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Development of an individual exposure model for application to the Southern California children's health study

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Abstract

The Southern California Children's Health Study (CHS) investigated the relationship between air pollution and children's chronic respiratory health outcomes. Ambient air pollutant measurements from a single CHS monitoring station in each community were used as surrogates for personal exposures of all children in that community. To improve exposure estimates for the CHS children, we developed an Individual Exposure Model (IEM) to retrospectively estimate the long-term average exposure of the individual CHS children to CO, NO_2, PM_{10}, $PM_{2.5}$, and elemental carbon (EC) of ambient origin. In the IEM, pollutant concentrations due to both local mobile source emissions (LMSE) and meteorologically transported pollutants were taken into account by combining a line source model (CALINE4) with a regional air quality model (SMOG). To avoid double counting, local mobile sources were removed from SMOG and added back by CALINE4. Limited information from the CHS survey was used to group each child into a specific time-activity category, for which corresponding Consolidated Human Activity Database (CHAD) time-activity profiles were sampled. We found local traffic significantly increased within-community variability of exposure to vehicle-related pollutants. PM-associated exposures were influenced more by meteorologically transported pollutants and local non-mobile source emissions than by LMSE. The overall within-community variability of personal exposures was highest for NO_2 ($\pm$20–40%), followed by EC ($\pm$17–27%), PM_{10} ($\pm$15–25%), $PM_{2.5}$ ($\pm$15–20%), and CO ($\pm$9–14%). Between-community exposure differences were affected by community location, traffic density, and locations of residences and schools in each community. Proper siting of air monitoring stations relative to emission sources is important to capture community mean exposures.

Keywords: Exposure model; Vehicle-related pollutants; Meteorologically transported pollutants; Local pollutant emissions; Time-activity patterns

1. Introduction

Children represent a population particularly vulnerable to air pollution, since children spend more time outdoors, are generally more active, and have immature

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lungs and higher ratios of ventilation rate to body weight than adults. The Southern California Children's Health Study (CHS) investigated the relationship between air pollution and chronic respiratory health outcomes for over 5000 children in twelve communities in southern and central California (Gauderman et al., 2000; McConnell et al., 1999, 2002; Peters et al., 1999). This 10-year longitudinal study found deficits in growth of children's lung function associated with higher ambient concentrations of particles (PM_{10}, $PM_{2.5}$, and $PM_{10-2.5}$), NO_2, and inorganic acid vapor.

In the original CHS data analyses, ambient air pollutant measurements from a single CHS monitoring station in each community were used as surrogates for personal exposures of all the children in that community. However, central monitoring station measurements may not be representative of personal exposures (Alm et al., 1998; Janssen et al., 1998; Wallace, 1993) because both children and adults spend the great majority of their time indoors or in vehicles (Klepeis et al., 2001; Leech et al., 2002), and higher exposures can result from indoor sources, the "personal cloud," and vehicle emissions (Brauer et al., 2000; Fruin, 2003; Long et al., 2000; Rodes et al., 1998; Wallace, 2000).

Moreover, the pollutant concentrations in urban areas may vary by more than an order of magnitude on spatial scales varying from tens to hundreds of meters. For example, Zhu et al. (2002a, b) found that highly elevated concentrations of CO, black carbon and ultrafine particles at the edges of freeways dropped to background levels within approximately 200–300 m downwind from the freeways. Hitchins et al. (2000) found the concentrations of submicron particles dropped by about 50% 150 meters away from a road. These studies suggest that, especially for vehicle-emitted pollutants, fixed-site monitoring stations may not adequately represent areas beyond their immediate vicinity, and a high spatial resolution down to tens of meters may be required to better characterize exposure to such pollutants.

In general there are two origins of outdoor pollution at any given location: local source emissions (e.g. traffic) and pollutants (both primary and secondary) transported from upwind regions. As noted above, local traffic may contribute significantly to the highly heterogeneous spatial distributions of outdoor pollutants; however, pollutants transported from upwind areas cannot be neglected, especially particulate matter. For example, Briggs and Gulliver (2002) reported that road traffic typically accounted for only about 50% of total particle emissions in the UK and secondary and meteorologically transported particles may make up two-thirds or more of monitored concentrations. Similar results were found in the California South Coast Air Basin (SoCAB), where the $PM_{2.5}$ concentrations at downwind sites were much higher than those at upwind sites due to the secondary particles formed from gaseous precursors transported from upwind sites (Hughes et al., 1999; Kleeman et al., 1999). Similarly, Lim and Turpin (2002) reported that nearly 50% of the particulate organic matter in Atlanta, Georgia was of secondary origin.

Although recent exposure studies have begun to use personal monitors to directly measure subject exposures to CO, NO_2, particulate matter and air toxics (Meng et al., 2004; Monn, 2001; Ozkaynak et al., 1996; Rodes et al., 2001), this approach is expensive, and necessarily limited in the number of subjects that can be studied. For example, personal monitoring was not used in the CHS (over 5000 children) for these reasons. However, appropriately formulated computer models with robust input data are an alternative approach to estimating personal exposures, by weighting pollutant concentrations in various microenvironments by the fraction of time children and adults spend in these locations. The overall objective of the present modeling study was to provide, retrospectively, more accurate and comprehensive assessments of the long-term average exposure of the individual CHS children to vehicle-related pollutants. The specific objectives of the present study were to quantify the variability of within-community exposures; to determine exposures due to local mobile source emissions (LMSE) relative to meteorologically transported pollutants and local non-mobile source emissions (LN-MSE); and to facilitate evaluation of relationships between exposure and health outcomes for the individual children in the CHS cohort.

2. Methods

2.1. Overview of modeling approach

An Individual Exposure Model (IEM) was developed to determine personal long-term average exposures to CO, NO_2, PM_{10}, $PM_{2.5}$, and elemental carbon (EC; $PM_{2.5}$ portion) for the CHS children cohort. Five microenvironments (residential outdoor, residential indoor, school outdoor, school indoor, and in vehicle) where children spend most of their time were investigated. The mathematical basis for the IEM is

$$\mathrm{TWE}_i = \frac{E_i}{T_i} = \frac{\sum_{j=1}^{J} C_i \Delta t_{ij}}{\sum_{j=1}^{J} \Delta t_{ij}}, \quad (1)$$

where E_i is the cumulative exposure of individual subject i; C_j is the pollutant concentration present in microenvironment j for the time period Δt_{ij}; Δt_{ij} is the time spent by individual subject i in microenvironment j; J is the total number of microenvironments; T_i is the

exposure averaging period; and TWE_i is the time-weighted average exposure for a specific individual.

2.2. Outdoor pollutant concentrations

2.2.1. Outdoor pollutant concentrations due to local traffic emissions

We used the CALINE4 model, developed by the California Department of Transportation (Caltrans) and the US Federal Highways' Agency (Benson, 1989, 1992), to estimate local-scale pollutant concentrations caused directly by motor vehicle emissions. The CALINE4 model is a Gaussian line-source dispersion model designed for the assessment of traffic emissions from roads.

Annual average daily traffic counts data for freeways, arterials, and collectors were obtained from Caltrans. Average hourly traffic volumes were estimated using diurnal traffic profiles and weekday/weekend volumes differences determined from weigh-in-motion sensors on freeways and traffic surveys on secondary roadways and surface streets. Diesel and gasoline vehicles were differentiated since diesel vehicles emit greater amounts of particles and elemental carbon than gasoline vehicles. In addition, significant differences exist for diesel and gasoline vehicle ratios among different freeways in the SoCAB. For example, the 710 freeway has about 7 times as many diesel vehicles as the 405 freeway on average (Zhu et al., 2002a, b; California Department of Transportation (Caltrans), 2000). Vehicle emission factors were obtained from the most recent California vehicle emissions model, EMFAC 2002 (California Air Resources Board, 2002). Hourly meteorological data (temperature, wind speed and wind direction) were obtained from air quality monitoring stations operated by the South Coast Air Quality Management District (1997).

ArcInfo (ESRI Inc.) was used to preprocess the roadway segment and traffic count data. Caltrans roadways were found to have as much as 250 m discrepancies from GPS-accurate TeleAtlas Roadway Network data (Wu et al., 2004). Therefore, an algorithm was developed within a GIS to transfer Caltrans' traffic activity data to the TeleAtlas Roadway Network (Wu et al., 2004). Children's homes, schools and air quality monitoring stations were also geocoded using the TeleAtlas Geocoding Service at an accuracy of ± 20 m for proper addresses in a dense area.

2.2.2. Outdoor pollutant concentrations due to meteorologically transported pollutants and local non-mobile source emissions

Our study area extended 360 km from west to east and 180 km from north to south; thus differences in "background" or transported pollutant concentrations, as well as "local" emissions, needed to be investigated for such a large region. "Background" concentrations and local non-mobile sources were taken into account by applying the Surface Meteorology and Ozone Generation (SMOG) airshed model developed by Lu et al. (1997a, b) and updated to include the MM5 prognostic meteorology model. The SMOG model is a regional air quality modeling system that has been applied in simulations of surface air pollutant concentrations and elevated pollution layers observed over the SoCAB (Lu et al., 2003).

In the present study, the SMOG model was applied using nested grids, with a grid resolution of 5 km × 5 km in the SoCAB. At least 90% of the children's residences and all of the schools in each CHS community were encompassed by a modeled area approximately 15 km × 15 km overlayed on that community (Fig. 1). Seven of the twelve CHS communities were covered by the fine resolution SMOG model domain and were the focus of this paper, including Lancaster (LAN), San Dimas (SDM), Upland (UPL), Mira Loma (MRL), Riverside (RIV), Long Beach (LGB), and Lake Elsinore (LKE).

Since we estimated local traffic emissions using the CALINE4 model, in the SMOG simulations we removed local mobile sources from each CHS community, to avoid double counting of vehicle emissions. In this study, local mobile sources included not only hot running vehicle exhaust emissions but also brake wear, tire wear, paved road dust, cold start exhaust, and hot start exhaust.

Since it was impossible to run the SMOG model for an entire year because of computational demands, two representative 3-day episodes were chosen for the warm and cold seasons, respectively. Intra-seasonal meteorological patterns in Southern California are not highly varied, making it reasonable to obtain regional pollutant distributions for an entire season using simulations of a well-chosen multiple-day episode. Moreover, given the uncertainty and variability in the models, we used the SMOG model outputs on a relative, rather than absolute basis (as described below), which reduced propagation of biases and uncertainties in the SMOG model.

2.2.3. Location-specific ambient pollutant concentrations

We calculated the ambient pollutant concentrations for each CHS residence and school by combining the central site ambient observations with results from the CALINE4 line source dispersion model and the regional SMOG airshed model using the following formula:

$$C^{L}_{out} = C^{L}_{LMSE} + \alpha C_{obs}, \quad (2)$$

$$\alpha = \frac{C_{PE}}{C_{AE}}, \quad (3)$$

where C^{L}_{out} is the calculated total outdoor pollutant concentration at each residence and school; C^{L}_{LMSE} is the

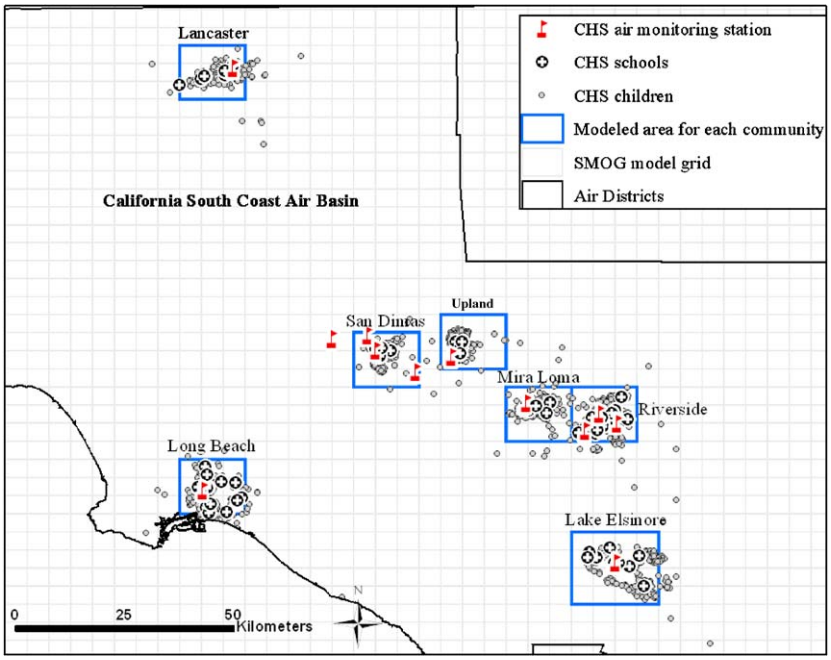

Fig. 1. The seven CHS communities within the SMOG fine resolution domain.

outdoor pollutant concentration at each residence and school due to LMSE, calculated by the CALINE4 model; C_{obs} is the pollutant concentration measured at each CHS central monitoring station; C_{AE} is the pollutant concentration at the center of each community calculated by running the SMOG model with all emissions (AE); C_{PE} is the pollutant concentration calculated by running the SMOG model with partial emissions (PE), i.e. without local mobile emissions; and α is the fraction of ambient air pollution in a community due to meteorologically transported pollutants and LN-MSE.

The basis for this approach is that vehicle-related primary pollutant concentrations can be separated into, and modeled as LMSE, and transported and LN-MSE. In order to reduce uncertainties in the models, we used both CHS ambient air monitoring station data and SMOG-estimated fractions of pollutant concentrations from transported and local non-mobile sources to obtain "background" pollutant concentrations. Pollutant concentrations due to local mobile emissions were calculated by the CALINE4 model. Detailed information on the CHS monitoring network instrumentation and data collection and validation can be find elsewhere (Lurmann et al., 1994; Peters et al., 1999, 2004).

2.3. Indoor pollutant concentrations

Recently, researchers (Burke et al., 2001; Wilson et al., 2000) have addressed the importance of separating contributions of outdoor and indoor air for personal exposures since the EPA currently controls or regulates only pollutants originating outdoors. In addition, indoor and outdoor sources have different origins and may exhibit different roles in health outcomes. For example, Ebelt et al. (2003) found stronger relationships between particulate matter pollution and adverse health effects for ambient-origin particles than non-ambient particles. To calculate indoor concentrations due to penetration of outdoor air, we used a single-compartment, steady-state mass balance equation, in which indoor and outdoor sources can be separated (Burke et al., 2001; Koontz et al., 1998; Koutrakis et al., 1992; Ozkaynak et al., 1996)

$$C_{in} = \frac{paC_{out}}{a+k} + \frac{Q_{is}}{(a+k)V}, \quad (4)$$

where p is the penetration coefficient; a the air exchange rate (AER) (h^{-1}); k the decay rate (h^{-1}); Q_{is} the mass flux generated by indoor sources ($\mu g\,h^{-1}$), $Q_{is}=0$ in this application; and V is the house volume (m^3)

We did not include indoor sources (the second term in Eq. (4)) for this specific application. Instead, the CHS epidemiologists separately examined factors associated with indoor sources as categorical variables in their statistical models (McConnell R., Personal communication), which removed potential uncertainties that could be introduced by including indoor sources in the exposure modeling. For example, although limited data on environmental tobacco smoke and pilot light usage were available from the CHS survey, on an individual

basis we did not have sufficient reliable information concerning house volume, indoor source emission rates, or activity data to calculate the specific exposures of children to elevated indoor pollutant concentrations. Since no indoor sources were included in the present study, the personal exposures obtained from the modeling are exposures due to pollutants of ambient origin.

The penetration coefficient p and the deposition rate k vary by particle size, season, and air exchange rate. For all gaseous pollutants, we assumed a unit penetration coefficient. In the case of particles, several studies indicate that p is close to one (Clayton et al., 1993; Liu and Nazaroff, 2001; Thatcher and Layton, 1995) for a wide range of particle diameters, while several other studies have presented data indicating the penetration factor may be significantly less than one (Koutrakis et al., 1992; Thornburg et al., 2001). The penetration coefficients for elemental carbon, $PM_{2.5}$, and PM_{10}, were all assumed to have a triangular distribution with minimum, mode and maximum values of 0.9, 0.95 and 1.0, respectively (Koontz et al., 1998). This is reasonable since homes in Southern California typically have higher air exchange rates than the national average (Murray and Burmaster, 1995), and penetration factors increase to around one at approximately two or more air exchanges per hour (Long et al., 2001).

We assumed an indoor decay rate of $1.0\,h^{-1}$ for NO_2 (Yamanaka, 1984), which was within the range of $0.2–1.3\,h^{-1}$ found by Nazaroff et al. (1993). A zero decay rate was assigned to CO because of its inert properties. We adopted normal distributions for the deposition rates of PM_{10} and $PM_{2.5}$ with means and standard deviations of $0.65\,h^{-1}$ and 0.28, and $0.39\,h^{-1}$and 0.16, respectively (Ozkaynak et al., 1994). At present, little information is available on the deposition rate of elemental carbon, for which we assigned the deposition rate of $PM_{2.5}$.

The air exchange rate depends on building construction, ambient conditions, and resident activities (Wallace, 1996). Distributions of air exchange rates have generally been fitted by log-normal parameters (Murray and Burmaster, 1995). Colome et al. (1994) found in the California Residential Indoor Air Quality Study that air exchange rates were closely associated with home volume, cooking type, and heating type (types of cooking and heating facilities were co-variables for building age); therefore, a mean air exchange rate was assigned to each CHS home according to the house type (single-detached and attached) and cooking facilities (electric stove, gas stove without pilot light, gas stove with pilot light) (Table 1).

Table 1
Average air exchange rates (h^{-1}) used in the IEM (Colome et al., 1994)

Building type	Cooking facility		
	Electric	Gas without pilot	Gas with pilot
Single family detached	0.4	0.5	0.7
Multi family attached	0.6	0.9	1.2

2.4. In-vehicle pollutant concentrations

Children may be exposed to significantly higher pollutant concentrations during their travel between home and school, and during other times spent in vehicles. Mainly, we used the California-specific measurement data (Fruin, 2003; Fruin et al., 2004; Rodes et al., 1998) to obtain the in-vehicle pollutant concentrations appropriate for California vehicle mix and roadways. The 7 CHS communities were grouped by traffic densities; for the communities with high traffic densities we used in-vehicle measurements from urban Los Angeles; otherwise, we used data from Sacramento. The values of in-vehicle concentrations we used in the IEM are listed in Table 2.

2.5. Time-activity patterns

2.5.1. Time-activity sequence generation

In the CHS, a time-activity survey was administered twice a year to each child, asking them how much time (by 5 categories) they spent outdoors in the afternoons (12 PM–6 PM), on the weekdays and the weekend days, and during the summer. The survey also asked the children if they spent more than 15 min daily traveling between school and home and by what means. The survey was more useful for classifying individual subjects by their time activities than for providing quantitative time activity data for use in exposure modeling.

The Consolidated Human Activity Database (CHAD) developed by the US EPA provides 24-h time-activity patterns based on recall diaries (US EPA, 1997). A time-activity submodel was developed to create 24-h time-activity series (15-min intervals) for each child in the CHS cohort by using information from both the CHS survey and the CHAD database. The CHS children and selected children in the CHAD database were separately grouped by age (9–12 and 13–18 years old), gender, and day type (non-summer weekdays, non-summer weekend days, and summer). The CHAD and CHS profiles for each age, gender, and day-type were then stratified into high and low time outdoors and time in vehicle subgroups. The median time outdoors and time in vehicles for the CHAD profiles are shown in Table 3.

For each of the 48 CHS categories a child was grouped into, the corresponding CHAD distribution was sampled. Only CHAD time-activity data collected

Table 2
In-vehicle pollutant concentrations used in the IEM

Pollutant	High-traffic community	Low-traffic community	References[a]
CO	5.2 ($mg\,m^{-3}$)	2.0 ($mg\,m^{-3}$)	1
NO_2	3 × ambient[b]	3 × ambient[b]	2,3
PM_{10} mass	60 ($\mu g\,m^{-3}$)	21 ($\mu g\,m^{-3}$)	1
$PM_{2.5}$ mass	49 ($\mu g\,m^{-3}$)	11 ($\mu g\,m^{-3}$)	1
$PM_{2.5}$ EC	7 ($\mu g\,m^{-3}$)	4 ($\mu g\,m^{-3}$)	4, 5

[a]References: 1 = Rodes et al. (1998); 2 = Fitz et al. (2003); 3 = Shikiya et al. (1989); 4 = Fruin (2003); 5 = Fruin et al. (2004).
[b]Central site ambient concentration.

Table 3
Median time outdoors and time in vehicles by age, gender, and day type for time-activity patterns extracted from CHAD

Age	Gender	Day type	Median time outdoors[a] (min)	Median time in-vehicles[a] (min)
7–12	Male	Non- summer week days	60	30
		Non-summer weekend days	90	45
		Summer	105	30
	Female	Non-summer week days	30	45
		Non-summer weekend days	45	60
		Summer	75	45
13–18	Male	Non-summer week days	15	45
		Non-summer weekend days	90	45
		Summer	60	45
	Female	Non-summer week days	<7[b]	60
		Non-summer weekend days	<7[b]	75
		Summer	15	30

[a]Times are truncated to nearest 15 min interval. The median was the cut-point for the low and high times in these locations.
[b]Based on 15 min interval time-activity data. For these two groups, all profiles with zero time outdoors were grouped into "low" category, and all profiles with greater than zero values were grouped into "high" category.

by the California Air Resources Board (Wiley, 1991; Wiley et al., 1991) and the National Human Activity Pattern Survey (Klepeis et al., 2001) were used, to ensure the quality and consistency of the data. Categories with <20 profiles were combined to avoid biases caused by sampling from too small a data set. The large number of CHAD codes for locations and activities were aggregated into the five corresponding microenvironments (residential indoor, residential outdoor, school indoor, school outdoor and in-vehicle) we studied. All other locations were grouped into a single category where residential indoor concentrations were assigned.

2.5.2. Travel time

As noted, the CHS survey asked the children if they spent more than 15 min traveling home from school (no exact travel times were available from the survey). We do not have sufficient individual information to characterize the mobility of subjects other than between their residence and school. Because the CHS children attended public schools in their communities, their school/home trips are generally short. Therefore, the mobility may not be particularly important for this cohort but obviously could be important for other cohorts.

We used services provided by www.mapquest.com to estimate the children's travel time for school/home trips using home and school address data. An apparent discrepancy was found between the Mapquest estimates and the survey results since about 28% of children who reported they spent more than 15 min traveling from school to home by vehicles had travel times estimated from Mapquest of <5 min. The most likely reason for this mismatch was that children did not go home directly from school, but this was not accounted for in the time-activity survey. Consequently, we used the survey results by separating the children into two categories: <15 min and more than 15 min vehicle travel time from school to home. Fifteen minutes is an appropriate grouping variable here since the 1995 and 2001 National Household Travel Survey reported an average of 15 and 18 min for children's travel to schools, respectively (NHTS,

2003). It was reasonable to group these children into two broad travel categories since we sampled from the CHAD diaries rather than using the CHS time-activity data directly.

3. Results and discussion

3.1. Outdoor pollutant concentrations due to meteorologically transported pollutants and local non-mobile source emissions

Previous work has established the capabilities of the SMOG model through comparisons of model simulations against observations across the SoCAB (e.g., Jacobson et al., 1996; Lu and Turco, 1996; Lu et al., 1997a, b, 2003; Jacobson, 1997). The model simulations in this study were generally in good agreement with air quality measurements, consistent with detailed evaluations of the SMOG model performances found in previous publications (e.g., Lu et al., 1997b). For example, the gross errors were 38% and 34% and the biases were −15% and −3%, for ozone predictions for the summer and winter episodes, respectively.

Fig. 2 shows an example of the ratio of hourly CO concentrations from transport and LN-MSE to the total CO concentrations from all sources (Note that "transport" here does not mean "traffic" but "movement" of pollution from upwind regions). These ratios correlated well inversely with traffic density, being higher at off-traffic time and significantly lower during morning and afternoon peak traffic hours. As expected, local traffic contributed more to the total CO concentrations during peak traffic times. The other directly emitted pollutants followed diurnal patterns similar to CO.

Average hourly fractions of ambient air concentrations due to transported pollutants and LN-MSE (α in Eq. (2)) are summarized by community and pollutant type in Table 4. Most PM_{10} and $PM_{2.5}$ came from transport and LN-MSE, in agreement with recent studies showing that PM_{10} and $PM_{2.5}$ act more like regional pollutants rather than reflecting direct emissions from motor sources (Levy et al., 2003). Similarly, although more than 90% of CO in the SoCAB originates from mobile vehicle emissions, meteorologically transported CO (including mobile source emissions from upwind locations) and LN-MSE still comprised over 60% of the CO concentrations at all of the CHS communities (except Lancaster, which is located at the northern edge of the SoCAB and receives only a small amount of transported pollutants under the meteorological conditions occurring in these episodes).

3.2. Outdoor pollutant concentrations due to local traffic emissions

We used the CALINE4 model to estimate pollutant concentrations due to LMSE at each residence and school. The performance of CALINE4 model has been evaluated in a number of studies (Benson, 1989, 1992; Gramotnev et al., 2003; Loranger et al., 1995; Marmur and Mamane, 2003). With high-quality input data, the model can estimate pollutant concentrations within a factor of two of measurement data. In this study, no validation was conducted specifically for the CALINE4 model application in Southern California. Although we predicted background pollutant concentration from the SMOG model, we used the model outputs on a relative basis (as ratios) instead of absolute values because of the inconsistency in emission inventories (the SMOG model currently uses gridded EMFAC7G outputs for mobile source emissions, while EMFAC 2002 vehicle emissions factors are applied to individual line sources in the CALINE4 emission inventory). Future improvement is

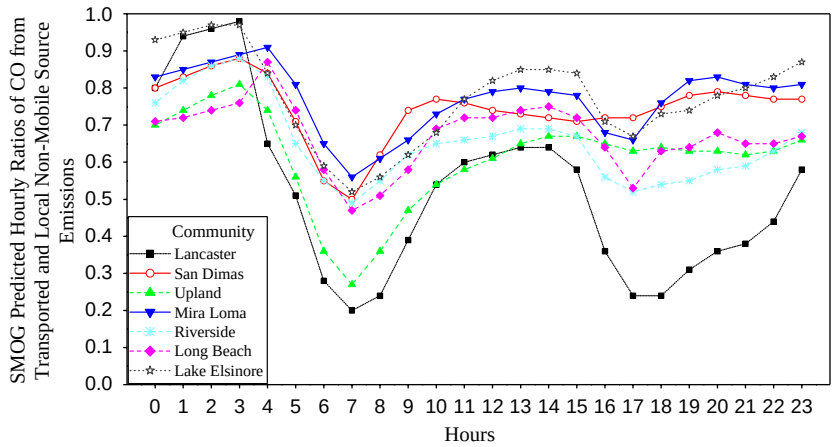

Fig. 2. Hourly ratios of CO concentrations from transport and LN-MSE to the total CO concentrations from all sources in the CHS communities predicted by the SMOG model for a winter season.

Table 4
Daily average ratios of pollutant concentrations from transport and LN-MSE to the total concentrations from all sources in each community

Communities	Season									
	Summer					Winter				
	CO	NO_2	EC	$PM_{2.5}$	PM_{10}	CO	NO_2	EC	$PM_{2.5}$	PM_{10}
Lancaster	0.79	0.33	0.86	0.94	0.90	0.39	0.31	0.55	0.87	0.88
San Dimas	0.80	0.78	0.91	0.97	0.92	0.74	0.85	0.88	0.94	0.90
Upland	0.79	0.69	0.84	0.94	0.87	0.61	0.74	0.75	0.89	0.83
Mira Loma	0.87	0.63	0.90	0.98	0.94	0.77	0.86	0.86	0.96	0.94
Riverside	0.78	0.53	0.84	0.96	0.91	0.64	0.72	0.71	0.88	0.84
Long Beach	0.61	0.88	0.84	0.90	0.76	0.65	0.85	0.82	0.89	0.82
Lake Elsinore	0.84	0.32	0.89	0.97	0.94	0.77	0.48	0.83	0.97	0.94

needed for consistency of emissions inputs used by urban-scale and microscale air quality dispersion models.

Fig. 3a shows CALINE4 estimates of hourly community mean CO concentrations due to local traffic in winter; these CO concentrations again corresponded well with morning and evening traffic peaks. Large variations in local traffic effects were observed among the CHS residences in Riverside, as illustrated in Fig. 3b. The hourly median CO concentrations at Riverside residences were about $0.5\,mg\,m^{-3}$, but a significant number of residences had concentrations above 1.0 and up to $2.8\,mg\,m^{-3}$, which suggested that certain residences in Riverside located close to freeways and major arterials were strongly affected by vehicular emissions. This local traffic effect was also observed in the other communities.

3.3. Within-community variability in personal exposures

Annual average daily exposures of children in 1997 were estimated using the procedures described above. We focused on children's long-term exposure because that was the primary target of the CHS study. No model validations were possible for the IEM since no personal monitoring data were collected during the CHS study. The lack of such data was one of the main reasons we conducted this retrospective exposure modeling, incorporating a number of factors not captured by a single air monitoring station but which strongly influence personal exposures, such as children's time-activity patterns, traffic densities on freeways and major roadways, and children's residential and school locations.

Model simulations were conducted for three scenarios: with local traffic adjustment and time-activity simulation (total exposure estimated using combined local and transported pollutant concentrations); without local traffic adjustment but with time-activity simulation (total exposure estimated using air monitoring station data); and without local traffic adjustment or time-activity simulation (annual average ambient concentrations). Personal exposures estimated for these three scenarios are plotted in Fig. 4, where differences between the first and second scenarios reflect within-community variability due to local traffic effects, while differences between the second and third scenarios reflect within-community variability due to time-activity patterns, housing characteristics and air exchange rates. For illustration purposes, only results for CO and PM_{10} are shown.

Local traffic significantly increased within-community variability for exposure to CO, NO_2, and PM-associated pollutants, especially at communities with heavy traffic such as Long Beach, Riverside and San Dimas. The overall within-community variability of personal exposures (including local traffic effects and time-activity differences) were highest for NO_2 ($\pm$20–40%), followed by EC ($\pm$17–27%), PM_{10} ($\pm$15–25%), $PM_{2.5}$ ($\pm$15–20%), and CO ($\pm$9–14%), where the range are across seven CHS communities. Local traffic alone contributed most to CO and NO_2 exposures, but less to PM-related exposures.

Significant within-community variability due to children's time-activity patterns was also observed for all pollutants except CO. By affecting indoor concentrations, pollutant penetration, decay or deposition, as well as air exchange rate, all contributed to the within-community variability due to time-activity patterns. However, since we assumed a unit penetration rate and zero decay rate for CO, indoor concentrations were the same as outdoor and only in-vehicle CO exposures contributed to the within-community variability due to time-activity patterns. The within-community variability of exposure due to children's time-activity patterns followed the pattern of the total within-community variability with the highest for NO_2 ($\pm$18–25%),

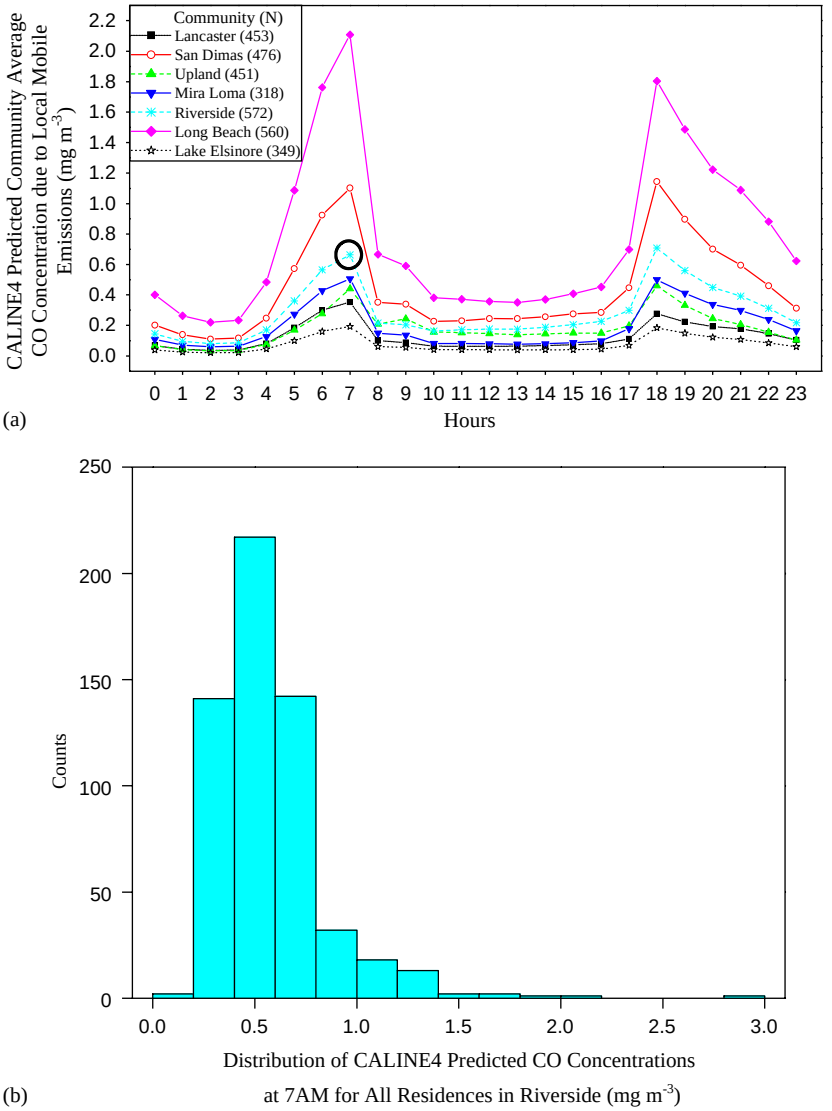

Fig. 3. CALINE4 model-predicted (a) hourly community-mean CO concentrations in the seven CHS communities for the winter season; (b) distribution of CO concentrations at all residences in Riverside at 7 AM for the winter season, the mean concentration of which is circled in (a).

followed by EC (±16–20%), PM_{10} (±12–18%), $PM_{2.5}$ (±10–15%), and CO (±3–9%).

3.4. *Between-community differences in community mean exposures*

Differences in community mean exposures were impacted by community location in the SoCAB (e.g. source vs. receptor areas), traffic density, locations of residences and schools within a community, and time-activity patterns of the children in each community. Since this study was restricted to the Southern California region, children's time-activity patterns and the housing characteristics associated with exposure (e.g. air exchange rate) might not differ greatly between the CHS communities within the SoCAB. However, community location, traffic density, and the relative locations of residences and schools differed significantly between the seven communities. For example, Long Beach was a source site, while Riverside was a typical downwind receptor site. Moreover, Long Beach is crossed by several freeways with high traffic density, such as the I-405, I-710, I-605 and Artesia Freeways, while for Lancaster, State Highway 14 was the only major arterial across the community. Residence and school locations also varied among the communities. Riverside had the

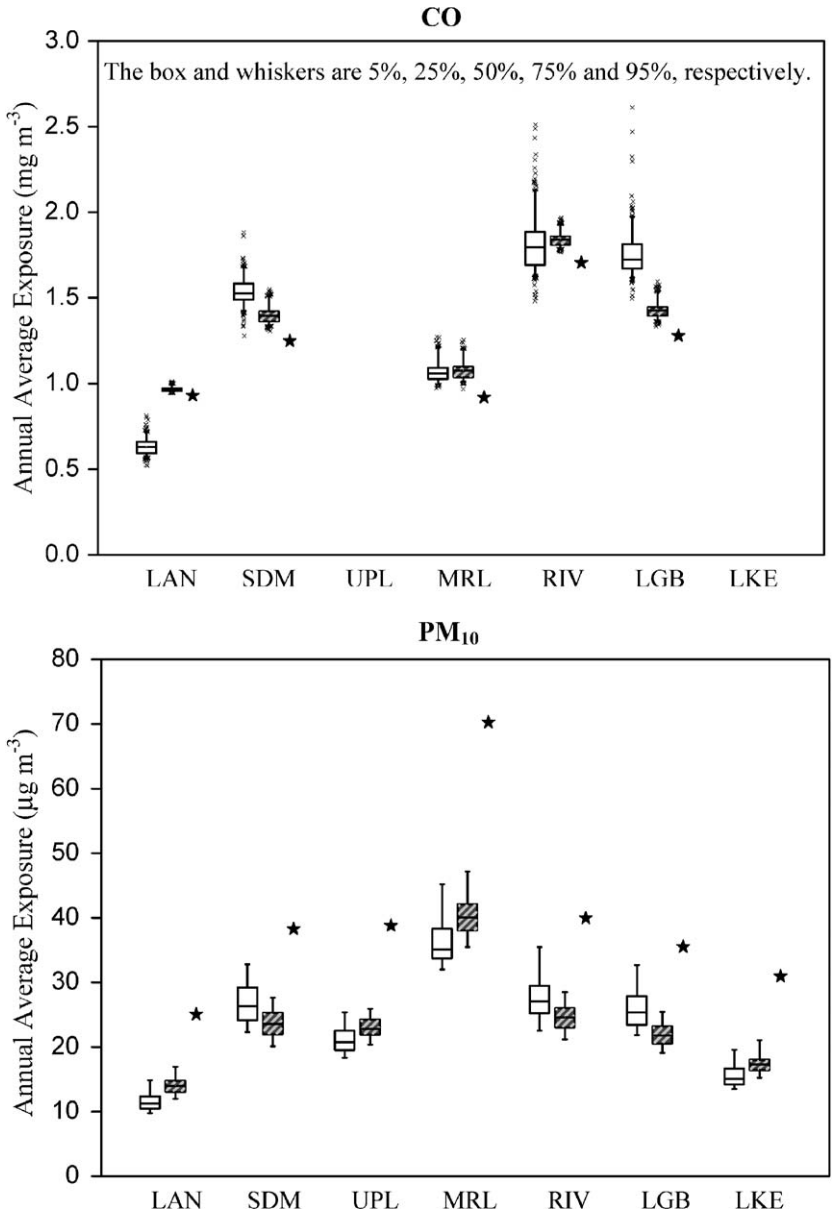

Fig. 4. Estimated annual average exposures of the CHS children in 1997 under three scenarios. ▭, exposure calculated using combined local and meteorologically transported pollutant concentrations; ▨, total exposure calculated using monitoring station data; ★, annual average ambient concentrations. *Note*: no CO exposure assessment was conducted for UPL and LKE due to the lack of measurement data.

highest number of children living close to freeways and had a school (J.W. North High School) located about 200 m from the I-215 Freeway.

All of the above factors affected the community mean exposures and caused them to differ from the ambient concentrations at the corresponding central monitoring station. For CO, the community mean personal exposure ranked RIV (1.8) > LGB (1.7) > SDM (1.5) > MRL (1.0) > LAN (0.7) (community mean exposure in $mg\,m^{-3}$). Although Lancaster and Mira Loma had similar ambient CO concentrations at the CHS station, higher traffic density in Mira Loma led to a higher mean CO exposure than for Lancaster. For PM_{10}, the community mean exposure ranked MRL (36.4) > RIV (28.0) > SDM (26.8) > LGB (26.1) > UPL (21.1) > LKE (15.7) > LAN (11.6) (in $\mu g\,m^{-3}$). Although Long Beach had a lower annual average PM_{10} concentration than Upland (35.5 vs. 38.8 $\mu g\,m^{-3}$) at

the CHS station, it had both a higher traffic density and a higher fraction of residences living close to freeways than Upland, which increased the community mean PM_{10} exposure significantly.

For EC, the community mean exposure ranked LGB (1.4) > SDM (1.2) ≈ RIV (1.2) > MRL (1.0) ≈ UPL (1.0) > LKE (0.5) > LAN (0.4) (in $\mu g\,m^{-3}$). Riverside had a lower annual average EC concentration than Mira Loma and Upland (1.0 vs. 1.2 and 1.1) at the CHS station, but a higher community mean EC exposure. The Riverside station was located in the agriculture fields at the University of California, Riverside, where there was little traffic, which made the station a good indicator of background pollutant concentrations instead of a representative monitor of the community mean exposure. This was especially true given the fact that a significant number of CHS children (34) in Riverside lived within 150 m of freeways and, as noted, the J.W. North High School was located only 200 m from the freeway.

3.5. Personal exposure due to different sources

We estimated personal exposure due to pollutant transport and LN-MSE, LMSE, and in-vehicle exposures (Fig. 5). For CO, transport and LN-MSE contributed most to the total exposures, while LMSE and in-vehicle exposures explained most of the variability in personal exposures. Similar results were observed for NO_2 and EC except that a higher fraction of variability was explained by pollutant transport and LN-MSE (at Riverside LMSE contributed the most to the total exposures). For PM_{10}, as we expected, transport and LN-MSE contributed most, and also accounted for a significant amount of variability in the total personal exposures. LMSE only contributed significantly to the exposure variability at Riverside, Long Beach, San Dimas and Upland.

3.6. Personal exposure in different microenvironments

Indoor locations were still found to be the most significant microenvironments for exposures to all pollutants. For CO, residential indoor, in-vehicle, school indoor, exposure occurring elsewhere, residential outdoor, and school outdoor accounted for 50–70%, 12–20%, 6–20%, 7–10%, 3–4%, and 1–3% of the total exposures, respectively, for the seven communities studied. For EC, residential indoor, in-vehicle, school indoor, exposure occurring elsewhere, residential outdoor, and school outdoor accounted for 36–56%, 21–38%, 7–26%, 7–9%, 4–6% and 2–5% of the total exposures, respectively, for the seven communities. Consistent with the findings of Fruin et al. (2004), in-vehicle EC exposure was approximately one third of the total EC exposures, indicating that accounting for time spent in vehicles was essential in characterizing exposures to elemental carbon.

4. Conclusions

This study integrated two distinct air quality models into a new IEM in order to estimate comprehensive exposure due to pollutants arising from transport, LN-MSE and LMSE, and to investigate quantitatively the contributions of these sources to total personal exposures. We found that for this children's cohort, local traffic significantly increased within-community variability for exposure to CO, NO_2, and PM-associated pollutants, especially in communities with heavy traffic. Exposures to PM-associated pollutants were impacted more by transport and LN-MSE at all communities except Riverside where a larger number of children lived close to freeways and major arterials. Significant within-community variability due to time-activity pattern differences was observed for all pollutants except CO (since we assumed a unit penetration and zero decay rate for CO). Time spent in vehicles was more important in determining exposures to elemental carbon than for other pollutants.

Between-community exposure differences were affected by community location, traffic density, locations of residences and schools, and time-activity patterns of the children in each community. The ambient pollutant concentrations measured by the CHS central monitoring stations did a reasonable job of capturing the range of residence and school outdoor concentrations at Lancaster, Lake Elsinore, Upland, and Mira Loma, but not at San Dimas, Riverside, and Long Beach. This monitoring station "siting" issue has implications for epidemiological studies since many of these studies use ambient concentrations as surrogates for personal exposures. As shown here by the example of the Riverside CHS station, a monitoring station located too far away from significant traffic sources will not be representative of the overall community exposures to vehicle-related pollutants. Conversely, a station located too close to major traffic sources will overestimate personal exposures in that community.

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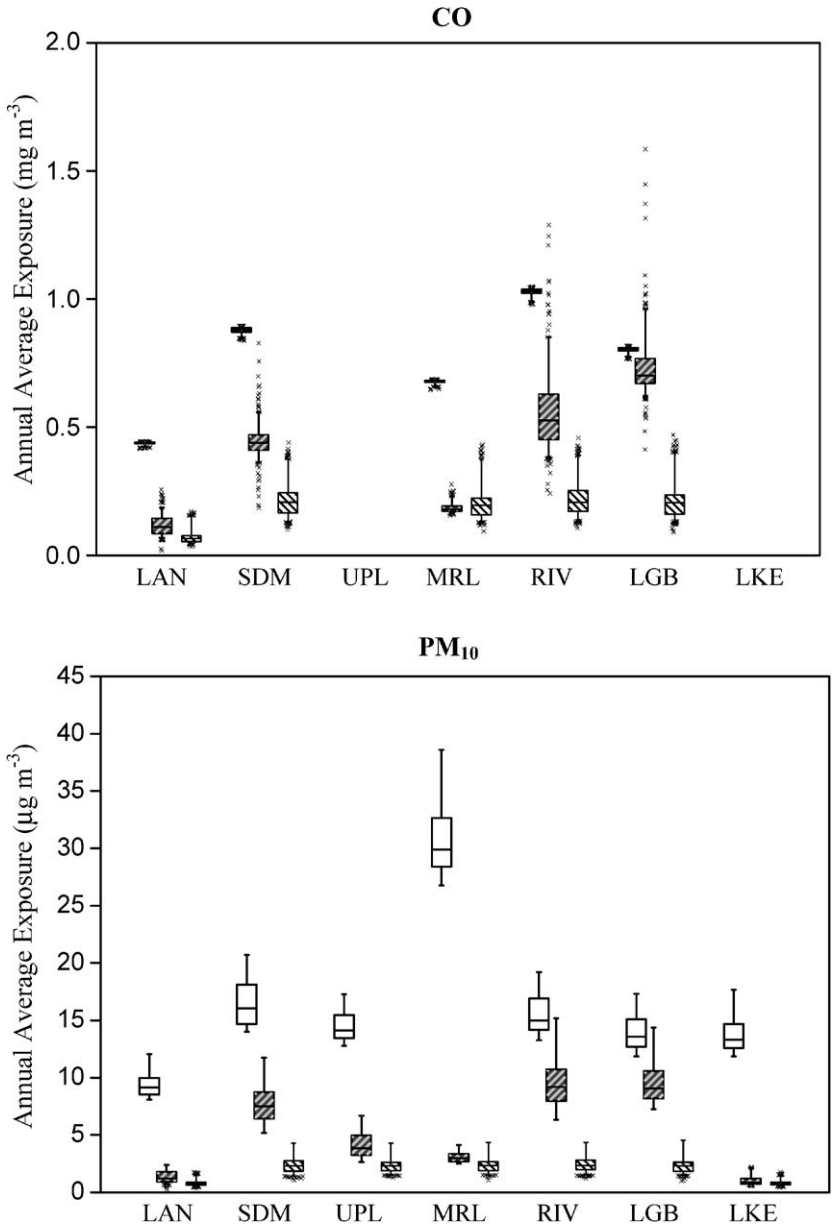

Fig. 5. Estimated annual average exposures of the CHS children in 1997 due to different sources. ▭, exposure due to meteorologically transported pollutants and LN-MSE; ▨, exposure due to LMSE; ▩, exposure due to travel in vehicles. Note: no CO exposure assessment was conducted for UPL and LKE due to the lack of measurement data.

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