The Harvard Southern California Chronic Ozone Exposure Study: Assessing Ozone Exposure of Grade-School-Age Children in Two Southern California Communities

Alison S. Geyh,[1] Jianping Xue,[2] Halûk Özkaynak,[3] and John D. Spengler[4]

[1]Health Effects Institute, Cambridge, Massachusetts, USA; [2]Genetics Institute, Cambridge, Massachusetts, USA; [3]U.S. Environmental Protection Agency, Research Triangle Park, North Carolina, USA; [4]Harvard University School of Public Health, Boston, Massachusetts, USA

The Harvard Southern California Chronic Ozone Exposure Study measured personal exposure to, and indoor and outdoor ozone concentrations of, approximately 200 elementary school children 6–12 years of age for 12 months (June 1995–May 1996). We selected two Southern California communities, Upland and several towns located in the San Bernardino mountains, because certain characteristics of those communities were believed to affect personal exposures. On 6 consecutive days during each study month, participant homes were monitored for indoor and outdoor ozone concentrations, and participating children wore a small passive ozone sampler to measure personal exposure. During each sampling period, the children recorded time–location–activity information in a diary. Ambient ozone concentration data were obtained from air quality monitoring stations in the study areas. We present ozone concentration data for the ozone season (June–September 1995 and May 1996) and the nonozone season (October 1995–April 1996). During the ozone season, outdoor and indoor concentrations and personal exposure averaged 48.2, 11.8, and 18.8 ppb in Upland and 60.1, 21.4, and 25.4 ppb in the mountain towns, respectively. During the nonozone season, outdoor and indoor concentrations and personal exposure averaged 21.1, 3.2, and 6.2 ppb in Upland, and 35.7, 2.8, and 5.7 ppb in the mountain towns, respectively. Personal exposure differed by community and sex, but not by age group. *Key words*: children, chronic, exposure, ozone, personal, sampler, Southern California. *Environ Health Perspect* 108:265–270 (2000). [Online 4 February 2000]
http://ehpnet1.niehs.nih.gov/docs/2000/108p265-270geyh/abstract.html

Almost three decades ago, in response to the Clean Air Act of 1970, the U.S. Environmental Protection Agency promulgated National Ambient Air Quality Standards (NAAQS) (*1*) for six air pollutants: ozone, total suspended particles, nitrogen dioxide, sulfur dioxide, carbon monoxide, and lead. At that time, it was generally believed that only residents of Southern California were at risk for exposure to high ozone concentrations. Now almost every statistical metropolitan area in the United States has reported violations of the 1979 ozone standards of 0.12 ppm for 1 hr during a single year. In 1995, 50 cities across the United States exceeded the air quality standard one or more times (*2*). In 1997, the NAAQS for ozone was changed to an 8-hr integrated value of 0.08 ppm. Compliance will be based on 3 years of monitoring, where the fourth highest 8-hr average in a calendar year cannot exceed 0.08 ppm. Analysis in anticipation of the new standard indicates that even more Americans will be living in areas that exceed healthy levels (*3*).

Chamber studies and other acute exposure studies suggest that short-term effects of ozone on respiratory function and sensory irritation are reversible. However, only a few investigations have studied the chronic effects of ozone exposures over months and years. Using ambient ozone data collected from local monitoring sites, Schwartz et al. (*4*) reported highly significant ozone-associated reductions in lung function for people living in areas where annual ozone concentrations exceeded 40 ppb. Time–series analysis of daily mortality in Los Angeles showed an association with ozone concentration that was significant for both respiratory and cardiovascular-related deaths (*5*). Further, the work of Burnett et al. (*6*) in Ontario, Thurston et al. (*7*) in New York (*7*), and White et al. (*8*) in Atlanta are consistent in showing an association among contemporary measures of ambient ozone and hospital admissions, particularly for asthma.

Although these studies suggest a chronic effect for ozone, they are still limited by a lack of understanding of the relationship between ambient measurements and personal exposures. Several questions about chronic ozone exposure remain unanswered. The relationship between ambient ozone and personal exposures of individuals living in a community has not been adequately addressed, and the interpersonal variability in ozone exposures that are expected because of behavior, housing characteristics, and spatial differences in ozone concentrations has not yet been quantified.

Until recently, collecting personal ozone exposure information has been difficult. Only ultraviolet (UV) photometric or chemiluminescence continuous ozone monitors have been available for ozone concentration measurements and they are too heavy and cumbersome to be carried around by individuals for personal monitoring purposes. Small lightweight passive ozone exposure monitors, however, are now available. These monitors make personal and microenvironmental monitoring feasible (*9–11*). The Harvard passive ozone sampler is one such device that depends on the reaction between ozone and the nitrite ion for ozone concentration measurement (*11*). Over the last several years, short-term personal ozone exposure studies have been carried out by several researchers using this monitor (*12–15*). These studies demonstrated the feasibility of monitoring personal exposure of both children and adults for periods of up to 1 week.

The purpose of this study was to profile personal exposure to ozone over a time period that would provide information for the discussion of potential chronic effects of exposure to ozone. Data obtained from this work will be used to develop a model for estimating annual personal ozone exposure. The study was designed to measure exposure over a time period that would capture seasonal variations in ambient ozone concentrations and in locations which would demonstrate the impact of geographical location on exposure. The Harvard Southern California Chronic Ozone Exposure Study measured personal exposure to, and the indoor and outdoor ozone concentrations of, elementary school children for 12 months (June 1995–May 1996). Two

Address correspondence to J.D. Spengler, Harvard University School of Public Health, Department of Environmental Health, 665 Huntington Avenue, Room 1305, Boston, MA 02115-6021 USA. Telephone: (617) 432-1255. Fax: (617) 432-4122. E-mail: spengler@hsph.harvard.edu

We thank A. Fox, L. Sanchez, C. Bench, L. Kole, N. Scopen, M. Simun, M. Palmer-Rhea, D. Belliveau, D. Burlow, and J. Arnold. We also thank the University of Southern California Department of Preventive Medicine. Most importantly we thank the children and their families for their participation and hard work; without them this study could not have succeeded.

This work was supported by National Institute of Environmental Health Sciences grant R01-ES06370 at the Harvard School of Public Health.

Received 11 May 1999; accepted 21 September 1999.

communities were selected because of certain characteristics that were believed to affect personal exposures. Neither of the communities was in compliance with the NAAQS for ozone. In this first paper from the study, we present the study methods and descriptive results. Annual and monthly ozone personal exposures are examined within and between communities. Age, sex, housing, and community factors that could potentially impact personal exposures are also presented to derive a first level of understanding of the variables that could be important in subsequent modeling efforts.

Methods

The communities selected for the study were Upland and several neighboring mountain towns in San Bernardino County, California. San Bernardino County is on the eastern edge of the Los Angeles air basin. Upland, elevation approximately 0.15 km above sea level and 80.5 km east of Los Angeles, was chosen as representative of Southern California communities with moderate to high ambient ozone levels. The mountain towns are approximately 50 km further east of Upland and were selected because they experience some of the highest ambient ozone concentrations in the country. The mountain towns are located between 1.2 km (Crestline) and 1.8 km (Running Springs) above sea level. Community selection decisions were based on 1994 ambient ozone data, which showed that the mountain locations experienced consistently higher ozone concentrations than Upland (*16*). Both communities have a distinct annual ozone distribution, with the summer months typically 2.5 times the winter month averages.

Study design. Children were recruited from elementary schools. After a presentation at their school, the children were recruited for participation by a questionnaire and a letter to their parents. Approximately 4,300 children were contacted. Of these, 634 returned questionnaires with a positive response to study participation. From this group, 224 children from 156 homes were selected. These children were in grades 1–5. The cohort was not intended to be a random sample. Because the study period was 12 months, children who were more likely to complete the study were chosen. Children were selected if they responded enthusiastically with additional comments on their questionnaire and/or if the parents requested participation. For purposes of a parallel study involving preschool-aged children, children were selected if they had siblings 4 years of age or younger. Most respondents indicated that they had gas appliances. To investigate the effect of cooking fuel type on exposure, children were preferentially selected if their home had an electric cooking range (80% had gas and 20% had electric). Selection on the basis of home air conditioning (AC) was not possible. Of the mountain homes, only 1.7% had AC, whereas 93% of the homes in Upland had AC. The initial cohort included 119 females and 105 males.

Personal, indoor, and, outdoor ozone concentrations were measured each month for 12 months starting 7 June 1995 and ending 29 May 1996. Personal samplers were worn on the chest, clipped directly to outer clothing, for 6 consecutive days each month. Samplers were worn continuously, except when the participant was sleeping, bathing, swimming, or engaged in an activity such as soccer, for which wearing the sampler was not allowed. During these times they were placed nearby in an open area. Indoor and outdoor ozone concentrations at participants' homes were monitored using passive ozone samplers. Indoor samplers were clipped to stands supporting small fans and were placed upwind of the fans, which provided constant air flow across the collection face of the sampler. Samplers were installed in the room where the family reported spending a large part of their time at home. Fan stands were placed on bookshelves or tabletops, situated so that the fan was drawing air from the center of the room; we avoided placing fan stands opposite of frequently opened windows and doors, working fireplaces, or ceiling fans. Outdoor samplers were located in the back of homes in an open area not covered by a tree canopy or roof overhangs. Samplers were placed at least 2 m off the ground, usually attached to deck railings or fence posts, and always protected by a polyurethane cap.

Each month the participating children wore a passive ozone sampler for 6 consecutive days (approximately 144 hr) and recorded their activities on a structured diary form. During the same 6 days, indoor and outdoor samplers were placed at their homes. Diaries were divided into 30-min increments across a 24-hr time period (the increment from 0000–0600 hours was 1 hr). A child was given one diary page for each day of sampling. The diary was divided into four categories: indoor, outdoor, travel, and activity. The children were asked to indicate whether they were at home, school, some other place, or traveling under the location categories. They gave a brief description of the actual activity for each time period, e.g., playing basketball, studying, or eating in a restaurant, and estimated travel time under the activity category.

The study population in each community was divided into four cohorts. Each cohort was monitored once each month; therefore, monitoring all of the children in each community required 4 weeks. The order in which the four cohorts were monitored throughout the month remained the same for the entire study year. During each study week, sampling began on Wednesday and concluded the following Tuesday. Field technicians visited homes during times when the children were present. The children were given a sampler to wear and a time activity diary to record their 30-min activities. At each home, personal, indoor, and outdoor samplers were deployed within approximately 5 min of each other. The field technician returned 6 days later to collect the samplers and review the diaries with the child and the parent.

We conducted sampling during 46 of the 48 weeks in the study year. No sampling was carried out during the first week of June 1995, when staff members were confirming participation with study families, and during the 1995 Christmas holiday week.

We encouraged participation with $50 savings bonds awarded at the end of 6, 9, and 12 months. At the beginning of the 1995–1996 school year, the effort of the participating children was acknowledged at school assemblies, where they were presented with study tee shirts. Children who completed at least 10 months of the study were awarded certificates of completion. During the study year, the field staff rewarded children with small homemade treats and demonstrated appreciation for hard work by attending birthday parties, soccer games, and school events.

Each week, approximately 15% of all samplers used in the field were set aside as field blanks. Blanks were handled by exposing them briefly to indoor air, returning them to their plastic bag and amber canister, and then leaving them at room temperature during the 6 sample days at a field technician's home. In addition, approximately 15% of all samplers each week were divided equally between indoor, outdoor, and personal exposures, and were exposed as duplicates.

Sampling method. Continuous ambient ozone measurements were obtained from two monitoring stations operated by the South Coast Air Quality Management District in Diamond Bar, California. These stations are in Upland and Crestline, one of the mountain communities. The Upland station is in a trailer park on the eastern edge of the town, approximately 2 km from the San Bernardino Freeway. The Crestline station is on the shore of Lake Gregory, a recreational lake approximately 1.2 km above sea level. The UV photometric ozone analyzers (Dasibi 1008-RS; Dasibi Environmental Corporation, Glendale, CA) used have a 1-ppb limit of detection (LOD).

Integrated personal, indoor, and outdoor ozone measurements were made using the

Harvard passive ozone sampler (*11*). All samplers were prepared at the Harvard School of Public Health (HSPH; Boston, MA) 1 week before deployment in the field. For shipping, samplers were sealed in resealable plastic bags, then placed in amber canisters. The samplers were shipped cooled to California by overnight delivery and used in the field the next day. At the end of the sampling period, the samplers were retrieved, stored in refrigerators, and then returned to HSPH in cold containers by overnight delivery. The samplers were refrigerated until they were analyzed. All of the samplers were analyzed between 1 and 3 weeks after returning to the HSPH.

Harvard passive ozone sampler. The Harvard passive ozone sampler is composed of a Teflon barrel containing two glass fiber filters, one at each end of the barrel (Ogawa and Co. USA, Inc. Pompano Beach, FL), as shown in Figure 1. The filters were coated with a previously described nitrite-containing solution (*11*). They were held in place by perforated endcaps that act as diffusion barriers. To deploy the sampler, the barrel was attached to a plastic badge equipped with a metal clip. The clip was used to secure the sampler to the sampling location.

The sampler collects ozone using the oxidation reaction of nitrite by O_3 to form nitrate. The average ozone concentration measured by the sampler was calculated from amount of NO_3^- accumulated, which was determined by ion chromatography (Dionex model 2000i; Dionex Corporation, Sunnyvale, CA), and the appropriate effective collection rate (ECR). Ozone concentrations were calculated as follows:

$$C_{O_3} = \frac{N \times V \times MW_{O_3} / MW_{NO_3} \times R}{ECR \times K \times MW_{O_3} \times T} \qquad [1]$$

where C_{O3} is the integrated ozone concentration in parts per billion; N is the corrected nitrate concentration (sample minus average blank, in milligrams per milliliter); V is the extraction volume in milliliters, 5 mL; MW_{O3} and MW_{NO3} are the molecular weights of ozone and the nitrate ion, respectively, in milligrams per micromole; R is the conversion factor 106 in cubic centimeters per cubic meter; ECR is the effective collection rate in cubic centimeters per minute; K is the conversion constant 0.0409 mg/(ppb/m^3) determined at 298 K and 1 atm; and T is the exposure time in minutes.

Interferences. Interferences from other pollutants are a potential concern with the chemistry of this method. Possible interferences include NO_2, HONO, PAN, H_2O_2, and SO_2, and interference testing was carried out in the environmental exposure chamber at the University of California, Riverside. When passive samplers were exposed to high concentrations of these potentially interfering species for relatively long time periods, researchers at the University of California, Riverside, found little interference from NO_2, HONO, PAN, and SO_2. The researchers found significant interference from H_2O_2 in the high-concentration range; however, the effect under ambient conditions is likely to be negligible. HNO_3, which is expected to present a positive interference, was not tested because of the difficulty of generating a stable nitric acid atmosphere. However, at concentrations typical of those found in Southern California, the interference for this method would be approximately 5% of measured ozone (*17*).

ECR. The theoretical ECR for the passive sampler is 21.8 cm^3/min. Under constant wind conditions, sampler performance is not affected by the large changes in temperature or relative humidity in ambient air and typical of residential locations. The precision of the passive sampler is approximately 10% over a wide range of concentrations. However, wind tunnel tests show that the collection rate of the sampler is significantly affected by variations in face velocity (*18*).

For outdoor sampling, we solved the effect of varying face velocity by using a protective cap. Use of the protective caps with this sampler in different studies gave an ECR close to theoretical: 21.6 cm^3/min (*19,20*). The ECR used in this study for determining outdoor ozone concentrations was also 21.6 cm^3/min.

It was important to ensure sufficient air movement across the face of the passive sampler for indoor air sampling. To control the face velocity for indoor sampling, we placed the sampler upstream of a small fan on a stand. The fan stand consisted of a small box fan supported by a lightweight metal frame. The passive ozone sampler was attached to the fan stand so that the sampling faces were parallel to the air flow. This method was similar to the timed exposure diffusion sampler (TEDS) used by the California Air Board in a Los Angeles study (*17*), but was less complicated and costly. Because face velocities for the indoor sampler were similar to those of the TEDS, which have an ECR of 21.3 cm^3/min, the same ECR was used to determine indoor ozone concentrations for this study.

The ECR used for personal sampling was determined from a controlled chamber experiment. To determine the collection rate of personal sampling, Liu et al. (*13*) studied five adult subjects who wore four passive samplers at different body locations while sitting in an exposure chamber. Using ozone concentrations measured in the chamber by continuous UV photometric ozone monitors, the ECRs for passive samplers at each body location were determined; they ranged from 17.7 ± 2.3 to 10.3 ± 2.9 cm^3/min. The mean ECR for samplers at all body locations was 14.8 ± 2.9 cm^3/min, which we used in this study.

Quality assurance. Ozone concentrations were calculated according to Equation 1. Background blank values, determined from week-specific field blanks, were subtracted from the sample nitrate measurements. The LOD was determined at 3 SDs of the average nitrate concentration from field blanks. The LOD based on a 144-hr exposure was 1.0 ± 0.57 ppb, with the weekly LODs ranging from 0.3 to 2.8 ppb. LODs for this study corresponded to the range reported by others [0.5–2.0 ppb; (*13,14,21*)].

Precision was determined from 602 duplicate comparisons. Figure 2 shows a comparison of the duplicate samples and the overall correlation coefficient (r^2 =0.95). We calculated precision by the root mean square estimate method and reported it as a percentage. The precision was 9% for personal (n = 158), 12% for indoor (n = 239), and 4% for outdoor (n = 205) samplers. The

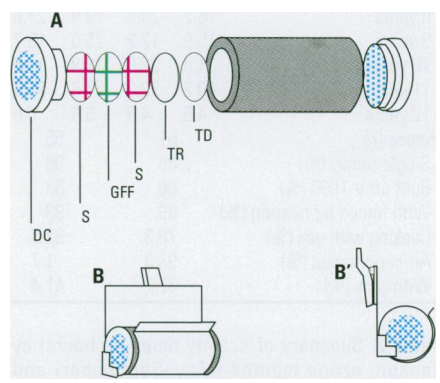

Figure 1. (*A*) Configuration of the Harvard passive ozone sampler. Abbreviations: DC, diffusion cap; GFF, coated glass fiber filter; S, screens that support the coated glass fiber filter; TR and TD, Teflon supports for the screens and filter, respectively. (*B*) Front view of the assembled badge. (*B'*) Side view of the assembled badge.

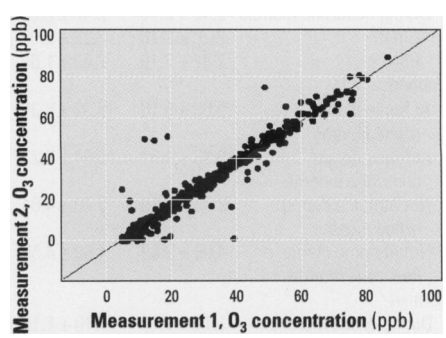

Figure 2. Comparison of the duplicate samples and the overall correlation coefficient. Slope = 0.99, intercept = 0.42, r^2 = 0.95, n = 602.

percentage of all samplers that fell below the LOD was 10.4% (6.1% personal, 19.7% indoor, and 0.05% outdoor). We used a value of 50% of the LOD, or 0.5 ppb, for the following analyses.

From June to August 1996, 30 triplicate sets of passive samplers were placed at the local monitoring stations. The sampling period was 144–168 hr. The mean passive sampler concentrations for each sampling period showed excellent agreement with the time-averaged hourly ozone concentrations, with an overall bias of + 3% and an r^2 = 0.995.

Statistical analysis. Statistical analyses of the data included simple least squares regression to estimate bias and the direction of the bias. Associated correlation coefficients were also determined. We used the *t*-statistic to test for differences in sample means. We assumed that the sample distributions were approximately normal, with missing values randomly distributed across seasons, sex, and location.

Table 1. Characteristics for children participating in the study for at least 6 months.

Characteristic	Upland Boys	Upland Girls	Mountain Boys	Mountain Girls
Number of subjects	40	44	34	51
Participants by age (%)				
6 years	2.3	0.0	2.8	2.0
7 years	9.1	14.6	11.1	15.7
8 years	18.2	24.4	13.9	21.6
9 years	15.9	12.2	25.0	15.7
10 years	31.8	22.0	13.9	11.8
11 years	18.2	22.0	27.8	33.3
12 years	4.5	4.9	5.6	0.0
Homes (*n*)	61		55	
Single family (%)	85		98	
Built after 1960 (%)	80		83	
With forced air heating (%)	89		93	
Cooking with gas (%)	78.3		82.8	
Air-conditioned (%)	93.3		1.7	
With pets (%)	37.7		41.4	

Table 2. Summary of activity times (in hours) by season: ozone months (May–September) and nonozone months (October–April).

Average daily time	Upland[a] (mean ± SD)	Mountain[b] (mean ± SD)
Outdoors		
During ozone months	3.47 ± 2.70	3.88 ± 3.15
Girls	3.21 ± 2.67	3.65 ± 3.23
Boys	3.78 ± 2.72	4.16 ± 3.11
During nonozone months	2.49 ± 2.18	2.42 ± 2.04
Girls	2.20 ± 2.16	2.22 ± 2.03
Boys	2.87 ± 2.18	2.69 ± 1.69
Indoors		
At home during ozone months	15.69 ± 5.10	15.32 ± 5.35
At home during nonozone months	16.53 ± 4.51	15.94 ± 5.23
Not at home during ozone months	3.73 ± 4.19	3.70 ± 4.64
Not at home during nonozone months	4.06 ± 3.89	4.60 ± 4.58
Transit		
During ozone months	1.11 ± 1.17	1.10 ± 1.15
During nonozone months	0.91 ± 1.01	1.05 ± 1.09

[a]Number of daily diaries = 5,325. [b]Number of daily diaries = 5,004.

Results

During the yearlong study, 25% of the study subjects did not meet the minimum requirement of at least six sampling periods of valid measurements and completed forms. Most of the children who dropped out (28 of 40) left in the first half of the study. One hundred eighty-four children completed the study, but of those, 15 lacked housing characterization questionnaires. For a variety of reasons, obtaining housing characteristics information from these 15 households proved problematic for the field staff, who ultimately failed to secure completed questionnaires from this group. Data from 169 children were used in the analysis. These children lived in 116 homes, of which 61 were in Upland and 54 were in the mountain communities.

The average number of measurements per child across the study year, 10.7 ± 0.3, was independent of sex, age, or location. We found a similar result by season. Upland children averaged 4.4 ± 0.8 measurements during the ozone months (May–September) and 5.7 ± 1.1 measurements during the nonozone months (October–April), whereas mountain children averaged 4.6 ± 0.8 (ozone months) and 6.3 ± 1.2 (nonozone months) measurements.

Table 1 provides a summary description of the study population. Several housing factors are noted. During the ozone months, children living in the mountains were outdoors longer than children from Upland. Boys in both communities spent on average 30 min longer outdoors than girls. During the nonozone months, children spent on average 1 hr less outdoors than they did during the ozone months. Table 2 summarizes features of the children's diaries.

Table 3 summarizes the seasonal averaged ozone concentrations for outdoor, indoor, and personal passive sampling.

Outdoor. Average monthly ozone concentrations from all homes and from each central site monitoring station are shown in Figure 3. Outdoor monthly concentrations were derived from the average of all outdoor passive measurements collected over all four sampling periods each month at participant homes in each community. The average monthly ambient central site concentration for each location was determined from data retrieved from the Aerometric Information Retrieval System (*22*). The seasonal pattern of ozone in Southern California is evident and is consistent with historical data. Although there is spatial variability within each community, the Mountain–Upland differences persist.

In Upland, monthly averages of the outdoor home ozone concentrations were approximately 13% higher that the Upland monitoring station measurements (r = 0.99). The average of the home outdoor concentrations was consistently higher than the monitoring station throughout the study year. In the mountains, monthly averages of the outdoor home concentrations during the ozone months were approximately 4% lower as compared to central monitoring station average monthly measurements. However, during the nonozone months the relationship between the monthly outdoor home and ambient concentrations was similar to that in Upland, with the home outdoor measurements on

Table 3. Descriptive statistic of seasonal integrated outdoor and indoor ozone concentrations and personal ozone exposure levels divided by community.

Statistic	June–September 1995 and May 1996: Upland	June–September 1995 and May 1996: Mountain	October 1995–April 1996: Upland	October 1995–April 1996: Mountain
Outdoor (ppb)				
Samples (*n*)	383	403	530	570
Mean	48.2	60.1	21.1	35.7
Median	47.6	57.6	19.3	35.8
SD	12.2	17.1	10.7	9.3
Minimum	9.1	3.9	0.5	13.6
Maximum	82.5	160.1	64.8	65.6
Indoor (ppb)				
Samples (*n*)	386	412	531	569
Mean	11.8	21.4	3.2	2.8
Median	9.5	19.7	1.5	0.6
SD	9.2	14.8	3.9	4.2
Minimum	0.5	0.5	0.5	0.5
Maximum	41.6	67.8	34.9	29.5
Personal (ppb)				
Samples (*n*)	345	367	479	520
Mean	18.8	25.4	6.2	5.7
Median	17.6	24.0	4.7	4.2
SD	10.1	13.4	5.4	5.1
Minimum	0.5	0.5	0.5	0.5
Maximum	62.6	72.3	40.7	31.2

average 15% higher than measurements made at the mountain monitoring station.

Outdoor home ozone concentrations in the mountain communities were higher than those found in Upland. During the high-ozone months, mountain concentrations were on average 20% higher than in Upland (two-tailed t-test, $p < 0.01$). During the nonozone months, concentrations in the mountains were on average 60% higher (two-tailed t-test, $p < 0.0001$).

Indoor. During the ozone months, average weekly indoor home concentrations in the mountain homes were almost 2 times those found in Upland (two-tailed t-test, $p < 0.01$). During the nonozone months there was no difference between the two communities (two-tailed t-test, $p > 0.05$). Figure 4 shows average monthly indoor concentrations across the entire study year for both communities.

Personal. During the ozone months, monthly average personal exposure measurements differed by community as well. Mountain community participants were exposed to, on average, 35% (two-tailed t-test, $p < 0.01$) more ozone than participants in Upland. In the nonozone months, there was no significant difference in average exposure (two-tailed t-test, $p > 0.05$). Figure 5 shows the average monthly personal concentrations across the entire study year in both communities.

Although there were differences among the four cohort groups within a month or season, the overall annual personal exposure concentrations were not significantly different. There were differences in exposure based on sex. Boys had higher personal exposures than girls independent of location of their homes or housing factors. Table 4 shows that this difference was larger when considering just the summer months.

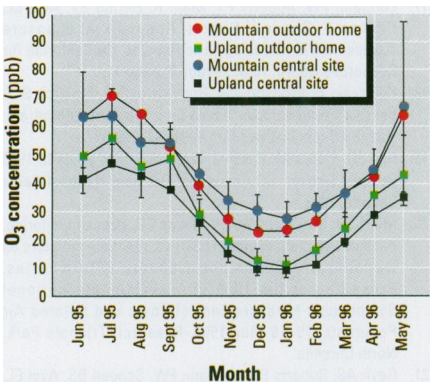

Figure 3. Ambient monthly ozone concentrations at central sites and across homes (SD) in each community.

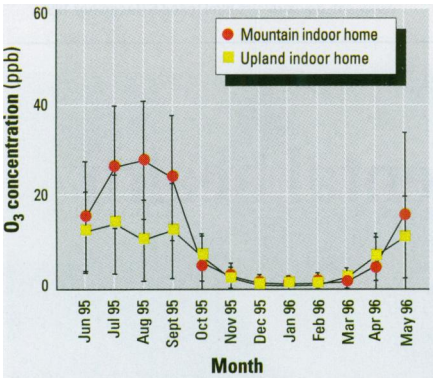

Figure 4. Indoor monthly ozone concentrations across homes (SD) in each community.

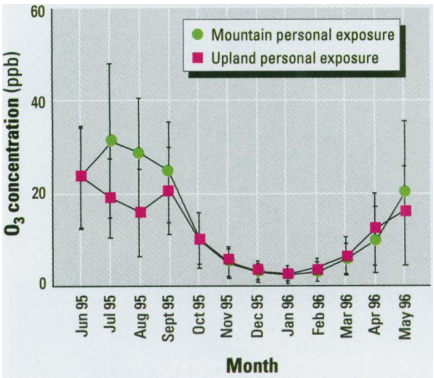

Figure 5. Personal monthly ozone concentrations across homes (SD) in each community.

Discussion and Conclusions

This study represents the first longitudinal estimation of exposure to ozone over a 1-year period. Personal, indoor, and outdoor ozone measurements were successfully collected for 184 children across a 12-month period in two high-ozone communities in Southern California. In addition to wearing a personal sampler for 6 consecutive days each month, the children recorded their activities during each day the sampler was worn. We collected information characterizing the home of each participant. Of the 184 children who completed the study, results from 169 were used in the analyses. We compared differences in ozone levels and exposure between communities in each season by outdoor and indoor ozone concentrations and by personal exposure. Personal exposure was evaluated between communities by sex and age.

The two communities were selected because of a large between-community difference in ambient ozone levels. Average monthly outdoor ozone concentrations measured at subject homes reflected the concentration trends reported by the local monitoring stations. The difference in ambient concentrations between communities was captured by the home outdoor measurements. In the mountains, winter concentrations remained approximately twice as high as those in Upland; monthly concentrations did not fall below 34 ppb. Spatial variation in ozone concentration across communities was reflected in the difference between the average of the home concentrations and the average ozone concentrations measured at the single-location monitoring stations. In Upland the difference between home outdoor and stationary site measurements across the study year was on average +13%. This may be a reflection of the fact that the majority of study homes were up slope and farther away from a major freeway than was the monitoring station. The difference between home outdoor measurements and stationary site measurements in the mountains varied with season. The average difference during the ozone months was -4%, whereas during the nonozone months the average difference was +15%. This difference may be an indication that the Crestline station, which was used for the mountain communities, was not a good indicator of ambient concentration for all of the mountain communities. The station was approximately 0.30 km below and 11.3 km west of Running Spring, where 24% of the participants lived.

Differences between communities were also reflected in indoor concentrations. During the ozone season, average indoor concentrations in the mountain homes were 3–17 ppb higher than in the Upland homes. This difference was due not only to higher ambient concentrations in the mountain communities, but also to differences in the way homes were ventilated. All but one of the mountain homes were ventilated by open windows, whereas all homes but six in Upland were air conditioned. As ambient levels decreased in the nonozone season months, the individual characteristics of homes influencing indoor ozone concentrations became less important. Although outdoor concentrations were higher in the mountains during these months, temperatures

Table 4. Comparison of personal exposure by sex and season.

	June–September 1995 and May 1996				October 1995–April 1996			
	Upland		Mountain		Upland		Mountain	
Personal exposure	Boy	Girl	Boy	Girl	Boy	Girl	Boy	Girl
Samples (n)	40	44	34	51	40	44	34	51
Mean	19.7	18.2	26.6	22.8	6.9	6.0	6.9	4.8
SD	7.8	6.5	8.7	9.3	2.8	2.9	3.3	2.1
Minimum	4.3	7.0	10.0	7.4	1.5	0.8	1.2	0.9
Maximum	39.4	35.2	48.9	49.7	12.6	14.9	17.9	10.5

were considerably lower in both communities and windows were kept closed. During the nonozone months there was essentially no difference in indoor concentration between the two communities.

Personal exposure also differed between communities during the ozone months. In the mountain communities, personal exposures were 0–12 ppb higher than in Upland, whereas during the nonozone months there was no difference in exposures between the two communities. In both communities boys' exposure was higher on average than girls', with boys and girls in the mountain communities experiencing higher exposures than boys and girls in Upland. We found no difference in exposure between the age groups that we investigated. This may be because children in elementary school are engaged in similar activities and have similar schedules during the school year.

Studies of chronic effects due to ozone exposure have been limited by lack of knowledge about personal exposure. The Southern California Chronic Ozone Exposure study provides personal exposure data across a time period that is relevant for understanding chronic effects and in geographical areas differing in ambient ozone concentrations. Current work is focused on developing models to estimate individual and community ozone exposure. The extent to which information about individual activities, time spent in different locations, and characteristics of participants' homes can be used to estimate exposure levels is being explored. Valid exposure models will yield estimates of ozone exposure in communities where no actual personal data are available, thus providing information for future epidemiologic studies.

References and Notes

1. Clean Air Act. National Primary and Secondary Ambient Air Quality Standards. 42 U.S.C. 7409 (1970).
2. Johnson J. News government: 1995 may be high ozone year in U.S., early data indicate. Environ Sci Technol 29:453A (1995).
3. Tony WA, ed. Which communities will be affected by the new standards? EM: Air Waste Manag Assoc Mag Environ Managers (January):19 (1997).
4. Schwartz J. Lung function and chronic exposure to air pollution: a cross-sectional analysis of NHANES II. Environ Res 50:309–321 (1989).
5. Kinney P, Ozkaynak H. Associations between ozone and daily mortality in Los Angeles County. Environ Res 54:99–120 (1991).
6. Burnett RT, Dales RE, Razienne ME, Krewski D, Summers PW, Roberts GR, Raad-Young M, Dann T, Brook J. Effects of low ambient levels of ozone and sulfates on the frequency of respiratory admissions to Ontario hospitals. Environ Res 65:172–194 (1994).
7. Thurston GD, Lippmann M, Scott MB, Fine JM. Summertime haze air pollution and children with asthma. Am J Respir Care Med 155:654–660 (1997).
8. White MC, Etzel RA, Wilcox WD, Lloyd C. Childhood asthma and ozone pollution in Atlanta. Environ Res 65(1):56–68 (1994).
9. Grosjean D, Hisham M. a passive sampler for atmospheric ozone. J Air Waste Manag Assoc 42:169–173 (1992).
10. Kanno S, Yanagisawa Y. Passive ozone/oxidant sampler with colometric determination using I_2/nylon-6 charge-transfer complex. Environ Sci Technol 26:744–749 (1992).
11. Koutrakis P, Wolfson JM, Bunyaviroch A, Froehlich SE, Hirano K, Mulik JD. Measurement of ambient ozone using a nitrite-coated filter. Anal Chem 65:209–214 (1993).
12. Liu L-JS, Koutrakis P, Suh HS, Mulik JD, Burton RM. Use of personal measurements for ozone exposure assessment: a pilot study. Environ Health Perspect 101:318–324 (1993).
13. Liu L-JS, Olson MP III, Allen GA, Koutrakis P, McDonnell WF, Gerrity TR. Evaluation of the Harvard ozone passive sampler on human subjects indoors. Environ Sci Technol 28:915–923 (1994).
14. Brauer M, Brook JR. Personal and fixed-site ozone measurements with a passive sampler. J Air Waste Manag Assoc 45:529–537 (1995).
15. Linn WS, Shamoo DA, Anderson KR, Peng R-C, Avol L, Hackney JD, Gong H Jr. Short-term air pollution exposures and responses in Los Angles area school children. J Expos Anal Environ Epidemiol 6:449–472 (1996).
16. Lurmann F. Personal communication.
17. Lurmann FW, Roberts PT, Main H, Hering SV, Avol EL, Colome S. Phase II Report, Appendix A: Exposure Assessment Methodology. Los Angeles, CA:California Air Resources Board, 1994; section 5, page 11.
18. Koutrakis P, Wolfson JM, Bunyaviroch A, Froelich S. A passive ozone sampler based on a reaction with nitrate. Health Effects Inst Res Rep 63:19–47 (1994).
19. Ray JD, Flores M. Passive Ozone Sampler Study II: 1993 Results. Denver. CO:Air Quality Division, National Parks Service, 1993.
20. Mulik JD, McClenny WA, Williams DD. Ozone monitoring: passive sampling devices (PSD) vs. real time monitors as National Dry Deposition Network (NDDN) sites. Presented at the 1995 EPA/AWMA International Symposium: Measurement of Toxic and Related Air Pollutants, 16–18 May 1995, Research Triangle Park, North Carolina.
21. Geyh AS, Roberts PT, Lurmann FW, Schoell BS, Avol EL. Initial field evaluation of the Harvard active ozone sampler for personal ozone monitoring. J Environ Anal Environ Epidemiol 2:143–149 (1999)
22. U.S. Environmental Protection Agency. Aerometric Information Retrieval System. Available: http://www.epa.gov/airs/airs.html [cited 6 December 1999].

The latest word on environmental health at your fingertips.

Check us out!

The Environmental Health Information Service (EHIS) offers online, searchable access to:

- *Environmental Health Perspectives*
- *Environmental Health Perspectives Supplements*
- National Toxicology Program Technical and Toxicity Reports
- *Report on Carcinogens*
- Chemical Health and Safety Database
- Rodent Historical Control Database

For more information on ordering see http://ehis.niehs.nih.gov/ or call 1-800-315-3010.

http://ehis.niehs.nih.gov/