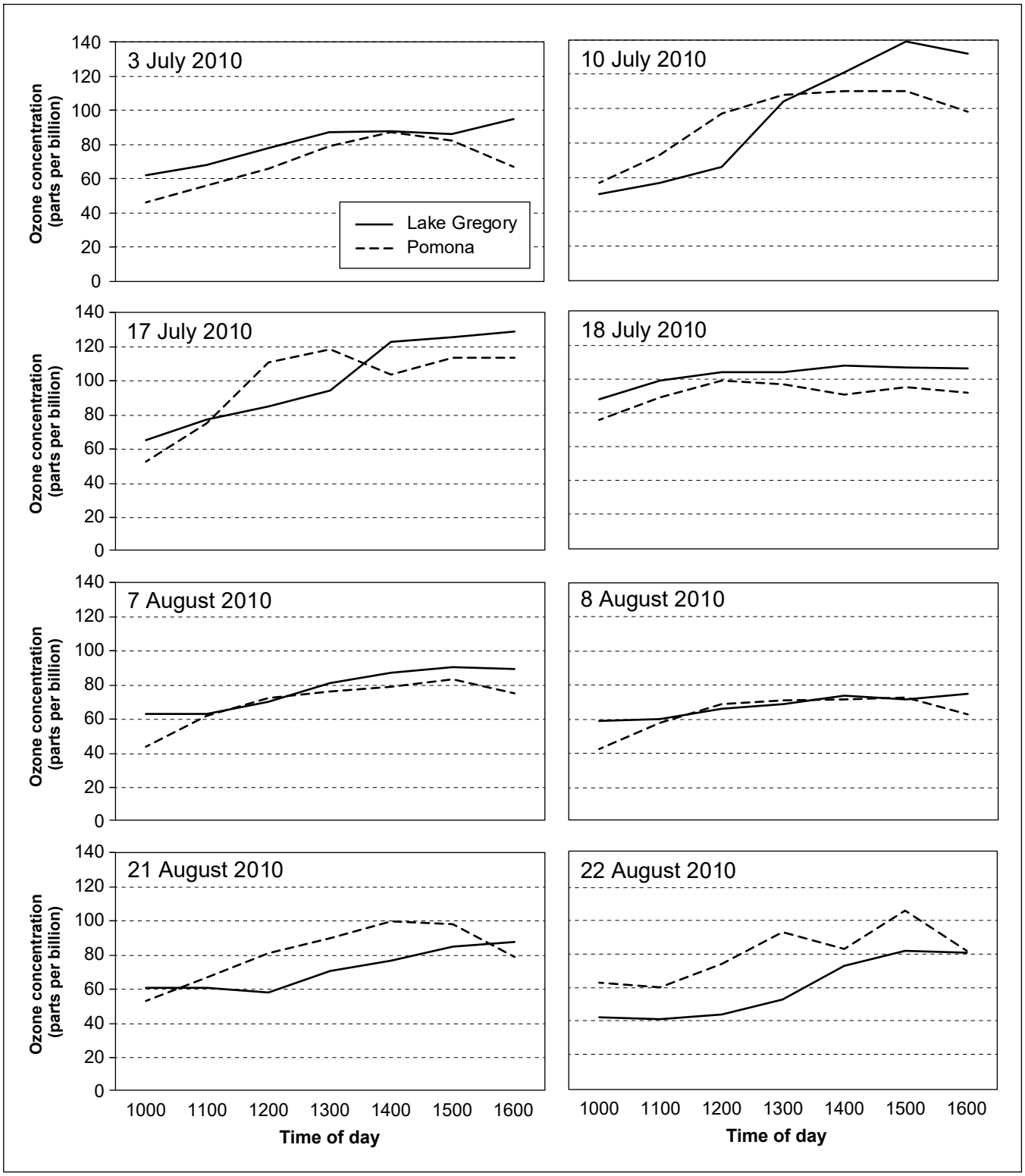


Figure 3.5—Ambient ozone concentrations measured at the two nearest active monitoring stations (see fig. 3.2). The Lake Gregory ozone monitoring station is closest in elevation to the passive monitoring site in San Antonio Canyon, but not in the same airshed. The Pomona ozone monitoring station is the same airshed, but at a lower elevation.

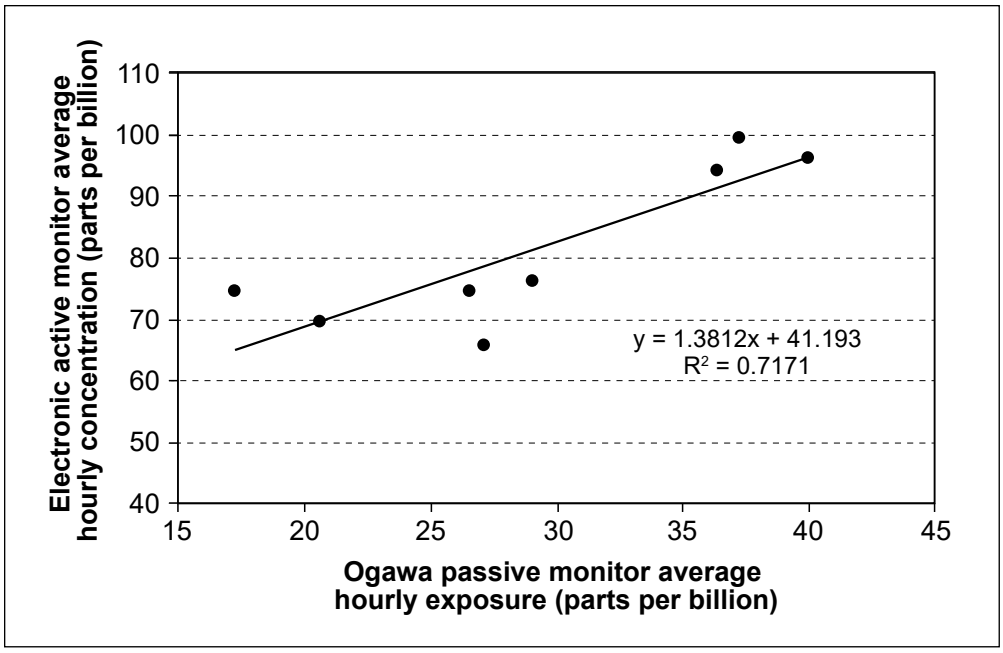


Figure 3.6—Relationship between the passive monitors deployed on individuals and the active monitoring data acquired from the nearest ozone monitoring stations.

Conclusions

- A 5-hour exposure period was sufficient to detect ozone concentrations significantly higher than field blanks and lab blanks.
- High variability across individual samplers and within samplers confounded statistical differences among the sampling days.
- The results of the passive samplers were well correlated to the ambient conditions captured by electronic ambient ozone monitors.

Proposed Changes for Future Experiments

1. Individuals would wear two Ogawa badges to reduce variability during the sampling period.
2. Formal training and requirements to follow protocols would be emphasized.

In this study, deployment of passive samplers was secondary to our primary objective to survey forest visitors; therefore, the protocol for sampler deployment may not have been stringently followed.

Chapter 4: Experiment 2—Application of Personal Ozone Exposure Monitoring in Urban Parks

Introduction

This study was part of a program of research investigating recreation activity, health benefits, and risks across socioeconomic differences in the City of Los Angeles, California. The scope of this research is designed to investigate multiple dimensions of community vulnerability and resilience under changing climate conditions in this socially complex metropolis (Winter et al. 2019). Potential study communities were evaluated for their percentage of residential land use (with an aim to stay away from zones that were primarily commercial or industrial), attributes of disadvantage and affluence, and amount of tree canopy cover (in this step, drawing primarily from McPherson et al. 2008 and the community profiles from a *Los Angeles Times* website called "Mapping L.A. Neighborhoods" [http://maps.latimes.com/neighborhoods]). The communities of Sun Valley and Brentwood were selected as case examples of a disadvantaged (DAC) and an affluent (AFF) community, respectively. In each community, two public parks were identified for focused study (fig. 4.1A). Each of the four parks chosen had similar physical amenities, including play areas for children; restrooms; open space for unstructured activities; at least one ballfield for soccer, baseball, or another sport; and picnic areas (fig. 4.1B).

Within the affluent community of Brentwood, Crestwood Hills Park (AFF) is tucked into a largely residential area consisting of single-family homes (fig. 4.2A). Barrington Recreation Center (AFF) is situated within a mixed development of multi-family housing, single-family homes, and commercial properties (fig. 4.2B). The two parks selected in Sun Valley were Sun Valley Recreation Center (DAC) (fig. 4.3A) and Fernangeles Recreation Center (DAC) (fig. 4.3B). Both were surrounded by a mix of commercial and residential land uses, and both had industrial centers nearby, including an active quarry and a battery recycling center (US Census Bureau 2014).

The Brentwood community (AFF) was characterized as having greater anticipated resilience to climate change effects as a result of its relative social, economic, and environmental attributes. Brentwood had a higher proportion of service and entertainment businesses, including health care services. In contrast, Sun Valley (DAC) had a higher percentage of manufacturing and other industrial sites as well as trucking and warehousing services. Sun Valley was anticipated to have greater vulnerability to climate change, owing primarily to socioeconomic disadvantages and other factors typical of low-income communities (see Winter et al. 2019). The Sun Valley population has less than half the median income of

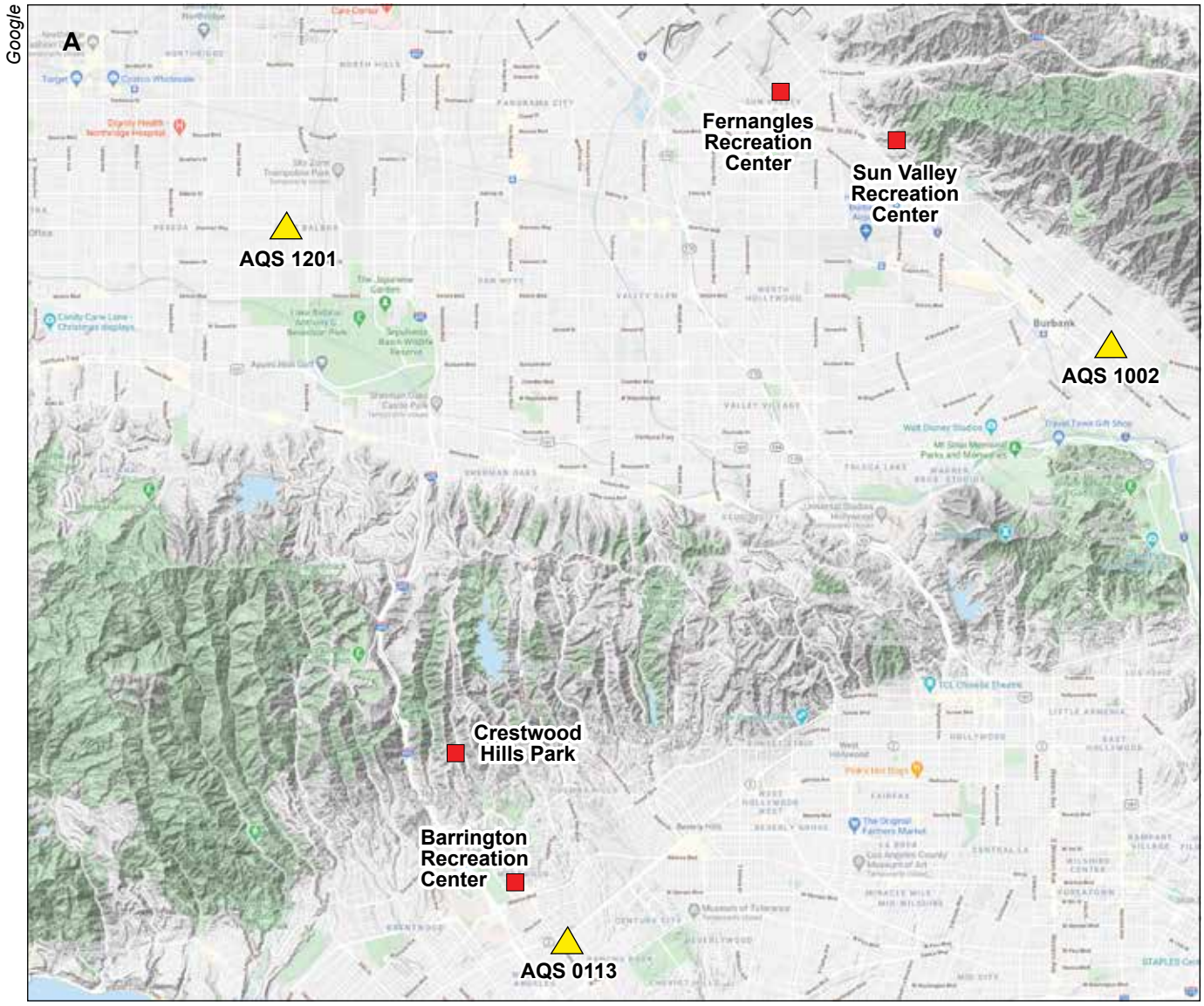


Figure 4.1A—Location of the four study sites and three electronic ozone monitoring stations. Crestwood Hills Park and Barrington Recreation Center are both in the coastal community of Brentwood (population). Fernangeles Recreation Center and Sun Valley Recreation Center are both in the inland community of Sun Valley (population). The yellow triangles are the locations of the electronic ozone monitoring stations.

Brentwood and a notably higher population density. Sun Valley had markedly less tree canopy cover, and residential areas were closer to major roadways.

Several differences between the Sun Valley area and Brentwood are evident (fig. 4.1B). Both Sun Valley parks are located near major highways and therefore are at greater risk of degraded air quality (O'Neill et al. 2003). Tree canopy cover appears more prevalent in the Brentwood community, and neither of the Brentwood parks appear to have industrial activity nearby. These characteristics (proximity to highways and industrial areas and tree canopy cover) confirm structural differences

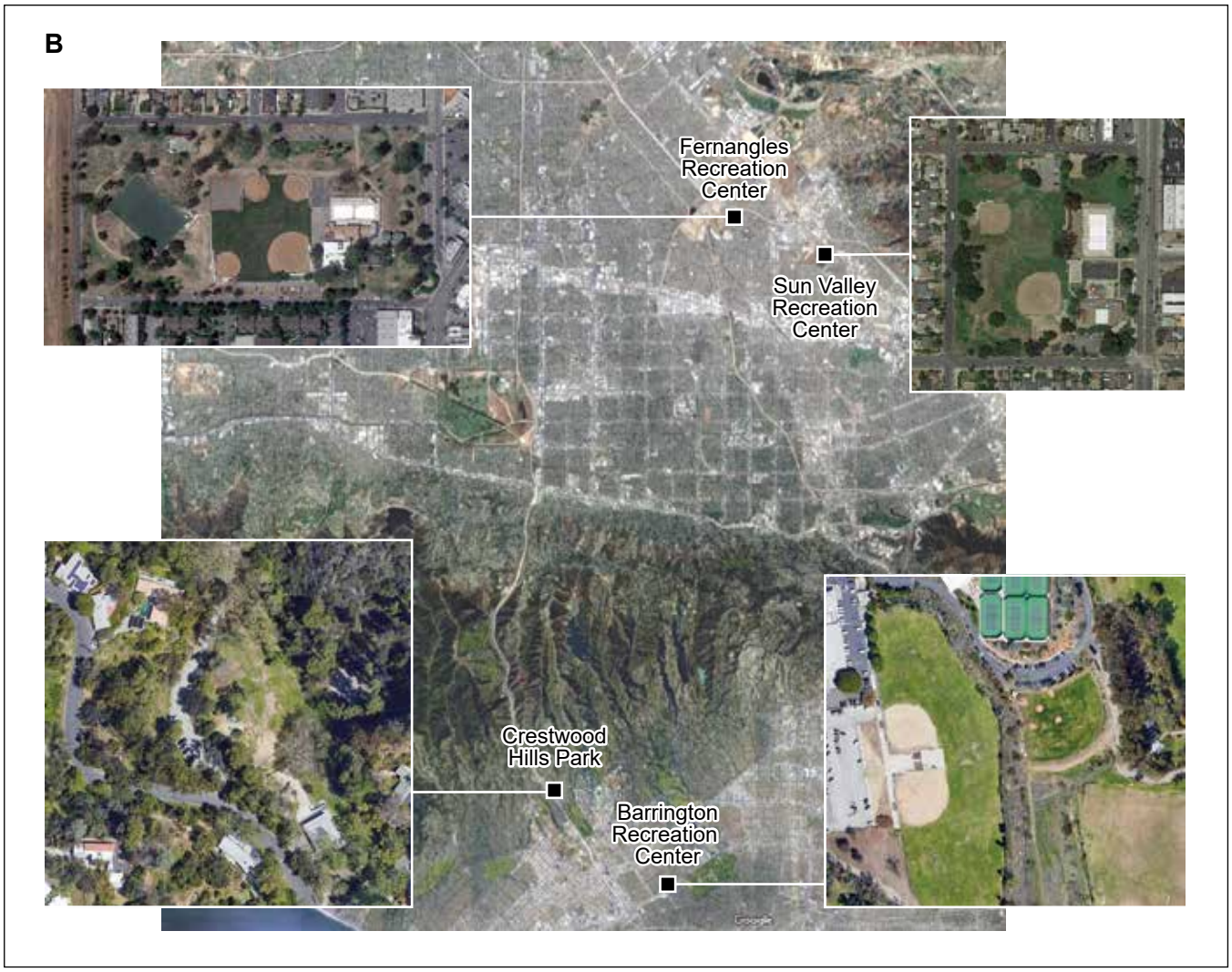


Figure 4.1B—Satellite views of the case study parks.

between the two communities. Our aim was to understand if these contrasting communities had different levels of environmental risk from impaired air quality, particularly from greater exposure to ozone.

Affluent and disadvantaged communities may be differently sensitive to environmental shifts associated with climate change, and furthermore, we might anticipate people with higher exposure to air pollution to make more use of urban parks on high-heat days (see Winter et al. 2019). This could result in an increased risk from elevated ozone, given the synergistic relationship between extended heat spells and ozone production.

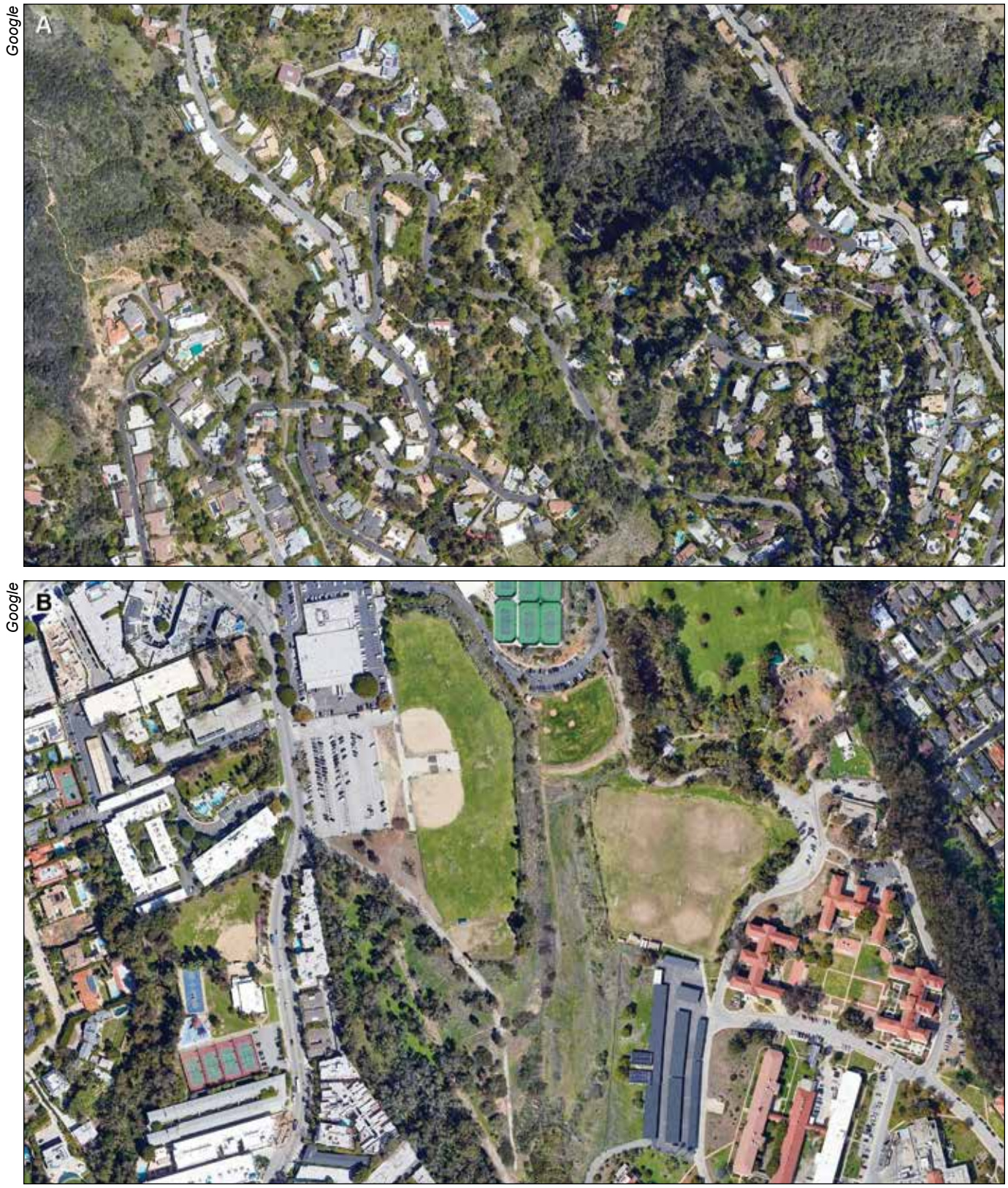


Figure 4.2—Satellite view of (A) Crestwood Hills Park and (B) Barrington Recreation Center in the affluent community of Brentwood. Note the presence of tree and vegetation cover.

Figure 4.3—Satellite view of (A) Sun Valley Recreation Center and (B) the Fernangeles Recreation Center in the disadvantaged community of Sun Valley. Note the nearby presence of major roadways.

Patricia L. Winter

Figure 4.4A—Front placement of personal ozone monitoring badges.

Patricia L. Winter

Figure 4.4B—Rear placement of personal ozone monitoring badges.

Approach and Methods

As described in previous chapters, Ogawa badge-type passive monitors were used to estimate exposures to ozone by park visitors. In experiment 2, two observers visited each park five times between late August and late September. Timing was determined by field team availability and months when ozone levels are typically high. We intentionally selected months expected to have higher ozone loads to increase the probability that we would be able to distinguish patterns of exposure between the two communities. Park observations took place between 10 a.m. and 3 p.m. on weekends and weekdays. During each visit, observers worked together to identity and record recreational use while following a prescribed path through the park. Observation sweeps occurred upon arrival at 10 a.m., 12 p.m., and 2 p.m. Between sweeps, observers remained in the park, outdoors. Each observer wore two badges, which resulted in the use of four filters per person and eight filters per

sampling day. A field blank was used during each park visit to check for systematic errors and correct for any effects of time and travel. The field blank was kept sealed in the container and carried in the field by one of the observers. The badges were pinned or clipped to clothing on the upper torso; one badge on the front and one badge on the back as shown in figures 4.4A and 4.4B. Specific placement on the torso varied based on garment and observer comfort.

The protocol for ozone monitoring and paired recreation observations was strictly prescribed and followed. The protocol ensured consistency of exposure methods across days, proper handling to prevent unintended contamination of collector badges and filter pads, and the ability to contrast data collected by different pairs of observers. Prior to deployment, the badges were stored in small sealable plastic bags placed in sealed plastic jars. Upon arrival, observers adhered to the protocol as shown in box 1.

Regional, Real-Time Ozone Monitoring

Los Angeles and much of the southern California region have a network of electronic ozone monitoring stations maintained by a colloquium of agencies including the U.S. EPA, California Air Resources Board, and South Coast Air Quality Management District (data are available online at http://www.epa.gov/airdata/ad_data.html). Monitoring locations are indicated by the triangles in figure 4.1A. The sites closest to each of the parks were selected both for comparisons of the passive ozone data and for comparisons of diurnal ozone concentration patterns. Search parameters were entered into the online data repository: state = California; county = Los Angeles; amount of ozone across range of dates; 24-hour period each day.[1]

Data and Results

Passive ozone monitors—

Ozone exposures determined by passive sampling differed widely across sampling days from a low of 29 ppb on September 5 to a high of 83 ppb on September 30 (fig. 4.5). The average across all days in parks in affluent versus disadvantaged communities was not significantly different: 46 ppb in the Brentwood (AFF) parks (black) as compared to 50 ppb (gray) in the Sun Valley (DAC) parks. In comparing the parks individually, Sun Valley Recreation Center (DAC) had the highest average value across sampling days, which was significantly higher than Barrington

[1] Investigators who want to replicate this procedure may benefit from knowing that there is a lag time between the days of interest and availability of the online data.

Box 1

1. Before exiting your vehicle, each field team member is to
   a. Open the jar
   b. Remove the baggie
   c. Open the baggie
   d. Remove the white badge by carefully holding the rectangular top piece (above the clip and safety pin on the badge) and **not** touching the ends of the cylinder
   e. Clip or pin the badge onto your clothing at the neck, chest, or shoulder area
      i. The badge must be attached to the top layer of clothing, uncovered, for the full field day (ideas for clipping places include top edge of pocket, collar of shirt, or edge of a short sleeve)
   f. Using the piece of tape on the jar, write the date and time that you attached the badge to your shirt, place the empty baggie in the jar, and leave the jar in your vehicle
2. During the field day be sure that you **do not**
   a. Blow onto or into the badge (the wind or a breeze will not affect the sampler)
   b. Touch the ends of the cylinder
   c. Get the device wet
   d. Smoke
   e. Apply sunblock onto or allow sunblock to drip or spray into the sampler
   f. If you happen to drop the badge, do not panic. Carefully pick up the badge by the rectangular top piece, shake gently (again, do not blow on badge), and continue on.
   f. If you drop the badge, write "dropped" on the tape on the jar at the end of the field day.
3. At the end of the field collection when you return to your vehicle
   a. Remove the badge from your clothing, again, being sure not to touch the ends of the cylinder
   b. Place the badge in the baggie and seal it
   c. Fold the baggie and place in the jar
   d. Close the jar
   f. Write down the time that you put the badge back in the baggie and sealed it (this information is written on the tape on the jar)
4. The sampling jar with badge sealed inside should be kept at room temperature until returned to the lab. Do not leave it in a hot vehicle, or in any vehicle other than to travel back and forth to the field site or from the lab to your home.

Recreation Center (AFF) (table 4.1). However, exposure at Fernangeles Recreation Center (DAC) was similar, on average, to that recorded at Barrington Recreation Center (AFF). Three days of measurement revealed ozone exposures that exceeded the U.S. EPA threshold for deleterious health effects, suggesting that limited outdoor activities would have been advisable on these days, especially for sensitive populations.

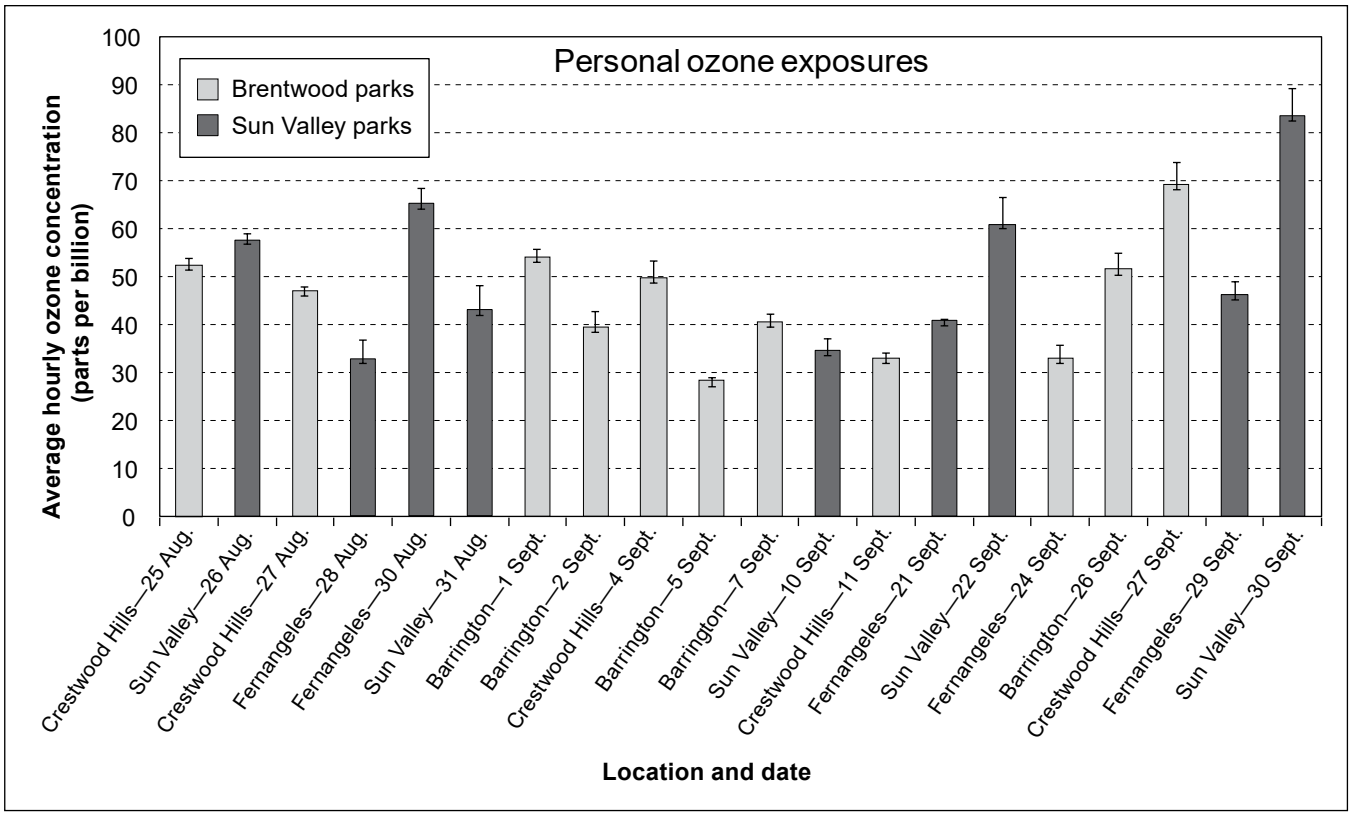


Figure 4.5—Personal ozone exposures as measured by passive samplers. Average exposure of two observers each wearing 4 filters (n = 8). Light grey bars are data from the affluent community parks (Barrington and Crestwood Hills); dark grey bars are from the parks in disadvantaged communities (Fernangeles and Sun Valley). Error bars = standard error of the mean; ppb = parts per billion.

Table 4.1—Average hourly ozone exposure across all sampling days at Brentwood and Sun Valley parks

Brentwood		Sun Valley	
Barrington	**Crestwood**	**Fernangeles**	**Sun Valley**
Parts per billion			
42.6	50.2	43.5	55.9

Active ozone monitors—

A common approach when calibrating passive monitoring systems is to compare the calculated data derived from the passive technique to the data produced from electronic monitoring instruments. Los Angeles has four ozone monitoring stations in relative proximity to the parks. For the Brentwood (AFF) parks, hourly ozone concentrations for each sampling day were downloaded from station number 0113 (fig. 4.1A). Two monitoring stations were available for the Sun Valley (DAC) parks—1201 and 1002. Linear regression analysis was conducted to determine if the data from the two active monitoring stations differed. Analysis of the two data

sources revealed that the average ozone concentrations on the 8 days monitored had a slope of 0.886 and an R^2 of 0.881, indicating a slight bias between the two, but a good correlation. As there was no way to determine which of the two monitoring stations was correct, we chose to use an average of the two stations for comparison to the passive monitor data.

A side-by-side comparison of active monitors and passive Ogawa badges did not provide definitive results (fig. 4.6). A regression analysis comparing the complete datasets for both passive monitor and active station data determined an R^2 of less than 0.3. However, only Barrington Park (AFF) was within 0.6 km (1 mi) of the active monitoring station; Crestwood Hills Park was 4 km (2.5 mi) from the station (fig. 4.1A); and monitoring stations for the Sun Valley parks (DAC) were more than 8 km (5 mi) away. When each park is analyzed separately, the regression value between the passive monitor and active station data for the Brentwood (AFF) parks is $R^2 = 0.68$, suggesting that distant active monitoring stations may not yield the most accurate ozone concentrations for the DAC parks (fig. 4.6).

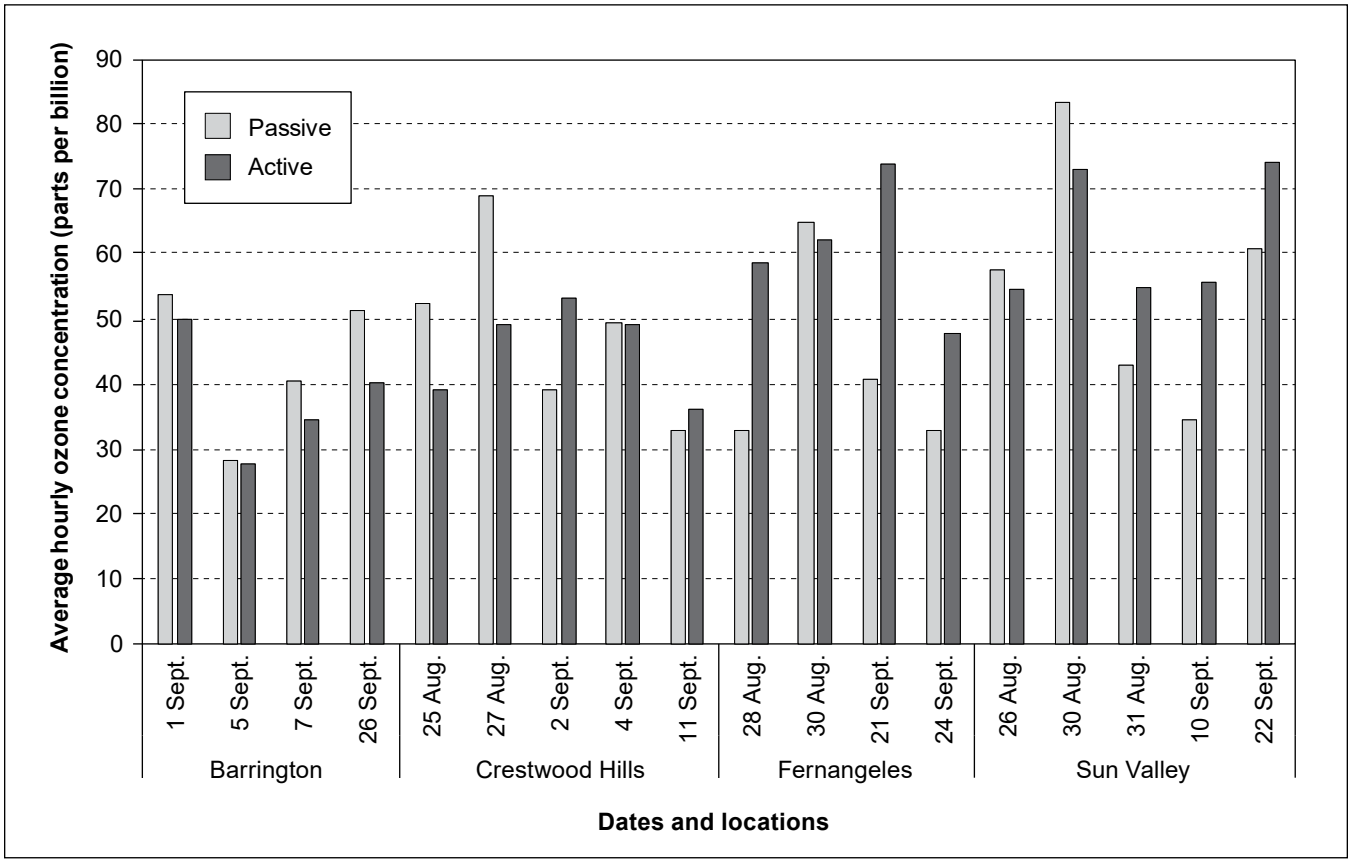


Figure 4.6—A comparison of results from the passive samplers and the electronic monitoring stations. The light gray bars indicate the average hourly concentrations from the passive samplers (n = 8, error bars = standard error of the mean). The dark grey bars are the average hourly values from active monitoring stations for the same period. Note that these data are a single datum for each day.

The passive ozone concentrations do not appear to indicate a substantial exposure difference between the parks in the Brentwood (AFF) and Sun Valley (DAC) communities (figs. 4.5 and 4.6). However, it was clear based on both active and passive monitoring data that there was wide variability among days. It was difficult to draw definitive conclusions from the data either in comparing area type,

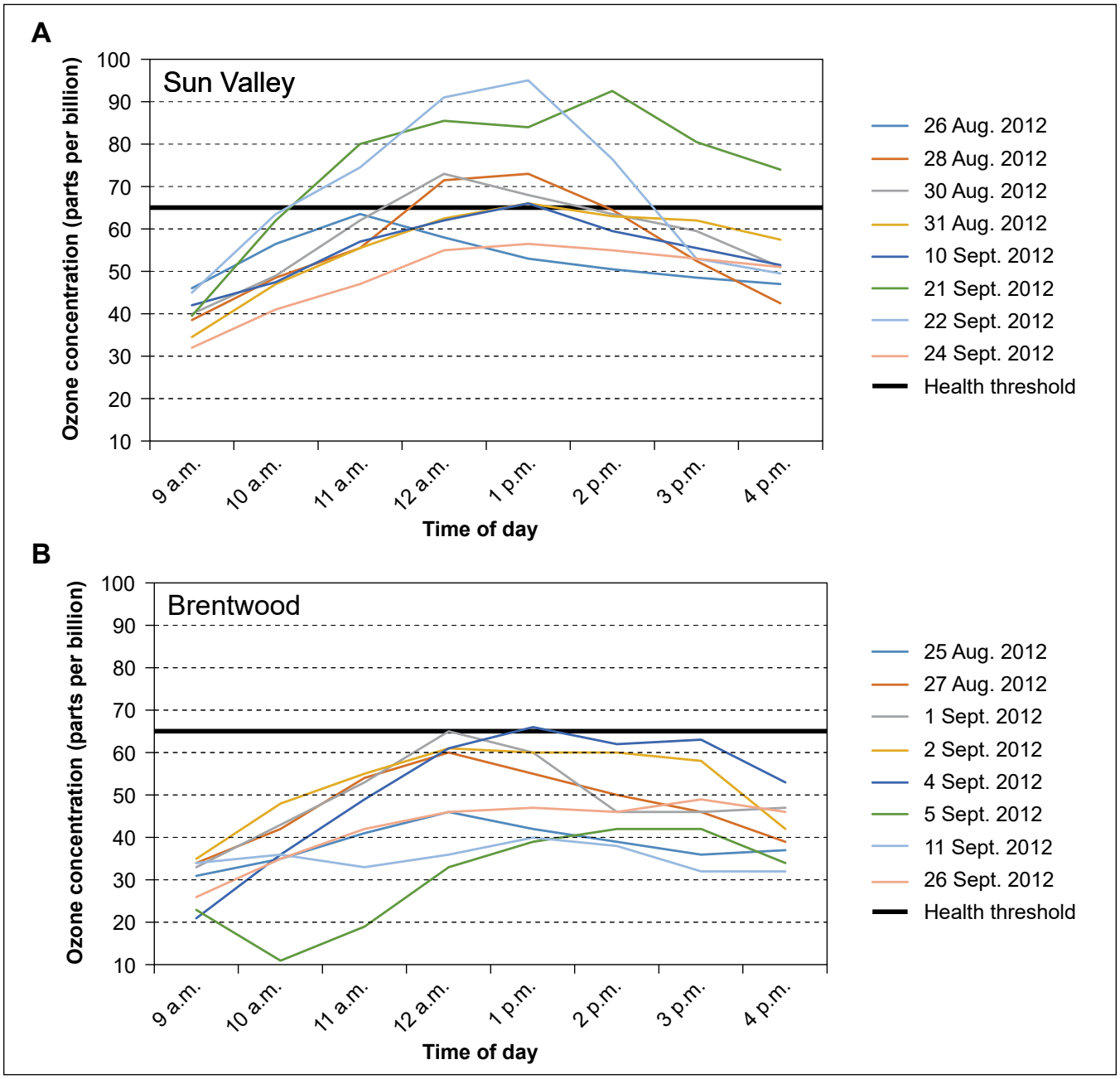


Figure 4.7—One-hour ozone averages for each of the study days, downloaded from the (A) Sun Valley (DAC) and (B) Brentwood (AFF) monitoring stations. The solid line at 65 parts per billion indicates the U.S. Environmental Protection Agency standard for harmful effects to sensitive individuals (the young, elderly, and those with compromising health issues).

or contrasting parks across days, because only one team was deployed each day, none of the observation days were paired, and only one park was visited on any single day. Furthermore, passive samplers can provide only an aggregate measure of the exposure during the 5-hour exposure period, with no indication of hourly variation or peak exposures. As a further validation of the passive sampler data, the daily concentrations from the active monitors were compared (fig. 4.7).

In reviewing ambient ozone concentration data from the nearest active monitoring stations, it is clear that the Sun Valley (DAC) community was at much greater risk from ozone exposure than the Brentwood community (AFF) (fig. 4.7) during the study period. On most days, both locations demonstrated the typical diurnal curves—lower ozone concentrations in the evening and early morning and higher concentrations during daylight hours, peaking between 12 p.m. and 2 p.m. Also notable is the high day-to-day variability. At the Sun Valley (DAC) monitoring station, the peak daily concentration ranged from 55 ppb on September 24 to more than 100 ppb on September 21. At the Brentwood (AFF) monitoring station, the peak concentrations were just over the 65 ppb threshold on September 1 and 7, and just under 40 ppb on September 11.

Conclusions

- All calculated ozone concentration from the passive samples were statistically higher than the field blanks and lab blanks.
- Replication among the four samplers deployed each outing was within the 20 percent standard.
- Ozone levels from the passive samplers were not always well correlated with the active monitors. This may have been a result of the distance between the park and its associated monitoring station.
- As a group, there were no significant differences between Brentwood and Sun Valley, the two areas based on passive data alone; however, the active electronic monitors indicated an overall higher mid-day ozone concentration in the Sun Valley parks.
- Daily ozone variability at the Brentwood and Sun Valley locations and the lack of paired sampling days likely contributed to the absence of statistical significance in comparisons of the passive sampling data.

Proposed Changes for Future Experiments

1. Pair sampling days and times to identify significant differences
2. Move badges from the torso to the head
3. Deploy stationary passive samplers
4. Add an affluent **inland** community and a disadvantaged **coastal** community to the study

Chapter 5: Experiment 3—Ozone Monitoring in Urban Parks

In the third and final experiment, additional parks were added to address concerns about the effects of inland versus coastal geographic position on ozone concentrations. The observations and ozone protocols were modified by (1) adding an extra hour to the exposure time (6 hours rather than 5 hours); (2) moving the position of Ogawa badges from the torso to a hat worn on the head, with each observer wearing only one badge rather than the two badges worn in previous studies; and (3) pairing all sampling dates so that one park in an affluent community was visited at the same time as one park in a disadvantaged community (table 2.1).

Parks and Communities

One of the concerns in interpreting the data from experiment 2 was the geographic covariance with socioeconomic structure that affects air quality. In experiment 2, the more affluent community was on the coast, which is prone to better air quality, while the disadvantaged community was inland where the number of poor air quality days tends to be greater. To balance our inland and coastal comparisons for this final round of study, we chose two additional communities, adding two parks within each community—Coldwater Canyon Park (fig. 5.1) and Laurel Canyon Dog Park (fig. 5.1). Wilmington Recreation Center (renamed Will Hall Park after the study was complete) and Banning Park are in the disadvantaged coastal community of Wilmington (fig. 5.2); The newly added coastal parks were similar in the type and range of amenities as the four parks in experiment 2. However, finding and choosing two additional inland parks with play areas, restrooms, open spaces, etc., within the affluent community of Hollywood Hills West was more challenging. The amenities at Coldwater Canyon Park include parking areas, an educational center with restrooms, a few picnic tables, an educational native plant and water area, and many trails leading through the park and connecting to adjacent parks and wild areas. Laurel Canyon Dog Park did have many of the same features including a parking lot, children's play area, picnic tables, benches scattered throughout the park, and portable toilets. However, the main purpose of this park is to provide a recreation and exercise space for canines as well as their human partners. In both cases these parks lacked sports fields, recreation centers for group programming, and other features more aligned with a developed urban park. They were also somewhat smaller in acreage, with Laurel Canyon Dog Park being the smallest of all parks in this study.

Air Monitoring Using Passive Samplers Worn on Hats

After evaluating the variation in ozone concentrations across badges as well as across individuals, we concluded that securing the badges to the front and back of

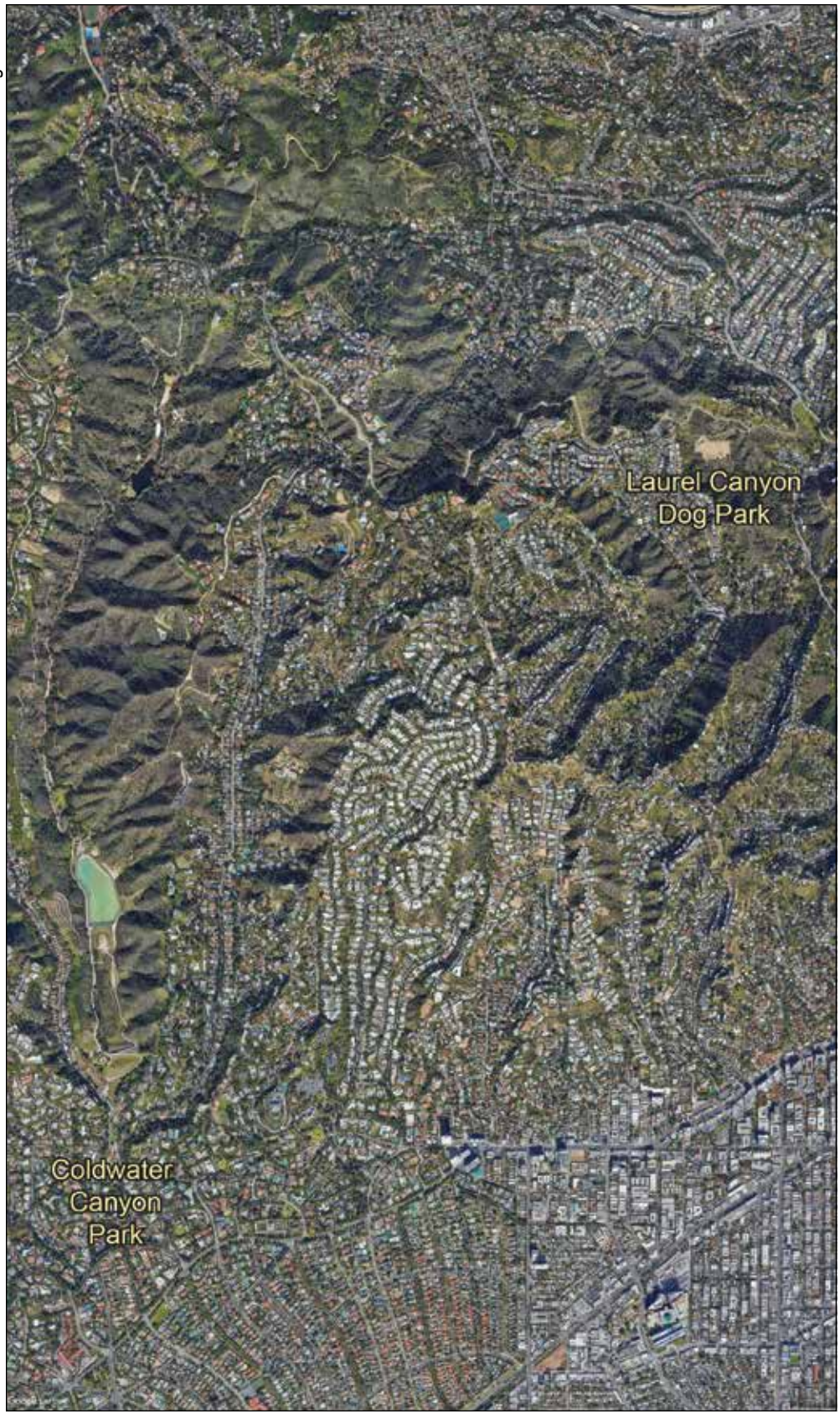
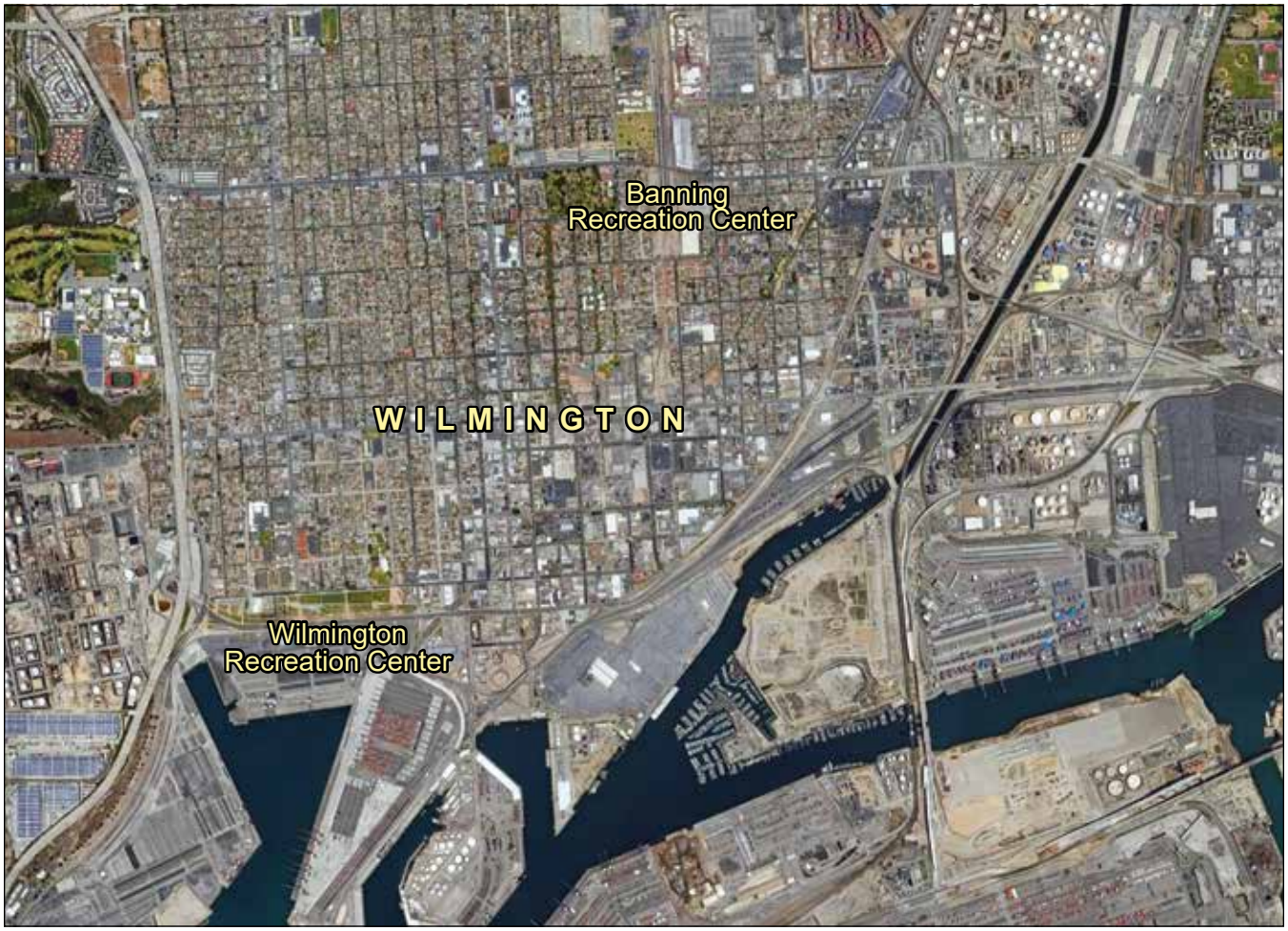


Figure 5.1—The two parks in the Hollywood Hills West community, added to the study in experiment 3, include Coldwater Canyon and Laurel Canyon Dog Park to the east of Coldwater Canyon.



Google

Figure 5.2—The two parks in the Wilmington community include Wilmington Recreation Center, adjacent to the Port of Los Angeles, and the Banning Recreation Center, about 3.2 km (2 mi) to the north.

the torso may have imposed systematic variability as the observers sat or leaned against surfaces while resting between observation sweeps. To reduce that variability, badges were moved from the torso to hats worn by all observers for the duration of the outdoor exposures (fig. 5.3). This placement also put the badge closer to the observer's nose and mouth to better represent ambient conditions experienced by a person breathing. Aside from changing the badge placement, the protocol for deploying badges was identical to that in experiment 2. Because of equipment and resource limitations, the number of badges worn by each observer was reduced to one instead of two.

Patricia L Winter

Figure 5.3—Deployment of passive ozone monitors on hats. Note that only one badge was worn by each of the observers.

Table 5.1—Schedule of observation days

Day of week	Time band	Date	A/D	Location
Saturday	6:45 a.m.–1:15 p.m.	22 August	A	Coldwater Canyon Park—Hollywood Hills West
Saturday	6:45 a.m.–1:15 p.m.	22 August	D	Banning Recreation Center—Wilmington
Sunday	6:45 a.m.–1:15 p.m.	23 August	A	Crestwood Hills Park—Brentwood
Sunday	6:45 a.m.–1:15 p.m.	23 August	D	Sun Valley Recreation Center—Sun Valley
Monday	6:45 a.m.–1:15 p.m.	24 August	A	Laurel Canyon Dog Park—Hollywood Hills West
Monday	6:45 a.m.–1:15 p.m.	24 August	D	Banning Recreation Center—Wilmington
Thursday	6:45 a.m.–1:15 p.m.	27 August	A	Crestwood Hills Park—Brentwood
Thursday	6:45 a.m.–1:15 p.m.	27 August	D	Fernangeles Recreation Center—Sun Valley
Saturday	12:45 p.m.–7:15 p.m.	29 August	A	Coldwater Canyon Park—Hollywood Hills West
Saturday	12:45 p.m.–7:15 p.m.	29 August	D	Wilmington Recreation Center—Wilmington
Sunday	6:45 a.m.–1:15 p.m.	30 August	A	Crestwood Hills Park—Brentwood
Sunday	6:45 a.m.–1:15 p.m.	30 August	D	Banning Recreation Center—Wilmington
Tuesday	6:45 a.m.–1:15 p.m.	1 Sept.	D	Sun Valley Recreation Center—Sun Valley
Tuesday	6:45 a.m.–1:15 p.m.	1 Sept.	A	Barrington Recreation Center—Brentwood
Wednesday	12:45 p.m.–7:15 p.m.	2 Sept.	A	Laurel Canyon Dog Park—Hollywood Hills West
Wednesday	12:45 p.m.–7:15 p.m.	2 Sept.	D	Wilmington Recreation Center—Wilmington
Friday	6:45 a.m.–1:15 p.m.	4 Sept.	A	Coldwater Canyon Park—Hollywood Hills West
Friday	6:45 a.m.–1:15 p.m.	4 Sept.	D	Wilmington Recreation Center—Wilmington
Saturday	12:45 p.m.–7:15 p.m.	5 Sept.	A	Crestwood Hills Park—Brentwood
Saturday	12:45 p.m.–7:15 p.m.	5 Sept.	D	Fernangeles Recreation Center—Sun Valley
Sunday	6:45 a.m.–1:15 p.m.	6 Sept.	A	Barrington Recreation Center—Brentwood
Sunday	6:45 a.m.–1:15 p.m.	6 Sept.	D	Fernangeles Recreation Center—Sun Valley
Tuesday	12:45 p.m.–7:15 p.m.	8 Sept.	A	Crestwood Hills Park—Brentwood
Tuesday	12:45 p.m.–7:15 p.m.	8 Sept.	D	Wilmington Recreation Center—Wilmington
Thursday	6:45 a.m.–1:15 p.m.	10 Sept.	D	Sun Valley Recreation Center—Sun Valley
Thursday	6:45 a.m.–1:15 p.m.	10 Sept.	A	Coldwater Canyon Park—Hollywood Hills West
Saturday	6:45 a.m.–1:15 p.m.	12 Sept.	A	Barrington Recreation Center—Brentwood
Saturday	6:45 a.m.–1:15 p.m.	12 Sept.	D	Wilmington Recreation Center—Wilmington
Sunday	12:45 p.m.–7:15 p.m.	13 Sept.	D	Sun Valley Recreation Center—Sun Valley
Sunday	12:45 p.m.–7:15 p.m.	13 Sept.	A	Barrington Recreation Center—Brentwood
Monday	12:45 p.m.–7:15 p.m.	14 Sept.	A	Barrington Recreation Center—Brentwood
Monday	12:45 p.m.–7:15 p.m.	14 Sept.	D	Fernangeles Recreation Center—Sun Valley
Wednesday	12:45 p.m.–7:15 p.m.	16 Sept.	A	Barrington Recreation Center—Brentwood
Wednesday	12:45 p.m.–7:15 p.m.	16 Sept.	D	Banning Recreation Center—Wilmington
Thursday	6:45 a.m.–1:15 p.m.	17 Sept.	D	Fernangeles Recreation Center—Sun Valley
Thursday	6:45 a.m.–1:15 p.m.	17 Sept.	A	Laurel Canyon Dog Park—Hollywood Hills West
Saturday	12:45 p.m.–7:15 p.m.	19 Sept.	D	Sun Valley Recreation Center—Sun Valley
Saturday	12:45 p.m.–7:15 p.m.	19 Sept.	A	Laurel Canyon Dog Park—Hollywood Hills West
Sunday	6:45 a.m.–1:15 p.m.	20 Sept.	A	Laurel Canyon Dog Park—Hollywood Hills West
Sunday	6:45 a.m.–1:15 p.m.	20 Sept.	D	Wilmington Recreation Center—Wilmington
Monday	12:45 p.m.–7:15 p.m.	21 Sept.	A	Crestwood Hills Park—Brentwood

Day of week	Time band	Date	A/D	Location
Monday	12:45 p.m.–7:15 p.m.	21 Sept.	D	Sun Valley Recreation Center—Sun Valley
Tuesday	12:45 p.m.–7:15 p.m.	22 Sept.	A	Coldwater Canyon Park—Hollywood Hills West
Tuesday	12:45 p.m.–7:15 p.m.	22 Sept.	D	Banning Recreation Center—Wilmington
Saturday	12:45 p.m.–7:15 p.m.	26 Sept.	A	Laurel Canyon Dog Park—Hollywood Hills West
Saturday	12:45 p.m.–7:15 p.m.	26 Sept.	D	Banning Recreation Center—Wilmington
Sunday	12:45 p.m.–7:15 p.m.	27 Sept.	D	Fernangeles Recreation Center—Sun Valley
Sunday	12:45 p.m.–7:15 p.m.	27 Sept.	A	Coldwater Canyon Park—Hollywood Hills West

Note: Each day and time band included one park from an affluent (A) community and one park from a disadvantaged (D) community. Parks were chosen randomly; there was no effort to match inland and coastal parks.

Duration, Time Bands, and Paired Observations

A third concern that emerged during analysis and interpretation of findings from experiment 2 was the inability to directly compare ozone exposures between affluent and disadvantaged communities. Knowing that daily and hourly variations have a considerable range, we deemed it essential to have direct contrasts between community types on the same day, with data gathered during the same periods. To address these issues, we deployed two teams of observers for each sampling day—one team at a park in a disadvantaged community and one team at one of the parks in an affluent community. These comparison pairs were randomized across communities and parks. We also increased the exposure time from 5 to 6 hours, hoping to increase the likelihood of "signal detection" in our passive monitoring. To address concerns regarding periods of park use and considerations of team safety, we established a morning observation period (7 a.m. to 1 p.m.) and an afternoon observation period (1 p.m. to 7 p.m.). Observation days and times were distributed equally across all parks. All parks were visited three times in the morning and three times in the afternoon, varied across weekends and weekdays for six visits total at each park. Details of the field sampling scheme can be seen in table 5.1.

Stationary Ozone Monitors

Ogawa passive samplers have been used most frequently as integrative air monitoring stations where real-time electronic monitors cannot be deployed. Although our study areas had electronic monitors relatively close by, as noted in chapter 1, local conditions can have significant effects on ozone concentrations. At each park, two passive samplers were set up using standard techniques as described by Bytnerowicz (2005, fig. 5.4). Stationary monitors were installed at the Banning, Barrington, and Crestwood Hills Recreation Centers and Wilmington Park on 27 August 2015; and at Coldwater Canyon, Laurel Canyon, Fernangeles,

and Sun Valley on 28 August. Most stationary monitors were located on roofs or behind locked gates to prevent tampering, and in areas with open and clear air fetches. At Laurel Canyon Dog Park, stationary monitors were mounted to a tall wooden post with cross beam, placed into soil slightly above the parking lot in a vegetated area, away from established paths and the main recreation areas. The passive samplers were collected from the Banning, Barrington, Crestwood Hills, and Wilmington sites on 28 September; and from the Coldwater Canyon, Fernangeles, Laurel Canyon, and Sun Valley sites on 30 September. Processing, extraction, and calculations were identical to those used for the personal ozone monitors, with the longer exposure of approximately 30 days taken into the calculations.

David Olson

Figure 5.4—Stationary passive ozone monitors. The Ogawa badges were mounted in a bell-shaped shelter to protect them from precipitation and the direct impact of wind.

Results and Data

All four parks in the two coastal communities had similar average hourly ambient ozone concentrations over the 30-day exposures (table 5.1). Unlike the personal badges worn by the observers, the stationary monitors incorporate nighttime ozone concentrations. Thus, the average hourly ozone concentration levels are measured over a full 24-hour period (30 days) rather than a 6-hour daylight-only period for the personal badges. Of the coastal parks, ozone concentrations at Barrington trended a little high and Crestwood Hills trended a little low. Of the inland parks, both of the parks in the more affluent neighborhoods (Laurel Canyon and Coldwater Canyon) showed average hourly values substantially higher than at either of the inland parks in the disadvantaged neighborhoods (Sun Valley and Fernangeles). This appears to be a function of topography. Although both Sun Valley parks are situated among busy freeways and industrial activity, the terrain is flat with unrestricted airflow. In contrast, Laurel Canyon and Coldwater Canyon Parks both are located in narrow canyons at higher elevations. As noted in chapter 1, mountainous areas tend to have higher nighttime ozone concentrations, which appear to be reflected in the higher hourly averages recorded by experiment 3.

Electronic Monitors

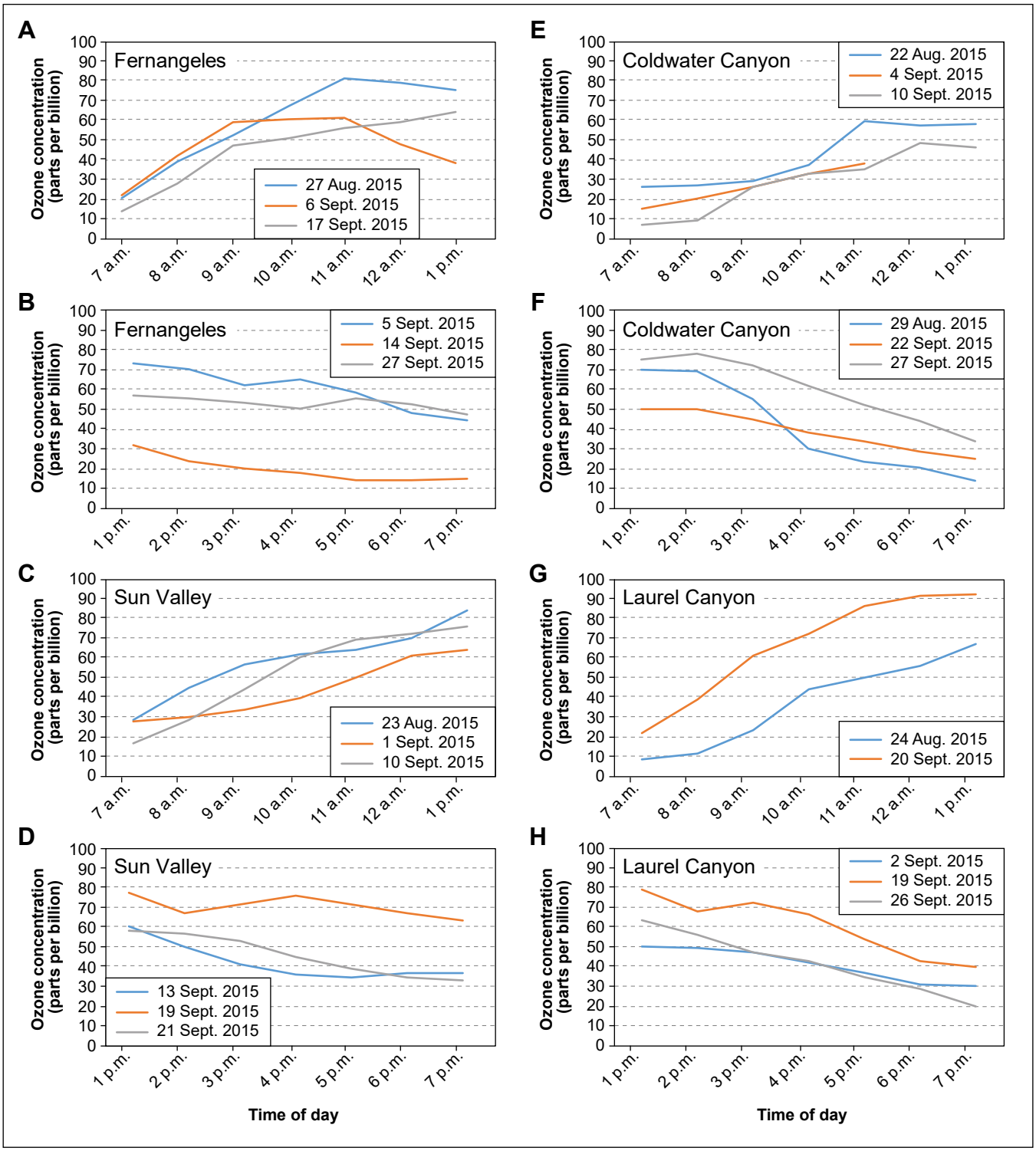
Visitation days for each park were chosen randomly. This approach allowed us to capture a wide range of ambient ozone concentrations (figs. 5.5 and 5.6). For the inland parks, the 7 a.m. ozone concentrations started at between 15 and 35 ppb. Thirty to 35 ppb is generally considered the average global background concentration (Vingarzan 2004), but lower ozone concentrations at specific locations are not unusual. Concentrations increased during the morning hours, peaking at roughly 1 p.m. Afternoon concentrations beginning at 1 p.m. started at the peak of the daily concentration and declined during the observation period, but in most cases did not reach background levels by the end of the observation period at 7 p.m. Coastal parks had slightly lower initial ozone concentrations, on average, in comparison to inland parks. This is typical of coastal conditions where the ocean's influence moderates evening concentrations.

Variability at any given park on any given day was high. Laurel Canyon (fig. 5.5G) is a good example of this. Morning concentrations on September 20 were nearly 50 percent higher than measured on August 24. The afternoon concentrations at Fernangeles (fig. 5.5B) on September 5 were nearly double the concentrations on September 14. By chance, other parks have very similar ambient concentrations such as the afternoon in Wilmington (fig. 5.6D) and Crestwood Hills (fig. 5.6H).

The wide daily variation within and across locations highlighted in figures 5.5 and 5.6 was not reflected in the stationary passive monitoring data (table 5.2). The integrated hourly ozone concentration calculated over the 30-day monitoring period was similar across all locations and did not capture the diurnal variability. The two datasets reflect two different approaches to quantifying air pollution exposure. In comparison to the 6-hour ozone concentrations shown in figures 5.5 and 5.6, the stationary passive data reflects a 24-hour sampling period, including approximately 12 hours of background ozone concentrations after dark. The difference between these two datasets has important implications for human health. Although the average 24-hour

Table 5.2—Average hourly ozone concentrations (n=4) as determined by the stationary passive samplers deployed at each of the eight urban parks

Locations	Average ozone
	Parts per billion
Coastal parks:	
Banning	13.852
Wilmington	13.323
Barrington	13.898
Crestwood Hills	12.899
Inland parks:	
Sun Valley	13.742
Fernangeles	14.003
Laurel Canyon	16.301
Coldwater Canyon	16.070



Figures 5.5—Data from active electronic monitors nearest to the inland parks. Charts A through D are the two parks in the disadvantaged community of Sun Valley, and charts E through H are the two parks in the affluent community of Hollywood Hills West. Each park is represented by two charts—the first displays morning concentrations during the time observers were in the field, and the second represents afternoon ozone concentrations, in parts per billion. (Note: no electronic data are available for Sept. 17, AM. at Laurel Canyon Dog Park, panel G)

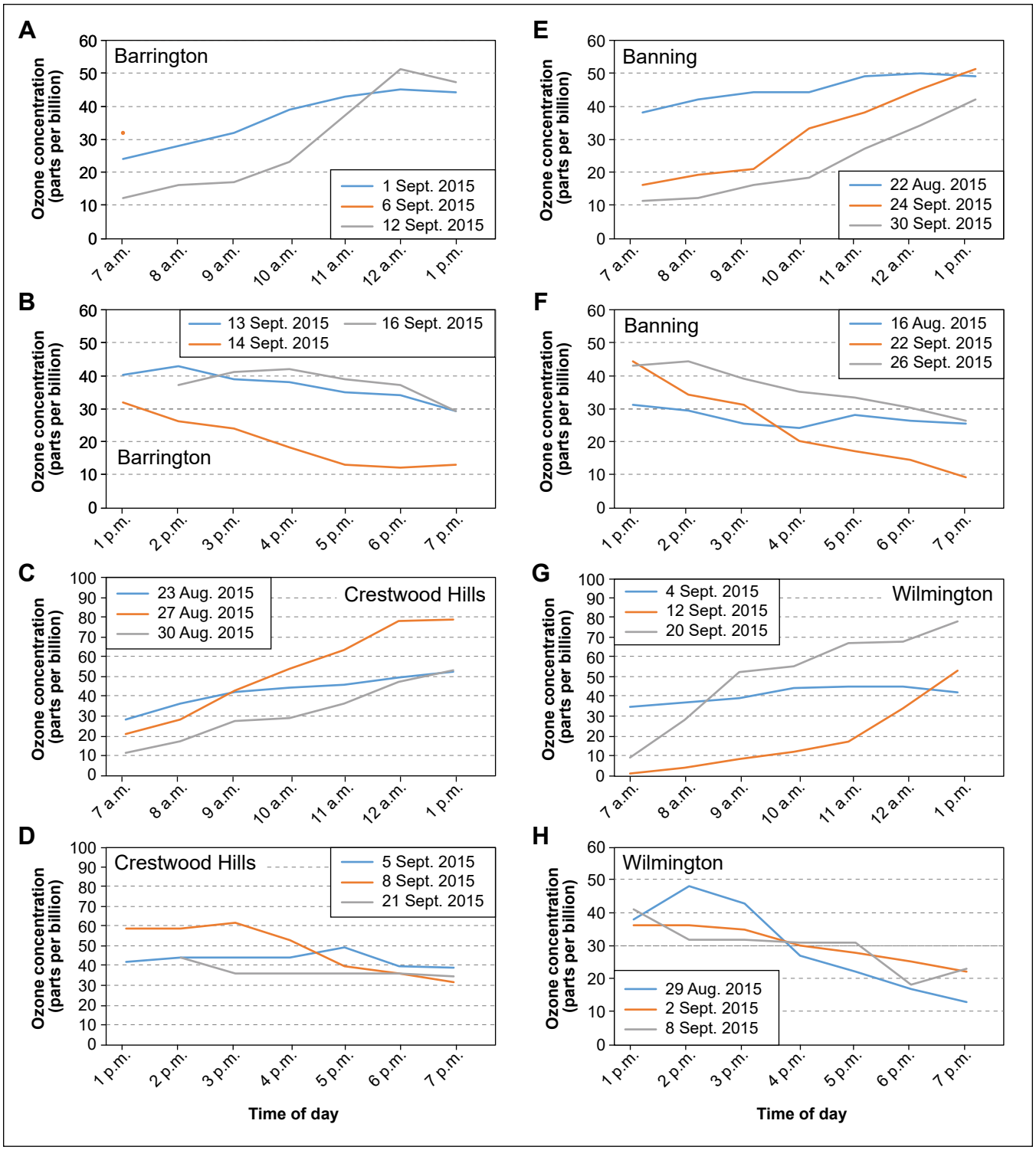


Figure 5.6—Data from the active electronic monitors nearest to the coastal parks. Charts A through D are from the disadvantaged community of Wilmington, and charts E through H are from the affluent community of Brentwood. Each park is represented by two charts—one displays morning concentrations during the time observers were in the field, and the second represents afternoon ozone concentrations, in parts per billion. (Note: no electronic data are available for Sept. 6, AM. at Barrington Park, panel A)

concentrations are well within the human health standards, people are generally exposed to ambient ozone during the day and for a few hours at a time. If these hours of exposure occur at the peak of ozone concentrations, the effective dose may well be in the critical zone for human health considerations. For ecological considerations such as forest health and sustainable habitat, 24-hour data are important. But it is also important to determine the diurnal concentrations to understand not only chronic exposures but the acute exposures that occur during the day.

Passive Personal Monitors

Ozone exposures calculated from the personal monitors worn by observers were much higher when the Ogawa badges were placed on a hat rather than on the torso (compare fig. 4.5 to fig. 5.7). Because of the increased volume of air affecting the sampler, passive samplers unprotected from wind would be expected to have higher ozone readings than passive samplers deployed in protective shelters that measure ozone by passive diffusion (as is typical in environmental monitoring). The large differences between badges worn on the torso and badges worn on the head was not expected. Research on the fluid mechanics of air movement around a living human body has shown that physical and physiological features of the body have important thermo- and aerodynamic influences on wind velocity and direction (Murakami et al. 1999). Measuring and modeling airflow around the human body is complicated (Li and Ito 2012, 2014; Murakami et al. 1999), but it has a significant effect on delivery of ambient pollutant loads (Rim and Novoselac 2009). In general, wind velocities are higher around the head than airflow around the torso at a given windspeed (fig. 5.8) (Arinami et al. 2017). However, many factors influence the magnitude of the difference: windspeed and direction; temperature and humidity of the air, wind, and human body; steady wind flows versus turbulent winds; and position and movement of the human subject all affect the airflow around the body. The act of breathing itself affects the wind velocity around the head and therefore has an important influence on pollution delivery to human lungs (Rim and Novoselac 2009). One conclusion reached by Schmees et al. (2008) is that personal monitors placed on the chest are probably not effectively monitoring actual pollutant concentrations at the mouth and nose.

Although placing Ogawa monitors at the side of the head on a hat is not the same as directly in front of the face, we maintain that ozone concentrations determined in experiment 3 are a more accurate representation of what observers were experiencing in each Los Angeles park.

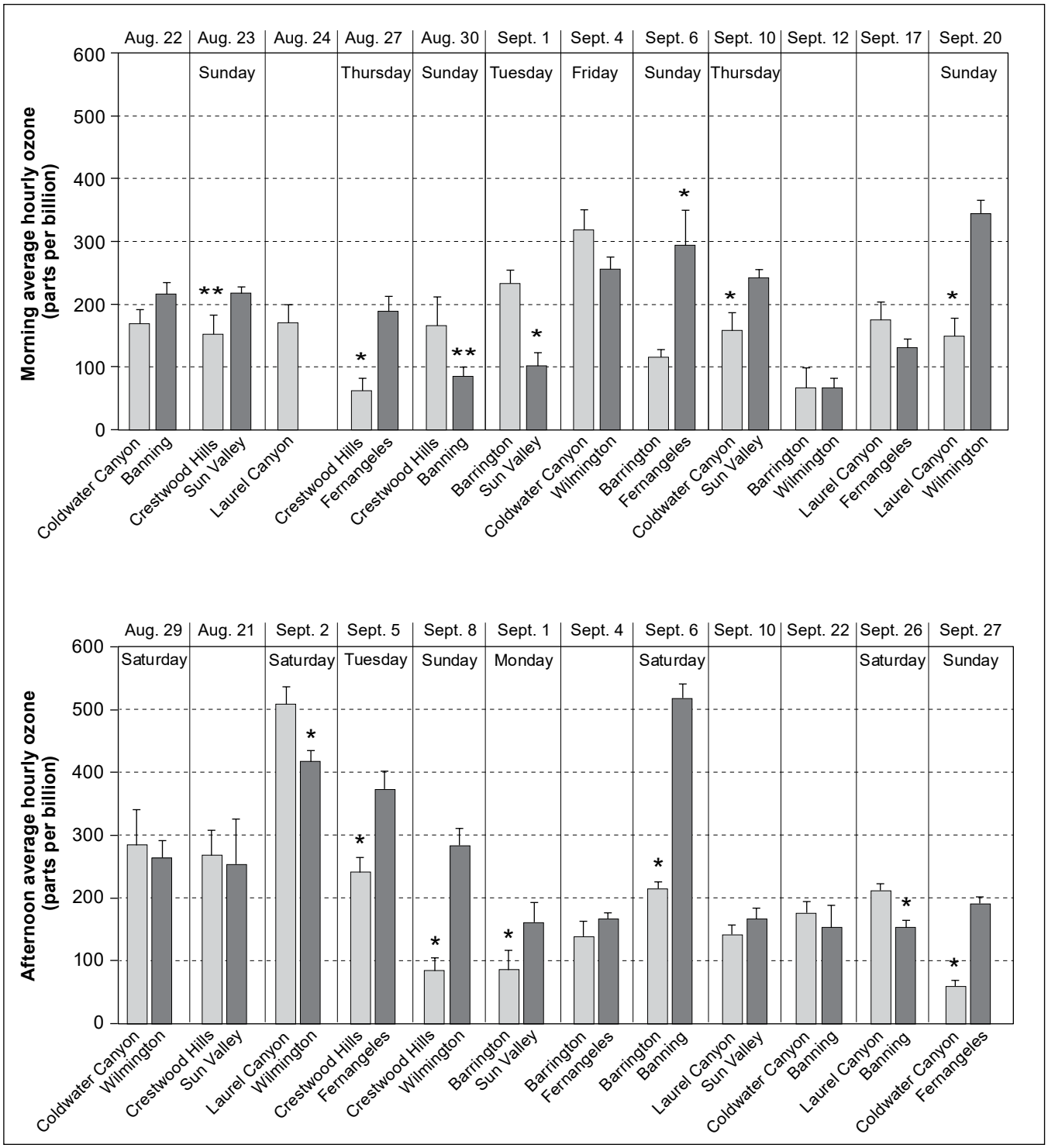


Figure 5.7—Ozone concentrations calculated from passive samplers measured at the observers' heads. The light gray bars represent parks in the affluent communities and the dark gray bars represent parks in the disadvantaged communities. Each bar displays the average of four filters (two per observer), and the error bar = standard error of the mean. Single asterisks denote $p < 0.05$ and double asterisks denote $p \approx 0.07$.

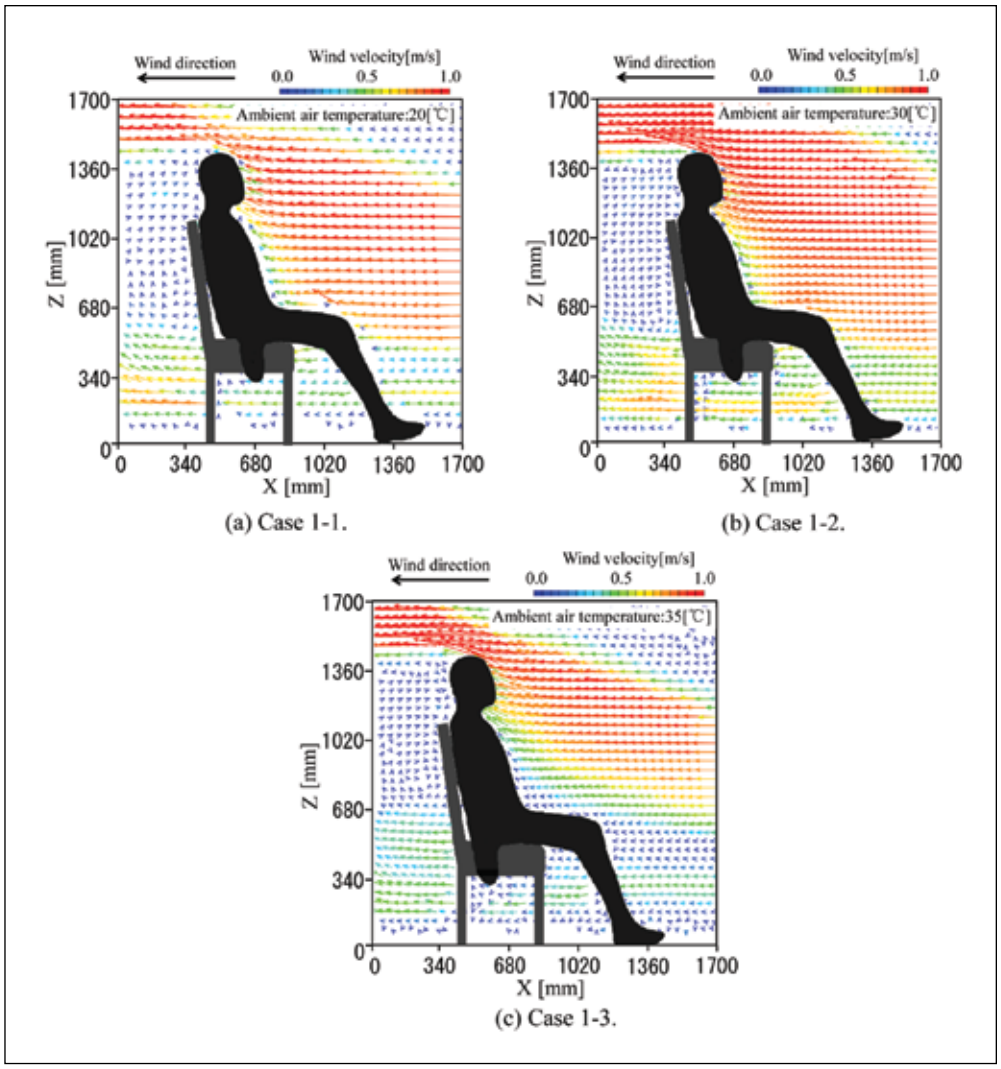


Figure 5.8—The effect of ambient temperature and body position on wind velocity around a human body. Wind velocities decrease to near zero as air approaches the torso, especially at higher ambient temperatures, but they slow only slightly, if at all, around the head. Note that at all temperatures the wind velocity at the back of the body is zero. Illustration adapted from Arinami et al. (2017).

The average hourly morning ozone concentrations were lower overall than the afternoon concentrations (fig. 5.7A). This is consistent with electronic monitoring data (fig. 5.6) showing 1 to 2 hours in the morning when levels were still at background concentrations, whereas all afternoon concentrations were above background levels. On six sampling mornings, differences in ozone concentrations between the affluent and disadvantaged community parks were significant at $p < 0.05$, and there were two examples in which the difference was slightly more than $p < 0.07$ (eight samples total). In five cases, the disadvantaged community parks had significantly higher ozone concentrations than affluent community parks, but there were three events at which ozone concentrations in affluent community parks

were higher during morning observations. During afternoon observations, there were 7 days when ozone concentrations at the disadvantaged community park were significantly higher (all $p < 0.05$), and 2 days when the affluent community parks were higher.

Teasing out human exposure to air pollution is difficult. As this study has shown, simply looking at the ambient concentrations whether using electronic data or passive data can yield only part of the story. Personal monitoring is a step toward certainty, but the human body is a complicated surface to monitor. In fact, the body is not a surface at all; it has its own biophysical chemistry that controls how it responds to the elements. However, we think that this study represents the best effort yet in estimating ozone exposures experienced by individuals in urban parks.

On 9 of 15 weekend days, there were significant differences between the different parks. On only 3 of 15 weekend days, the differences were not significant. This is consistent with the known phenomenon of the "weekend effect" on ozone concentrations in the South Coast Air Basin (Chinkin et al. 2003). Weekend ozone concentrations have been reported as much as 55 percent greater than weekday concentrations. This is not unique to Los Angeles. Many major urban areas have the same phenomenon (Wolff et al. 2013). Los Angeles is unique in that while most urban centers have seen a reduction or loss of the weekend effect, Los Angeles has maintained that pattern (Wolff et al. 2013). The cause is attributed to a change in the proportion of ozone-forming precursors. Emissions, in general, are lower on weekends, but NO_x is proportionally higher than VOCs. Nitrogen oxides are a precursor to the formation of ozone, but they also participate in degradation reactions, particularly after dark (Chinkin et al. 2003, Wolff et al. 2013). Our data support the notion that ozone is a more chronic problem during weekends when people are more likely to engage in outdoor activities.

Although the trend is for a higher ozone concentration to occur in the disadvantaged communities, the data are not entirely consistent. An analysis of effect size pointed out that effects were overwhelmingly stronger in parks in the disadvantaged than in the affluent community parks (table 5.3). The parks in affluent areas had one event at which ozone levels were higher than the comparison disadvantaged community park, and Crestwood Hills had two such events. However, because the effect size differential was higher in the disadvantaged community parks than in the affluent community parks, the health risk for outdoor recreationists is higher in the disadvantaged communities than in affluent communities.

Implications

The degree of explicit exposures of people to ambient pollution are difficult to measure. On a daily basis, people move from outdoor spaces to transportation

Table 5.3—Result from the Cohen's d effect size determination

Morning measurements					Afternoon measurements				
Date visited	**Park**	**P**	**Cohen's d**	**Delta**	**Date visited**	**Park**	**P**	**Cohen's d**	**Delta**
Disadvantaged community parks higher					*Disadvantaged community parks higher*				
27 Aug.	Crestwood Hills				8 Sept.	Crestwood Hills			
	Fernangeles	0.0017	5.66	6.116		Wilmington	0.021	5.062	5.641
23 Aug.	Crestwood Hills				13 Sept.	Barrington			
	Sun Valley	0.074	2.863	2.153		Sun Valley	0.004	8.218	9.95
6 Sept.	Barrington				14 Sept.	Barrington			
	Fernangeles	0.002	4.35	13.284		Fernangeles	0.001	2.428	2.251
10 Sept.	Coldwater Canyon				19 Sept.	Laurel Canyon			
	Sun Valley	0.0003	3.809	2.996		Sun Valley	0.0001	17.139	27.798
20 Sept.	Laurel Canyon				27 Sept.	Coldwater Canyon			
	Wilmington	0.0007	7.79	6.877		Fernangeles	0.000004	13.147	14.505
Affluent community parks higher					*Affluent community parks higher*				
30 Aug.	Crestwood Hills				5 Sept.	Crestwood Hills			
	Banning	0.079	2.443	1.824		Fernangeles	0.02	3.873	3.273
1 Sept.	Barrington				26 Sept.	Laurel Canyon			
	Sun Valley	0.002	6.094	5.919		Banning	0.005	5.333	5.619
4 Sept.	Coldwater Canyon								
	Wilmington	0.0005	2.394	2.019					

Note: Data are missing for Aug 29.

enclosures to indoor spaces, all of which have different ambient concentrations of air pollutants, differing air movement patterns—and as ours and other research suggest, differing patterns of exposure simply owing to the aerodynamics of the human body. Quantifying human exposure to air pollution is difficult. As these experiments have shown, studying ambient concentrations of ozone using electronic or passive data yields only partial understanding. Personal monitoring is a step toward higher accuracy, but the human body has its own biophysical chemistry that influences its response to the elements.

The approach used in these studies demonstrates the importance of in-situ monitoring of ozone exposure for individuals, particularly where exposure in outdoor recreation settings may be essential to better understanding the full array of health benefits and risks experienced by outdoor recreationists. Personal ozone monitoring as outlined here is likely of interest across a variety of settings and purposes. Although monitoring of ambient pollution levels will continue to be critical to evaluate effective health standards, studies such as these provide an important link between atmospheric concentrations and human exposure.

Our findings also point to the importance of gathering multiple days of data, paired across the settings one is aiming to contrast, within the same periods. Daily and hourly variation in ozone concentrations are well known, but the influence of buildings, vegetation, and prevailing winds in urban settings causes significant differences in local concentrations even between sites a few miles apart. Assumptions about variations in ozone concentrations by day and time based on a single monitoring station may produce undesirable confounds that mask important similarities and differences across communities.

The effects of topography play a role in ambient concentrations when one is comparing ozone exposures in urban versus rural settings. However, the extended site data presented here indicate that such topographic influences can occur within the urban setting. Residents in the affluent canyon areas of Los Angeles may be at elevated exposure to ozone because of physiographic influences on air movement that result in the settling of ozone into canyon zones.

Stationary and in-situ personal monitoring combined in this study provided valuable information for varying patterns of exposures, all of which may help inform our understanding of the relationship between air quality and human health and well-being. Differential exposure to ozone risks was evident when considering the evidence drawn from the affluent and disadvantaged communities, further adding to the body of evidence suggesting that additional attention to disadvantaged communities is warranted for an informed response to mitigation and adaptation for climate change effects.

Chapter 6: Synopsis

Exposure to ozone pollution has serious health risks. Damage to lungs and impacts to cardiovascular health are of particular concern for vulnerable populations including children, the elderly, and the disadvantaged whose health may already be compromised. The current regulatory standards for ozone exposure were established to protect human health; however, direct monitoring of individual exposure is seldom done. Our data suggest that outdoor recreationists were exposed to several times the regulatory standard of 65 ppb for vulnerable populations on a number of observed days and locations.

The development of the monitoring approach described here had three iterations. Experiment 1 was a proof of concept and exploration of potential system errors. Experiment 2 was a more extensive use of personal monitoring to evaluate potential differences in ozone exposure based on differing urban structure, but in relative proximity. We know that significant differences in tree cover, industrial activity, and housing were related to the differences in socioeconomic strata. We did not know how highly variable ozone concentrations would be from one day to the next, making passive ozone concentrations taken on different days across communities impossible to compare directly. Furthermore, the codependence of affluent communities being located along the coast where the air tends to be cleaner and disadvantaged communities being located inland where air quality typically is poorer prevented robust interpretation of the data. Our original impetus for moving the passive monitors from the observer's torso to the head was to further reduce cross-sampler variability. However, we also speculate that placing the passive monitor on a hat might better represent the ozone exposure to the mouth and nose and thus the respiratory system.

Experiment 3 added two new parks in an affluent inland community and two new parks in a disadvantaged coastal community to address the codependence of clearer air at the coast and higher air pollution loads inland. Observation periods were paired so that an affluent community was matched to a disadvantaged community for each of the observation periods. This greatly increased the sensitivity of the study.

The most remarkable modification was placing the passive samplers on the observer's head. Although we expected better replication because of more uniform air circulation, we did not expect the apparent exposure to increase by nearly an order of magnitude. Upon further study, it seems clear that while humans are moving through the environment, the environment is constantly moving around their bodies. Unlike the effort to monitor "ambient conditions" in which samplers are protected from the direct impacts of wind because diffusion is the primary mechanism for ozone to chemically react with the treated filters, when it comes to

human exposure, wind, body movement, and position are the mechanisms by which ozone is delivered to the respiratory system.

Several other protocol modifications could be made if the study were to continue, such as the following:

- Make direct comparisons of samplers worn on both the head and the torso by the same individual.
- Add meteorological data from each of the parks during the observation period (to clarify the influence of weather on ambient ozone variability).
- Explore geographic and topographic effects more broadly (to assist in understanding potential risks to outdoor recreationists and lead to better mitigation measures).

The future forecast for southern California is hotter, dryer, and longer summers. Research suggests that this will likely lead to increases in ozone concentrations regionwide, with areas currently experiencing high ozone levels likely to experience the largest increases in ozone concentrations. Because these high ozone areas are frequently situated in disadvantaged communities where health concerns are already elevated (Winter et al. 2019), it seems likely that increased ozone concentrations will add to the health burdens of recreationists who reside in these communities.

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