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Inhalation intake of ambient air pollution in California's South Coast Air Basin

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Abstract

Reliable estimates of inhalation intake of air pollution and its distribution among a specified population are important for environmental epidemiology, health risk assessment, urban planning, and environmental policy. We computed distributional characteristics of the inhalation intake of five pollutants for a group of ~25,000 people (~29,000 person-days) living in California's South Coast Air Basin. Our approach incorporates four main inputs: temporally resolved information about people's location (latitude and longitude), microenvironment, and activity level; temporally and spatially explicit model determinations of ambient concentrations; stochastically determined microenvironmental adjustment factors relating the exposure concentration to the ambient concentration; and, age-, gender-, and activity-specific breathing rates. Our study is restricted to pollutants of outdoor origin, i.e. it does not incorporate intake in a microenvironment from direct emissions into that microenvironment. Median estimated inhalation intake rates ($\mu g\,d^{-1}$) are 53 for benzene, 5.1 for 1,3-butadiene, 8.7×10^{-4} for hexavalent chromium in fine particulate matter (Cr-$PM_{2.5}$), 30 for diesel fine particulate matter ($DPM_{2.5}$), and 68 for ozone. For the four primary pollutants studied, estimated median intake rates are higher for non-whites and for individuals in low-income households than for the population as a whole. For ozone, a secondary pollutant, the reverse is true. Accounting for microenvironmental adjustment factors, population mobility and temporal correlations between pollutant concentrations and breathing rates affects the estimated inhalation intake by 40% on average. The approach presented here could be extended to quantify the impact on intakes and intake distributions of proposed changes in emissions, air quality, and urban infrastructure.

Keywords: Diesel particulate matter; Environmental justice; Exposure analysis; Mobility; Ozone; Geographic information system (GIS)

1. Introduction

A major challenge in environmental health research and practice is accurately estimating

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pollutant exposure or intake. Environmental epidemiology relies on exposure assessment to determine dose–response relationships. Health officials use exposure levels to estimate the total health impact of air pollution. Urban planners often consider the impacts on air pollution and health of urban area attributes, such as zoning, population density, and the transportation network.

Inhalation intake—i.e., the mass of pollutant inhaled by one or more members of a population over a given period of time—is an important exposure metric. Intake is often considered to be better than emissions or ambient concentrations as a proxy for air pollution health effects (Bennett et al., 2002; Marshall, 2005; Ott et al., 2007; Smith, 1993). Air pollution policy becomes more effective at improving public health when one focuses attention on limiting intake rather than on reducing emissions or ambient concentrations without accounting for how these reductions would affect population doses. Understanding and addressing distributional issues, such as correlations between intake rate and demographic attributes such as ethnicity and income, is important for establishing equitable environmental policy goals.

Because of time and budget constraints, modeling and measurement approaches for estimating air pollution exposures for real individuals are often limited to a small number of people—typically several tens of individuals and rarely more than a few hundred individuals. Some models generate a large number of simulated individuals via Monte Carlo sampling from distributions of values for exposure-relevant attributes (see www.epa.gov/nerl for examples). Other models overlay ambient concentrations onto US Census data (e.g., Morello-Frosch et al., 2002), sometimes accounting for microenvironments (see, e.g., www.epa.gov/ttn/atw/nata). In this study, we estimate the inhalation intake rate of air pollutants of outdoor origin for ~25,000 individuals in California's South Coast Air Basin. Our approach incorporates time-location-activity survey data, which report people's location (latitude and longitude) over time; temporally and spatially resolved ambient pollutant concentrations, determined from an Eulerian photochemical air pollution model; microenvironmental adjustment factors; and breathing rates. We evaluated the results to determine (1) the statistical characteristics of the variation in intake among the exposed population, (2) the impact on estimated intake rates introduced by accounting for microenvironments, breathing rates that vary diurnally and among individuals, and changes in location, and (3) relationships between air-pollution intake rate and two demographic attributes (ethnicity and income category). The results of this study present a unique exploration of these issues because of the novel method developed; the large sample size (~29,000 person-days, representing real rather than synthetic individuals); and the inclusion of several distinct air pollutants of concern.

2. Methods

Inhalation intake of an air pollutant by an individual can be estimated as

$$I = \int_{T_1}^{T_2} Q_B(a(t)) C_{amb}(x, y, t) \gamma_{\mu(t)} \, dt, \quad (1)$$

where I is the mass of pollutant inhaled (μg) by an individual integrated over time t from T_1 to T_2 (h); $Q_B(a(t))$ is the individual's volumetric breathing rate ($m^3 h^{-1}$), which depends on that person's time-varying activity level, $a(t)$; $C_{amb}(x,y,t)$ is the ambient pollutant concentration ($\mu g\, m^{-3}$) near the individual, which is a function of location (x, y) and time; and $\gamma_{\mu(t)}$ is a dimensionless factor for each microenvironment, $\mu(t)$, that accounts for differences between the ambient concentration and the exposure concentration (attributable to ambient sources) in that microenvironment. Exposure concentration is the instantaneous average concentration ($\mu g\, m^{-3}$) in a person's breathing zone. In this work, the integral in Eq. (1) was evaluated as a series of sums over discrete time intervals chosen such that the three variables (breathing rate, ambient concentration, and microenvironmental adjustment factor) are reasonably represented as constant for the duration of each interval. The input data for these three variables, and our methods for combining them, are described next. The model was run in C++, and model results were processed using SAS and ArcGIS software.

2.1. Travel survey data

We extracted activities and locations for individual members of the study population from geocoded activity diaries in the Southern California Association of Government (SCAG) year-2000 transportation survey (SCAG, 2003). The primary purposes of this survey are to support

travel-forecast modeling and to inform infrastructure investment decisions. The sampling universe is households in the six counties of Southern California (Imperial, Los Angeles, Orange, Riverside, San Bernardino, Ventura) that have a telephone and that speak English or Spanish. Households were selected at random, using a database of listed working telephone numbers. Details about the survey are contained elsewhere (SCAG, 2003).

A total of 25,184 survey respondents spent 100% of their travel diary time within the South Coast Air Basin (SoCAB). We removed 120 records (0.5%) that contained erroneous or missing data or that yielded infeasible results, leaving the records for 25,064 individuals used in the present study. Most (21,382 out of 25,064, or 85%) of the individuals have one 24 h weekday record. The rest (3682 out of 25,064, or 15%) have a 48 h record that includes one weekend day. The 28,746 person-days in the data subset are from 11,749 households.

SCAG travel survey data were collected during three phases, corresponding to the following approximate dates: 1 March 2000 to 30 June 2000 (17 weeks), 1 September 2001 to 15 December 2001 (15 weeks), and 15 January 2002 to 30 June 2002 (24 weeks). It is not possible to match travel survey records directly to dates from the ambient concentration model results for two reasons. First, the travel survey period (year-2000 through 2002) is after the air-pollution modeling period (year-1998 and 1999). Second, the survey data provide the day-of-week and the survey phase, but not the specific date. To address these two limitations, we matched travel survey data and ambient concentration fields using a random selection process that preserves fidelity to day-of-week and time-of-year. For example, to determine the ambient concentrations for a Tuesday travel survey record in the second phase, we used air-pollution model results for one Tuesday, chosen at random, between 27 August 1998 and 16 December 1998.

2.2. Ambient concentrations

The time-dependent ambient concentration assigned to an individual was determined from that individual's location at a particular time and from the output of a spatially and temporally explicit model of the dispersion and transformation of urban toxic air pollutants. In this study, we obtained ambient concentration estimates from the CAMx air quality model (www.camx.com). The air dispersion modeling period is 1 April 1998–31 March 1999, and the location is the South Coast Air Basin modeling domain. CAMx is a three-dimensional Eulerian grid model that incorporates emissions, advection, and chemical reactions. Ground-level ambient concentrations are given as average values in each hour for each $2\,km \times 2\,km$ grid cell in the $210\,km \times 120\,km$ domain. Details about model formulation, uncertainty, and validation, and about the broader Multiple Air Toxics Exposure Study (MATES) are available elsewhere (ENVIRON, 2002; Morris and Jia, 2003; SCAQMD, 2000). The model performs reasonably well compared to other similar models and compared to available concentration data (ENVIRON, 2002; Morris and Jia, 2003).

We analyzed intake for five species modeled in CAMx: benzene, 1,3-butadiene, ozone, fine particulate matter emitted from diesel engines ($DPM_{2.5}$), and hexavalent chromium in the form of fine particulate matter ($Cr\text{-}PM_{2.5}$). These five species vary in several important pollutant attributes: primary versus secondary pollutants, mobile-source-dominated versus stationary-source-dominated emissions, and toxic versus criteria pollutants. Furthermore, pollutants in this group have been determined to be significant contributors to the total human health impact of ambient air pollution in the SoCAB (SCAQMD, 2000; US EPA, 2004).

2.3. Breathing rates

We employed age-, gender-, and activity-specific breathing rates given by Layton (1993). Age and gender are recorded in the transportation survey. Survey activity data indicate when people are exercising, but otherwise do not provide information on metabolic level. We assigned an exercise breathing-rate during time spent exercising. If the individual was at home during 11 PM–7 AM, we used sleeping breathing rates. All other activities (e.g., shopping, employment, household chores) were assigned a light-activity breathing rate. The calculated average breathing rate (units: $m^3\,d^{-1}\,person^{-1}$) for the study population is 13.1, which lies between the population average estimates of 12 and 15 given by Layton (1993) and Marty et al. (2002).

2.4. Combining travel survey data with modeled ambient concentrations

During non-travel activities, the ambient concentration assigned to a person is the ambient

concentration for the CAMx grid cell in which they are located. During transportation activities, people may travel through multiple grid cells. The survey provides the time and location for the origin and the destination of each trip, but not the route traveled. We modeled people as moving in a straight line at a constant speed from their origin to their destination. Assigned ambient concentrations during travel are the concentrations in each of the CAMx grid cells the person traverses during that trip, for the duration spent in that cell.

Fig. 1 summarizes the SCAG travel survey data in terms of the cumulative distribution of distance from home at two times (3:30 AM and 3:30 PM), and of the daily maximum distance from home. The two times correspond approximately to those with the fewest and most trips: 0.1% of trips begin during 3:00–4:00 AM, and 8.4% of trips begin during 3:00–4:00 PM (US DOT, 2003). Fig. 1 indicates that at 3:30 AM, most people (~98%) are within 1 km of home (most likely, they are at home), while at 3:30 PM, only 58% of people are within 1 km of home. During an average travel diary day, 27% of people surveyed stay within 1 km of home all day, 38% travel at least 1 km but never more than 10 km from home, and 35% travel to at least one location that is more than 10 km from home.

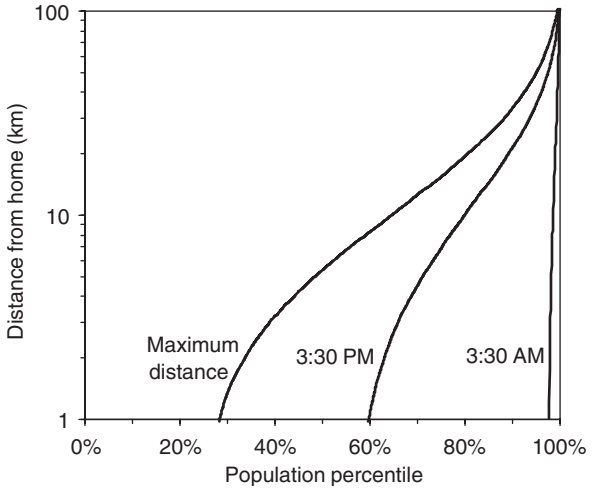

Fig. 1. Cumulative distribution of distance from home for the Southern California Association of Governments travel survey data for the 28,746 person-days simulated in the South Coast Air Basin. Three distributions are shown: at 3:30 AM, at 3:30 PM, and the daily maximum distance away from home.

2.5. Microenvironmental adjustment factors

The concept of microenvironments is used to account for times when the exposure concentration attributable to outdoor emissions is different from the ambient concentration corresponding an individual's geographic location. For example, concentrations of primary vehicle emissions such as benzene tend to be higher in a vehicle than in nearby ambient air because the in-vehicle microenvironment is in closer proximity to vehicle emissions than the average position in the 2 km × 2 km modeling grid cell. In contrast, concentrations attributable to outdoor air pollution are lower indoors than in ambient air for pollutants such as ozone that are removed chemically or physically within indoor environments or as air migrates from outdoors to indoors. The attributable concentration in a microenvironment is estimated as the product of the ambient concentration and the applicable microenvironmental adjustment factor (MAF).

This investigation evaluated exposure to air pollution of outdoor origin and therefore did not incorporate indoor sources such as cigarette smoke or emissions from building materials. We treated individuals as always being in one of four microenvironments: outdoors, indoors in a residence, indoors in a non-residence, and in or near motor vehicles. The exposure concentration for all pollutants in the outdoor microenvironment was taken as the ambient concentration (i.e., the outdoor MAF is 1.0). Benzene and butadiene can penetrate the building envelope without significant loss. For these two gases, the time-average indoor (residential and nonresidential) concentration attributable to ambient emissions was taken to be equal to the time-average outdoor concentration, and hence the corresponding MAFs are equal to 1.0. For other species and microenvironments considered in this work, MAFs were determined stochastically, with values chosen from a distribution representing variability in the relationship between the ambient and the exposure concentrations (see Table 1).

To evaluate the MAFs for $DPM_{2.5}$ and Cr-$PM_{2.5}$ in residences, we employed a mass-balance modeling approach (Burke et al., 2001):

$$\gamma_{\text{residence, PM}_{2.5}} = \frac{Pa}{a+k}. \quad (2)$$

Here, $\gamma_{\text{residence, PM}_{2.5}}$ is the MAF (dimensionless) for ambient $PM_{2.5}$ (i.e., $DPM_{2.5}$ or Cr-$PM_{2.5}$) in

Table 1
Summary of microenvironmental adjustment factors[a]

	In-vehicle	Indoor, residence	Indoor, other
Benzene	Tri(2,4,6)[b]	1	1
Butadiene	Tri(2,4,6)[b]	1	1
Chromium $PM_{2.5}$	1	Using Eq. (2), $P = 1$, $k = 0.39 \pm 0.16\,h^{-1}$, and geometric means (GM) and geometric standard deviations (GSD) for a are GM $= 0.55\,h^{-1}$, GSD $= 1.97$ in winter and GM $= 1.05\,h^{-1}$, GSD $= 2.39$ in summer[c]	0.72 ± 0.053[d]
Diesel $PM_{2.5}$	Tri(2,3,6)[e]	Same as for chromium $PM_{2.5}$	0.63 ± 0.11[d]
Ozone	Tri(0.15,0.20,0.60)[f]	Randomly select a value from an empirical dataset, based on the season (summer/winter)[g]	Tri(0.3,0.5,0.8)[h]

[a]This table lists three of the four microenvironments employed in this work. The factor for the fourth microenvironment, outdoors, is 1.0 for all species.

[b]Based on Flachsbart (1995, 1999), Rodes et al. (1998), and Marshall et al. (2003). "Tri(2,4,6)" indicates a triangular probability distribution; the lowest value of the distribution is 2; the most common value is 4; the maximum value is 6.

[c]Penetration efficiency, P, is based on Ozkaynak et al. (1996) and Riley et al. (2002). Deposition rates, k, were determined by Ozkaynak et al. (1996) from co-temporal measurements indoors and nearby outdoors, assuming P equals unity. Note that empirical (Liu and Nazaroff, 2003; Long et al., 2001) and modeling (Liu and Nazaroff, 2001) studies of $PM_{2.5}$ suggest P values less than unity, typically between 0.6 and 1.0. However, to the extent that P is less than unity, this fact is accounted for in the experimentally determined k values (Kopperud et al., 2004). Air-exchange rates, a, are based on Wilson et al. (1996). For each residence, values for a and for k were randomly selected from lognormal and normal distributions, respectively, and then the residence microenvironmental adjustment factor for $DPM_{2.5}$ and Cr-$PM_{2.5}$ was calculated using Eq. (2), with a maximum value of 1.0.

[d]Based on results by Riley et al. (2002) for generic $PM_{2.5}$ (for Cr-$PM_{2.5}$) and for elemental carbon (for $DPM_{2.5}$). The distributions were treated as normal, with the indicated means and standard deviations, and with a maximum value of 1.0.

[e]Based on CARB (2004), Flachsbart (1995, 1999), Rodes et al. (1998), and Marshall et al. (2003).

[f]Based on Chan et al. (1991).

[g]Based on indoor and outdoor ozone measurements at 126 homes in the Los Angeles area (Avol et al., 1998). We divided this dataset of 235 indoor–outdoor ratios into two subsets: the 159 values taken during 15 April through 15 October ("summer"; range of values: 0.0–99.7%; mean = 34%), and the 76 measurements taken during 15 October through 15 April ("winter"; range: 0.0–71.0%; mean = 11.7%). When calculating the exposure concentration, an indoor–outdoor ozone ratio was chosen at random for each residence from the appropriate set of empirical indoor–outdoor ratios, based on the travel diary date.

[h]Based on Weschler (2000).

residential buildings, P is the penetration efficiency (dimensionless) of $PM_{2.5}$, a is the air-exchange rate (h^{-1}) for the building, and k is the particle removal rate (h^{-1}) for $PM_{2.5}$ by means of deposition or filtration indoors. Values for the input parameters in Eq. (2) (P, a, k) are provided in Table 1. Arithmetic means and standard deviations for the resulting values for $\gamma_{\text{residence, PM}_{2.5}}$ are 0.61 ± 0.06 in winter and 0.71 ± 0.07 in summer.

3. Results and discussion

3.1. Inhalation intake rates

Model results yield the inhalation intake rate ($\mu g\,d^{-1}$) of air pollution of ambient origin for each person-day in the dataset for each of the five species (benzene, butadiene, $DPM_{2.5}$, Cr-$PM_{2.5}$, and ozone). Table 2 provides statistics summarizing the inhalation rates and other model parameters, such as individuals' breathing rates and daily travel patterns. Mean intake rates ($\mu g\,d^{-1}$) for the five pollutants are 67 for benzene, 7.3 for butadiene, 47 for $DPM_{2.5}$, 0.0016 for Cr-$PM_{2.5}$, and 120 for ozone.

Fig. 2 presents cumulative distribution plots for the five species. Except at the high and low ends, all five distributions conform reasonably well to lognormal distributions, which would appear as straight lines in these plots. The deviation from the line at the low end of the ozone distribution reflects that some of the CAMx-estimated ambient ozone concentrations are unrealistically low.

Fig. 3 presents the average diurnal breathing rate profile for this investigation. To our knowledge, there are only two extant estimates for this diurnal profile (Mortola, 2004, 2006)—one presented in this

Table 2
Summary statistics for time–location-activity survey data and for estimated inhalation intake rates[a]

Variable	Percentile					Mean	Std	GM	GSD
	10th	25th	50th	75th	90th				
Number of household vehicles	1	1	2	2	3	2.0	1.1	1.9	1.6
Survey respondent's age (y)	7	19	37	54	70	38	23	28	2.5
Breathing rate ($m^3\,d^{-1}$)	9.5	10.5	12.1	15.0	16.3	13.1	4.4	12.6	1.3
Total straight-line distance traveled[b] ($km\,d^{-1}$)	0	0	13	38	79	29	46	23	3.1
Maximum distance from home (km)	0	0	5	15	33	12	18	9.3	3.1
Inhalation intake rate									
Benzene ($\mu g\,d^{-1}$)	19	30	53	89	130	67	54	51	2.1
Butadiene ($\mu g\,d^{-1}$)	0.80	1.9	5.1	10	16	7.3	7.6	4.1	3.4
Chromium $PM_{2.5}$ ($ng\,d^{-1}$)	0.18	0.38	0.87	1.9	3.7	1.6	2.6	0.83	3.4
Diesel $PM_{2.5}$ ($\mu g\,d^{-1}$)	8.8	15	30	63	110	47	51	30	2.6
Ozone ($\mu g\,d^{-1}$)	11	30	68	160	280	120	170	52	6.2[c]

[a]Values estimate inhalation intake of pollutants of ambient origin. Indoor emissions are excluded. Abbreviations used in this table are Std for standard deviation, GM for geometric mean, and GSD for geometric standard deviation.

[b]This is the total distance traveled on each person-day, assuming that each trip occurs in a straight line between the origin and the destination.

[c]The high GSD values for ozone are caused by some of the CAMx-estimated ambient ozone concentrations being unrealistically low. Excluding the bottom 10% of the ozone inhalation intake rate distribution, values ($\mu g\,d^{-1}$) for the mean, Std, GM, and GSD are 133, 176, 83, and 2.7, respectively.

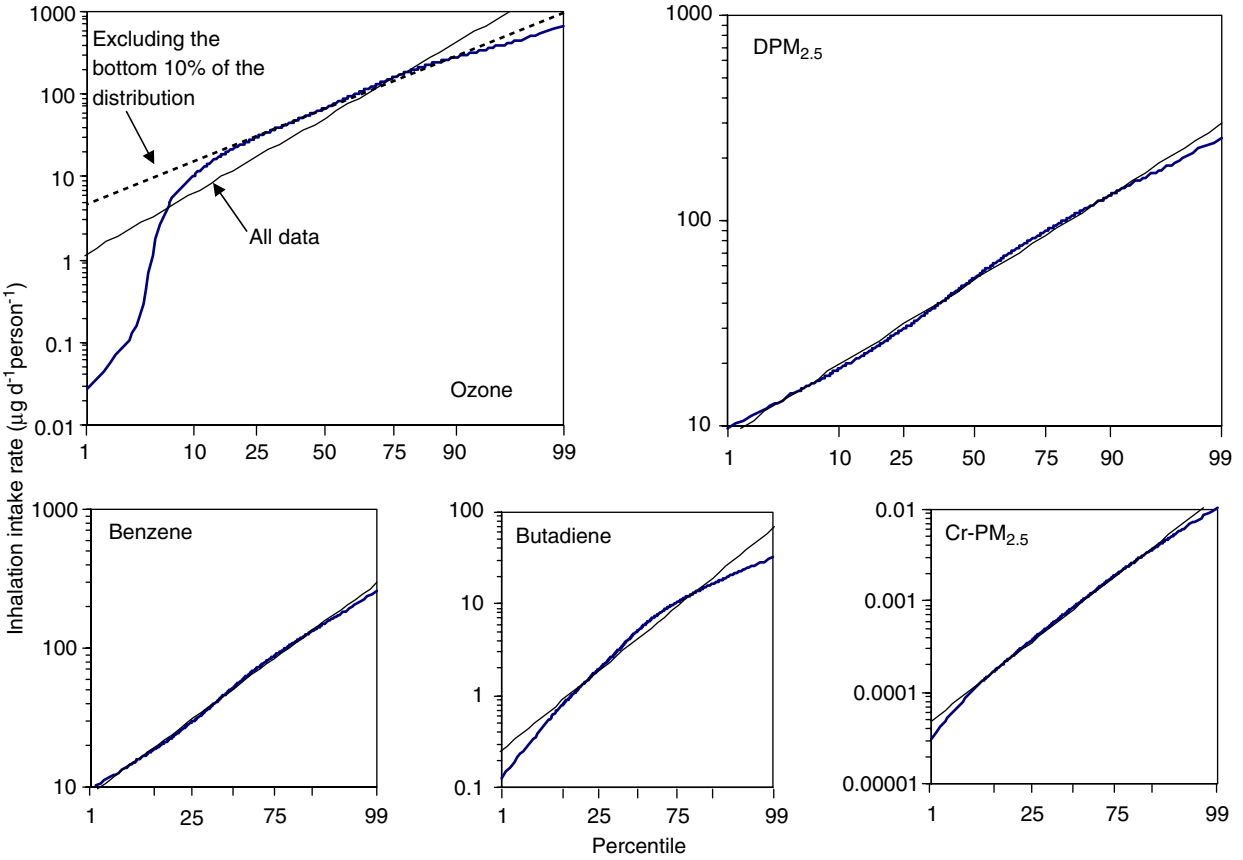

Fig. 2. Cumulative distribution plots of inhalation intake rate for the 28,746 person-days simulated, for each of the five chemicals studied. The *x*-axes tick marks correspond to the same percentile on all five plots; *y*-axes are logarithmic. The straight line in each plot represents a lognormal distribution. For ozone, an alternative line is also shown, representing the lognormal distribution that excludes the bottom 10% of the intake results.

work and a separate estimate in Marshall et al. (2003). Both estimates are approximate rather than definitive.

For ozone, the intake-relevant exposure concentration (defined in Table 3) is substantially lower than the basin-wide average concentration, mainly because of ozone decomposition in indoor microenvironments. For the other four pollutants, the intake-relevant exposure concentration is significantly higher than the basinwide average ambient concentration, for reasons that are explained in the following subsection.

Intake rates estimated here are for the ~29,000 person-days in the travel survey employed. These results are not necessarily representative of daily intake rates by the ~16 million people in the South Coast. For example, relative to US census data for the Los Angeles Metropolitan Statistical Area (LA-MSA), survey person-days include a higher proportion of whites (51% for the survey versus 22% for LA-MSA); a lower proportion of Hispanics (27% versus 43%), Asian/Pacific Islanders (6% versus 11%), and people who listed their ethnicity as "other" or "don't know/refused" (10% versus 17%); and the same proportion of African-Americans (7%). Survey person-days consist of only 13% weekend days, rather than 29% (i.e., two-sevenths). The survey sample slightly undersamples 5+ person households, zero-vehicle households, and low-income households (SCAG, 2003). Travel diaries likely underestimate mobility because of trip underreporting, which is estimated at ~35% of trips for the survey used here (SCAG, 2004), and because comparatively mobile individuals may be underrepresented in survey data since they are harder to contact (Schafer, 2000). Further discussion about the representativeness and reliability of the travel survey is available elsewhere (SCAG, 2003, 2004).

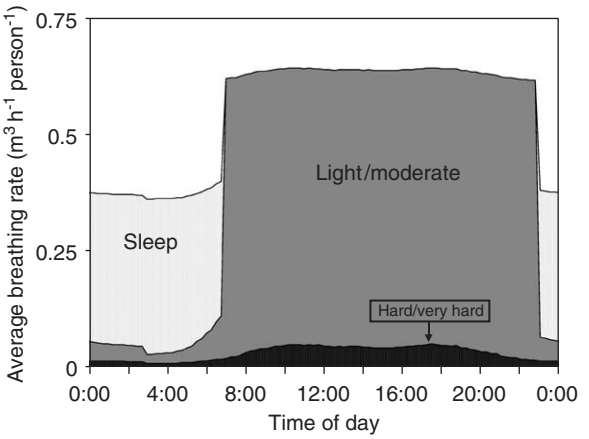

Fig. 3. Estimated diurnal profile of the population mean breathing rate for the 28,746 person-days simulated, by activity level. The daily average breathing rate is 13.1 $m^3 d^{-1}$ $person^{-1}$.

3.2. Evaluating factors that influence exposure variability

In assessing exposure to urban air pollution, it is common to use ambient concentration as a surrogate for exposure concentration. Three key factors that may not be considered in such cases are (1) population mobility, (2) breathing rate variability that exhibits temporal correlation with air pollution concentrations, and (3) microenvironments. The approach developed for this study permits

Table 3
Average ambient and intake-relevant exposure concentrations

Species	Mean ambient concentration[a] ($\mu g\,m^{-3}$)	Mean intake-relevant exposure concentration[b] ($\mu g\,m^{-3}$)	Ratio of mean intake-relevant exposure concentration to mean ambient concentration
Benzene	2.7 (0.82 ppb)	5.1 (1.6 ppb)	1.9
Butadiene	0.19 (85 ppt)	0.55 (240 ppt)	2.9
Chromium $PM_{2.5}$	96×10^{-6}	120×10^{-6}	1.3
Diesel $PM_{2.5}$	2.5	3.6	1.5
Ozone	50 (25 ppb)	9.2 (4.6 ppb)	0.18

[a]Annual-average on-land ground-level ambient concentration, based on CAMx model output for 1 April 1998 through 31 March 1999. The South Coast Air Basin modeling domain is 25,200 km^2 (120 km × 210 km), and contains 6300 grid cells of size 2 km × 2 km. The on-land portion of the air basin incorporates 4408 grid cells.

[b]Intake-relevant concentration is the concentration that, when multiplied by the mean breathing rate (here, 13.1 $m^3 d^{-1}$), yields the mean intake rate (given in Table 2).

quantification of the effects on estimated intake rate of these three factors for the population considered. To quantify the effects, we first computed intake rates using the basic approach described above, but with five distinct sets of assumptions: (1) neglecting the three key factors (''base case''); (2) accounting for people's movement throughout the air basin but not for microenvironments or for diurnal or between-individual variability in breathing rates (''mobile''); (3) accounting for microenvironments but not mobility or breathing rate variability (''microenvironments''); (4) accounting for diurnal and between-individual variability in breathing rates, based on activity level, gender, and age, but not mobility or microenvironments (''breathing rate variability''); and (5) accounting for mobility, breathing rate variability, and microenvironments (''all three factors''). (The default approach for results presented in this paper is ''all three factors''.) We then calculated the relative change in individuals' intake rates attributable to each set of assumptions, compared to the base case. Table 4 presents the average value among individuals for these relative changes in intake rates. We also calculated the average magnitude (i.e., average absolute value) among individuals (not shown). For both metrics (average value and average magnitude), mobility is the most important of the three factors considered for butadiene and chromium $PM_{2.5}$, and the least important factor for the remaining three pollutants. Averaged among the five pollutants, breathing rate variability influences intake rates by 14% for average value (Table 4) and by 26% for average magnitude (not shown); all three factors combined influence intake rates by 40% for average value and by 64% for average magnitude. For ozone, microenvironments are significantly more important than the other two factors (mobility; breathing rate variability); for the other four pollutants, this is not true. Thus, two important findings that emerge from our results are that (1) ignoring the three factors yields factor-of-2 or smaller errors, on average, in individuals' estimated daily intake rates, and (2) all three factors may usefully improve estimates of inhalation intake rates.

3.3. Inhalation intake rate by ethnicity and income

Equitable environmental policy seeks not only to reduce the population-average health risk attributable to air pollution, but also to ensure that specific subpopulations are not unduly burdened, relative to the population as a whole. Such concerns are components of the broader theme of environmental justice (Holifield, 2001; Levy et al., 2006; Taylor, 2000).

To investigate exposure variations among key subpopulations, we examined how estimated intake rates vary with demographic attributes such as ethnicity and income. Fig. 4 presents the median intake rate as a function of subpopulation for two household income levels and for four ethnic groups. (The transportation survey coded respondents' household income category as ''Above $50,000,'' ''Below $50,000,'' and ''Don't know/Refused''.) Fig. 4 excludes the 14% of respondents who did not provide their ethnicity or household income and the 3% of respondents who listed ''other'' as their ethnicity. Two types of differences are immediately apparent in this figure: among demographic groups (income category and ethnicity) and among pollutants (mainly, ozone versus the four primary pollutants). For the primary pollutants, median

Table 4
Mean change in individuals' estimated inhalation intake rate attributable to incorporating one or all of three factors relative to the base case[a]

Species	Microenvironment	Mobility	Breathing rate variability	All three factors
Benzene (%)	16	5	12	41
Butadiene (%)	18	30	14	93
Chromium $PM_{2.5}$ (%)	−31	27	11	8
Diesel $PM_{2.5}$ (%)	−19	8	13	4
Ozone (%)	−67	2	19	−55

[a]''Microenvironments'' accounts only for exposure concentrations indoors or in-vehicle, and not for mobility or breathing rate variability. ''Mobility'' accounts only for individuals' time-varying location (latitude and longitude), and not for microenvironments or breathing rate variability. ''Breathing rate variability'' accounts only for diurnal and between-person variability in breathing rates, and not for microenvironments or mobility. ''All three factors'' is the best estimate, accounting for microenvironments, mobility, and breathing rate variability. ''Base case'' accounts for none of the three factors.

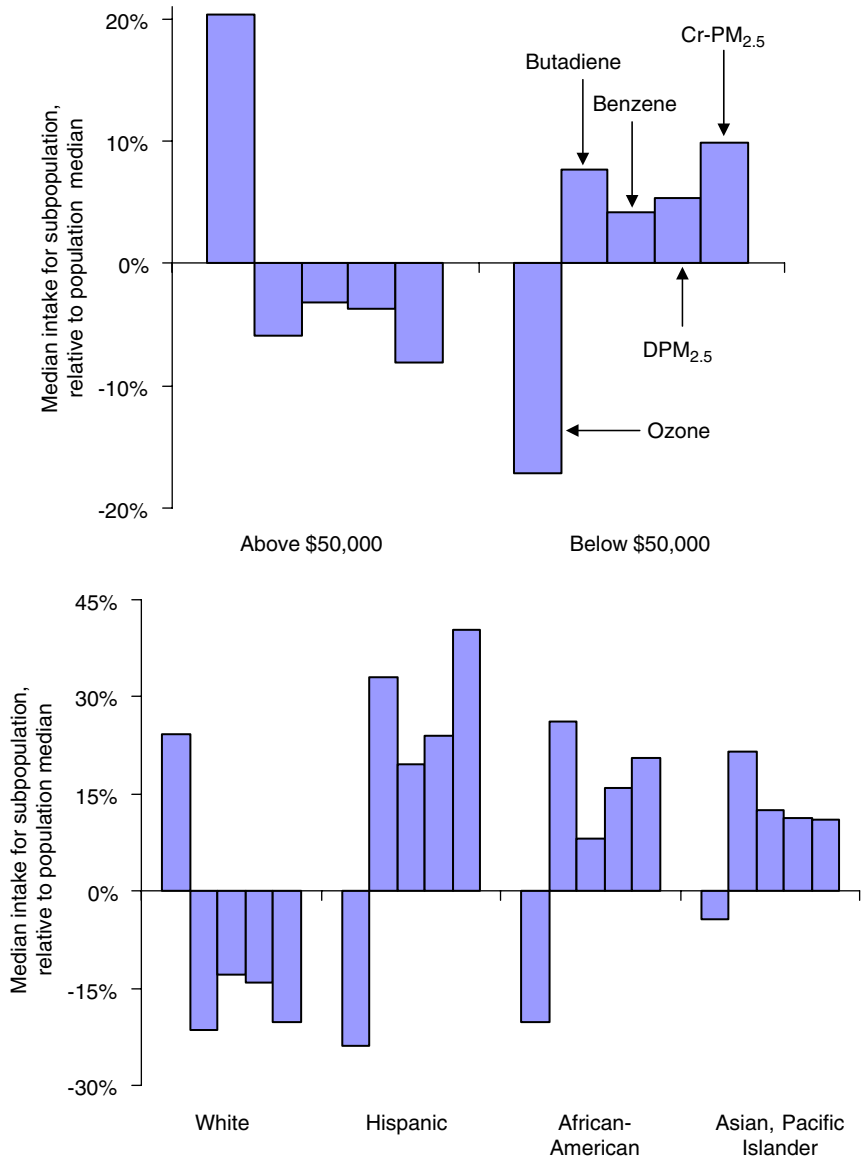

Fig. 4. Estimated median inhalation intake rate for the subpopulation, relative to the population median, based on household income category (upper plot) and ethnicity (lower plot). Values for the five pollutants are listed in the same order for each subpopulation (from left to right: ozone, butadiene, benzene, diesel $PM_{2.5}$, chromium $PM_{2.5}$).

intake rates are lower for whites, and higher for Hispanics, African-Americans, and Asians/Pacific Islanders, than for the population as a whole. Ozone intake rates exhibit the opposite trend. For the four primary pollutants, individuals in higher income households have lower intake rates than individuals in lower income households. For ozone, the reverse is true. On average, for the case study considered in this work, exposure levels differ more among ethnic groups than between high- and low-income households. While Fig. 4 only compares median intake values, the same general trends hold throughout the distribution.

In all likelihood, the main factor underlying these trends is proximity to emission sources. Typically, in California's South Coast Air Basin, non-whites and low-income households are in closer proximity to emission sources than the average person (Gunier et al., 2003; Houston et al., 2004; Pastor et al., 2004). For primary pollutants, proximity increases

exposure concentrations. For ozone, because advection moves air masses during the time required for precursor emissions to form ozone, high ozone concentrations are *not* proximate to emission sources but rather occur in downwind locations.

Although not accounted for in our model, another factor that would also cause correlations between income level and ambient air pollution intake rates is building and vehicle air-exchange rates. Older, ''leakier'' residences, which are more likely to be occupied by low-income than high-income families, offer less protection against outdoor particles and ozone than do newer and well-maintained buildings.

Results presented here are largely but not entirely consistent with related information presented elsewhere. Several studies have reported higher exposures for low-income groups and non-whites for primary pollutants (Brown, 1995; Schweitzer and Valenzuela, 2004), including populations in California in general (Gunier et al., 2003; Pastor et al., 2004) and specifically in Southern California (Morello-Frosch et al., 2001, 2002). Consistent with the findings presented here, the proportion of upper income households and whites are higher in high-ozone areas downwind of New York City and Philadelphia than in the urban core where ozone precursors are emitted (Liu, 1996). Others have found that ozone levels in the South Coast are positively correlated with the percentage of whites in the community, but are inversely correlated with income (Brajer and Hall, 2005; Korc, 1996).

All else being equal, the ambient concentration difference between near-source and not-near-source would be larger for rapidly decaying emissions (e.g., primary ultrafine particles) than for slowly decaying pollutants (e.g., benzene). We expect, therefore, that the difference in intake rates between whites and non-whites and between high- and low-income households is greater for rapidly decaying emissions than for slowly decaying emissions. Our results for butadiene and benzene support this idea. Butadiene decays more rapidly than benzene: characteristic lifetimes are ~6 and ~500 h, respectively (US EPA, 1993), and the fraction of emissions that decay before air leaves the air basin by advection is small for benzene but not butadiene (~1–3% versus ~50–70%) (Marshall, 2005). In the present study, the difference in intake rates among subpopulations is greater for butadiene than for benzene. (See Fig. 4.) Gini and Atkinson coefficients (see below) corroborate this finding.

Median intake rates are ~44% greater for men than women. Most (97%) of this difference is attributable to differences in median breathing rates (men: 14.9 $m^3 d^{-1}$; women: 10.5 $m^3 d^{-1}$); the remainder (3%) is attributable to minor differences in exposure concentrations. Median exposure concentrations tend to decline with age, with an exception being that median ozone exposures are lower during ages 0–5 yr than during ages 6–65 yr.

We calculated two inequality metrics for inhalation intake rates: (1) the Gini coefficient, because it is commonly used in equity literature, and (2) the Atkinson coefficient, because it may be preferred in exposure assessments for theoretical reasons (Levy et al., 2006). In general, these coefficients range in value from zero (complete equality: values are identical for all individuals) to one (complete inequality: values are nonzero for a single individual and zero for all other individuals). For the five pollutants, Gini coefficients average 0.50 (range: 0.40 for benzene to 0.57 for Cr-$PM_{2.5}$), and Atkinson coefficients (employing $\varepsilon = 0.75$) average 0.32 (range: 0.18 for benzene to 0.41 for ozone). Considering Gini coefficients and Atkinson coefficients (ε range: 0.25–2.0), benzene intakes are always the least inequitable, followed by $DPM_{2.5}$ and then butadiene; either ozone or Cr-$PM_{2.5}$ are always the most inequitable. As a comparison, these metrics for US household income are 0.46 (Gini) and 0.21 (Atkinson; $\varepsilon = 0.75$) (US Census, 2000, 2005), indicating that inhalation intake rates are generally more inequitable than US household income.

4. Conclusions

We have estimated inhalation intake rates for five pollutants (benzene, 1,3-butadiene, fine particulate matter emitted by diesel engines, hexavalent chromium in the form of fine particulate matter, and ozone) for ~25,000 individuals in California's South Coast Air Basin. The combined mean effect on estimated intake rate of mobility, temporally varying breathing rates, and microenvironments is between 4% and 93% for the five pollutants considered here. For the four primary pollutants studied, median intake rates are higher for non-whites than for whites, and higher for individuals in households with less than $50,000 income per year than for those in households with more than $50,000 income per year. For ozone, the reverse is true. The approach developed here could usefully be

applied to other urban areas. One could build on our approach to estimate the impact on intakes and intake distributions of potential changes to air quality and its determinants (e.g., fuel reformulation or proposed transportation projects). Compared with current approaches, which emphasize reducing mass emissions or meeting ambient concentration standards, such activities offer the potential of improving the cost efficiency of air pollution control programs for protecting public health.

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