Black carbon concentrations in California vehicles and estimation of in-vehicle diesel exhaust particulate matter exposures

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1. Abstract

This research assessed in-vehicle exposures to black carbon (BC) as an indicator of diesel particulate matter (DPM) exposures. Approximately 50 h of real-time Aethalometer BC measurements were made inside vehicles driven on freeway and arterial loops in Los Angeles and Sacramento. Video tapes of the driver's view were transcribed to record the traffic conditions, vehicles followed, and vehicle occupant observations, and these results were tested for their associations with BC concentration. In-vehicle BC concentrations were highest when directly following diesel-powered vehicles, particularly those with low exhaust pipe locations. The lowest BC concentrations were observed while following gasoline-powered passenger cars, on average no different than not following any vehicle. Because diesel vehicles were over-sampled in the field study, results were not representative of real-world driving. To calculate representative exposures, in-vehicle BC concentrations were grouped by the type of vehicle followed, for each road type and congestion level. These groupings were then re-sampled stochastically, in proportion to the fraction of statewide vehicle miles traveled (VMT) under each of those conditions. The approximately 6% of time spent following diesel vehicles led to 23% of the in-vehicle BC exposure, while the remaining exposure was due to elevated roadway BC concentrations. In-vehicle BC exposures averaged 6 $\mu g\ m^{-3}$ in Los Angeles and the Bay Area, the regions with the highest congestion and the majority of the state's VMT. The statewide average in-vehicle BC exposure was 4 $\mu g\ m^{-3}$, corresponding to DPM concentrations of 7–23 $\mu g\ m^{-3}$, depending on the Aethalometer response to elemental carbon (EC) and the EC fraction of the DPM. In-vehicle contributions to overall DPM exposures ranged from approximately 30% to 55% of total DPM exposure on a statewide population basis. Thus, although time spent in vehicles was only 1.5 h day^{-1} on average, vehicles may be the most important microenvironment for overall DPM exposure.

Author Keywords: Exposure assessment; In-vehicle concentrations; Diesel vehicle emissions; Aethalometer

2. Article Outline

3. 1. Introduction

In-vehicle exposures to vehicle-related pollutants are frequently high, due to a vehicle's proximity to relatively undiluted emissions from other vehicles, the typically rapid air exchange rate inside vehicles (e.g., Park et al., 1998), and the average 95 min day^{-1} spent in the in-vehicle microenvironment (Klepeis et al., 2001). In-vehicle pollutant concentrations are up to 10 times higher than ambient levels for exhaust- or gasoline-related volatile organic compounds (VOCs) (e.g., Jo and Park, 1999; Leung and Harrison, 1999; Duffy and Nelson, 1997; Lawryk et al., 1995; Chan et al., 1991). Therefore, the in-vehicle route of exposure can contribute a significant fraction of a person's overall exposure to vehicle-related pollutants (Marshall et al., 2003; Kim et al., 2002; Rea et al., 2001; Chan et al., 1991).

Although both diesel exhaust (Lipsett and Campleman, 1999; Bhatia et al., 1998) and diesel particulate matter (DPM) (ARB, 1998a) are suspected carcinogens, in-vehicle DPM exposures have not been well characterized. Accurate in-vehicle exposure assessments are needed to provide a direct measure of the exposure-reduction benefits of more stringent DPM emission standards. For example, federal heavy-duty diesel (HDD) particulate matter (PM) emission standards were first promulgated for 1988 and newer model years and will have been reduced by nearly two orders of magnitude for model years 2007 and later.

The objective of this research was to calculate in-vehicle BC and DPM exposures based on in-vehicle measures of BC. To do this we first measured BC using an Aethalometer inside an instrumented vehicle driven over arterial roads and freeways, under rush-hour and non-rush-hour conditions, in Los Angeles and Sacramento (Rodes et al., 1998), and we observed BC to be a sensitive marker of diesel exhaust. The sensitivity of the Aethalometer to diesel exhaust reflects the relatively high fraction of elemental carbon (EC) in DPM (Kleeman et al., 2000; Shi et al. 2000; Birch and Cary, 1996; Zaebst et al., 1991), and the strong association between EC and BC as measured with an Aethalometer (Moosmüller et al., 2001; Babich et al., 2000; Allen et al., 1999). Although EC is also present in PM emissions from gasoline-powered vehicles, PM emissions from HDD vehicles are significantly higher on a distance or fuel-burned basis than gasoline-powered light-duty (LD) vehicles, as has been observed in tunnel studies. For example, the studies of Kirchstetter et al. (1999) and Miguel et al. (1998), measuring emissions in the Caldecott Tunnels near San Francisco, CA, found BC emission rates 37–48 times higher for HDD vehicles compared to LD vehicles on a mass of fuel consumed basis, as shown in Table 1.

Table 1. Heavy- and light-duty vehicle emission rates measured in the Caldecott Tunnel, CA

Pollutant	Heavy duty vehicle emission rate($g\,kg^{-1}$ fuel)	Light duty vehicle emission rate($g\,kg^{-1}$ fuel)	Ratio	Year of study	Reference
PM2.5	2.5 ± 0.2	0.11 ± 0.01	24	1997	Kirchstetter et
BC	1.3 ± 0.3	0.035 ± 0.003	37	1997	Kirchstetter et
BC	1.4 ± 0.16	0.030 ± 0.002	48	1996	Miguel et al.

We used the in-vehicle BC measurements and statewide vehicle activity and volume data to estimate in-vehicle exposures to DPM. Because diesel vehicles were purposefully followed during the field measurements, we needed to compensate for the unusually high diesel vehicle influence. We did this by conducting an analysis of the video tapes made during these runs (driver's view), linking in-vehicle BC concentrations to the vehicle being followed, along with other traffic-related observations. We then compiled distributions of BC concentrations measured while behind each of the major vehicle types (diesel vehicles were grouped by axle-number) and used a stochastic exposure model to sample from these distributions in a manner simulating typical driving in California. Typical driving was characterized by average driving times from time-activity diary surveys; relative congestion based on traffic volume per road mile; and estimated fractions of time driving behind each vehicle type.

4. 2. Methods

5. 2.1. Field study measurements

The field study (Rodes et al., 1998) was conducted in the fall of 1997, and consisted of 29 two-hour runs over a variety of road types (collector, major and minor arterial, highway, interstate, carpool), with 13 runs conducted in Sacramento and 16 in Los Angeles.

For this field study, a 1991 Chevrolet Caprice sedan was outfitted with instrumentation to continuously record 60-s averages of carbon monoxide (CO), fine particle count concentrations, and BC concentrations, along with integrated concentrations of speciated VOCs and PM mass. Instruments were mounted on a platform designed to damp vibration. The minute averages of CO, particle counts and BC alternated between inside the vehicle and outside (from the base of the windshield) by means of a solenoid valve. The outside sampling line pulled air continuously, minimizing lag time to less than two seconds. Windows were closed for all runs, and fan settings were set to either "high" or "low" for the duration of each run. Heaters were never used; AC was set as needed for comfort, but used for most of the runs (24 of 29), with five of the 24 runs set at maximum AC. "Vent only" was used for the other five runs.

Speed was measured with a digital transducer mounted on the drive shaft, accurate to ±10%, while following distance was measured by a grill-mounted, pulsed laser range finder (Laser Atlanta, Norcross, Georgia) that used pulse reflection times to calculate distance, accurate to two feet. Speed and following distance were recorded as 60 s averages on a laptop computer.

Each run was recorded via video camera aimed to capture the driver's forward view. Time of day (a.m./p.m., rush-hour/non-rush-hour), day of week, predominant road type, and ventilation (high/low) were fixed for each run. Diesel vehicles were targeted for following, resulting in diesel vehicles being followed more frequently and for longer periods than expected for normal driving.

Fine particle counts in 12 size bins (0.15–2.5 μm) were measured with a LAS-X optical particle counter (Particle Measuring Systems, Boulder, CO). Calibrations were performed by Aerosol Dynamics (Berkeley, CA) for each bin size using ambient Berkeley air, sized by a differential mobility particle sizer. Flow rates were kept at 1 cc s^{-1}, checked daily with a bubble meter, and were within 10% for all runs.

6. 2.2. Black carbon measurements

To measure black carbon (BC) concentrations we used a Model AE16 Aethalometer (Magee Scientific, Berkeley, CA), which is based on the principle of detecting the changing optical absorption of light transmitted through the accumulated particles captured on a quartz fiber filter tape. Light came from a high-intensity, light-emitting diode operating at a wavelength of 880 nm. A flow rate of 5 lpm was used, controlled by an internal mass flow meter. The minimum detection limit was approximately 0.2 μg m^{-3}. The manufacturer's calibration was used, which assumed an 880 nm specific attenuation cross section for BC of 19 m^2 g^{-1}.

When the deposit of particles on the Aethalometer filter tape became sufficiently opaque, measuring the relative changes in attenuation became less accurate, and the instrument automatically advanced the filter tape. Several minutes were required to advance the filter tape and reach equilibrium. This resulted in 37 gaps of 2–4 min during the approximately 50 h of measurement in this field study. These gaps amounted to 1.5% of the Sacramento measurements and about 6% of the Los Angeles measurements, but because the opacity threshold was more likely to be exceeded during times of high concentrations and rapid filter build-up, it was important to estimate BC concentrations during these gaps. Because the correlation between the Aethalometer and the LASX optical counter was high (Pearson's r^2 averaged 0.82), we used fine particle counts to estimate any missing BC data using the best-fit least squares fit of the LASX total particle count concentrations versus BC. The average equation used was:

$$\text{BC } [\mu\text{g m}^{-3}] = (\text{fine particle counts cc}^{-1} \times 0.0034) - 6.1 \text{ (Los Angeles)}, \tag{1}$$

$$\text{BC } [\mu\text{g m}^{-3}] = (\text{fine particle counts cc}^{-1} \times 0.0045) + 2.0 \text{ (Sacramento)}. \tag{2}$$

Fig. 1 shows an example of the excellent correlation between fine particle counts and BC for a sample of the 2-h runs (#27 of 29).

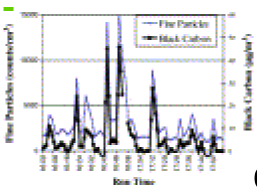
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Fig. 1. Fine particle counts versus BC concentration for run 27 of 29, Los Angeles.

7. 2.3. Generation of traffic observations database from video tape

To correct for non-representative driving, and to aid in further analysis of the in-vehicle concentration data, we analyzed videotape records of each run. We used these observations to assign labels to each 60-s BC concentration based on the type of vehicle followed, number of axles, location of exhaust, visibility of exhaust, and freeway or road number. Where a variable could not be unambiguously observed on tape, the label was designated as uncertain and these were excluded from further analysis. However, these instances were rare: the exhaust height and the number of axles of a followed diesel vehicle were identified with certainty 90% and 95% of the time, respectively. For all vehicles followed, the fuel type was identified with certainty 99.5% of the time. Of the 13 diesel-powered passenger cars, all were clearly identifiable as diesel-powered by manufacturer and model number.

8. 2.4. Treatment of serial correlation

To minimize serial correlation (thereby ensuring statistical independence of the BC data) yet also retain the vehicle-specific information in the data, we averaged the BC data for each individual vehicle followed. This reduced serial correlation to non-significant ($p<0.05$) levels, as measured via the standard autocorrelation function (Box and Jenkins, 1976).

9. 2.5. Corrections for unusual driving

We ran multiple regression tests on the labeled BC data set to determine which of the observation labels were most strongly predictive of in-vehicle BC concentration. Because the type of vehicle followed was the best predictor of BC, we made adjustments for purposeful following of diesel vehicles by grouping the BC data into followed-vehicle scenarios that captured most of the variability in the BC data, but could also be re-sampled to simulate realistic driving. These followed-vehicle scenarios were:

• No target followed or gasoline-powered passenger car (GPPC), on arterial roads;
• No target or GPPC, on freeways;
• Low-exhaust, 2-axle, diesel-powered bus;
• Other 2-axle diesel vehicle; and
• 3–5-axle diesel vehicle.

To estimate the frequency of occurrence for each of the above scenarios, we used California traffic count and vehicle miles traveled (VMT) data (US DOT, 2002). We assumed the average time spent following diesel vehicles was proportional to the diesel vehicle fraction of total VMT. That is, if 5% of the vehicle miles traveled on a length of freeway were HDD truck miles, then on average, 5% of the passenger car fleet would be following a HDD truck during typical travel on that freeway.

In-vehicle exposure is a function of the time spent in the vehicle rather than mileage driven. Because arterial road speeds are lower than freeway speeds, time spent on each road type was not proportional to VMT, but rather VMT divided by average speed. The average arterial and freeway speeds in the field study were 22 and 41 mph, respectively, in Los Angeles, and 24 and 45 mph, respectively, in Sacramento. The resulting adjustment factors used to increase arterial road time were 1.9 for Los Angeles and 1.7 for Sacramento.

To account for differences in traffic density in various parts of California, we apportioned California VMT into areas of light, moderate, and heavy congestion by county based on the ratio of county total daily VMT (DVMT) to total county road miles (DVMT/road-mile) (CalTrans, 1998). This ratio is a frequently used measure of congestion (Schrank and Lomax, 1999). For high and moderate congestion conditions we used the BC measurements made in Los Angeles and Sacramento, respectively. We assumed rural VMT to be lightly congested and to have BC concentrations equal to ambient air.

10. 2.6. Exposure modeling

To calculate in-vehicle BC exposures during realistic driving, we re-sampled the BC data from the followed-vehicle scenarios, weighted by their expected frequency of occurrence, using the California Population Indoor Exposure Model (CPIEM) (Koontz et al., 1998; ARB, 1998b). The CPIEM is a stochastic model that samples from concentration distributions for up to nine microenvironments. It contains a database of 2962 24-h sequences of time-activity patterns, including time spent in vehicles, based on the ARB's 1987–1990 survey of Californians (Jenkins et al., 1992). Only BC concentrations from inside the vehicle were used in estimating exposure.

We converted the in-vehicle BC exposures to a range of DPM exposures based on published studies comparing the Aethalometer with various EC measurement methods, and studies that measured the EC content of DPM. We then compared this range of in-vehicle DPM exposures with previously calculated estimates of total DPM exposure (ARB, 1998a), to judge the relative importance of the in-vehicle exposures.

11. 3. Results and discussion

12. 3.1. Factors affecting in-vehicle concentrations

The type of vehicle followed was the most important predictor of in-vehicle BC concentrations, accounting for 74% and 34% of the variability in the measurements for Los Angeles and Sacramento, respectively. Other variables such as speed, following distance, ventilation rate (high versus low), and time of day (morning versus afternoon) were not good predictors. Mean BC concentrations associated with following specific vehicle categories are listed in Table 2 and show a range of more than an order of magnitude.

Table 2. Mean in-vehicle BC concentrations associated with following certain vehicle types

Vehicle type followed	Los Angeles			Sacramento
	BC (μg m^{-3}) AM±SD	No. instances	No. minutes	BC (μg m^{-3}) AM±SD
GPPC	4.1 ± 6.7	196	523	2.2 ± 3.0
No target followed	4.8 ± 5.4	138	394	3.4 ± 4.6
3-axle vehicles, diesel	11 ± 9	16	76	7.9 ± 4.8
Tractor trailer (5-axle)	13 ± 10	47	291	11 ± 11
Transit bus, high exhaust	16 ± 13	13	50	1.5 ± 1.7
Passenger car, diesel	19 ± 20	6	22	4.7 ± 4.0
Delivery truck, low exhaust	21 ± 22	14	62	16 ± 5.5
Transit bus, low exhaust	92 ± 117	16	93	na

GPPC=gasoline-powered passenger car; AM=arithmetic mean; ASD=arithmetic standard deviation.

The BC concentrations encountered while following a GPPC were not statistically distinguishable from those encountered while not following any vehicle (Mann-Whitney rank sum test). Only 5% of the 355 GPPCs followed were associated with significant in-vehicle BC concentrations, i.e., >10 μg m^{-3}, and half of these were due to the observable influence of nearby diesel vehicles. Only 1% of GPPCs were visibly smoking and associated with significant BC concentrations. Thus, BC emissions from well-tuned, GPPCs meeting California emission standards were rarely significant.

Vehicle exhaust pipe height had a strong effect on BC concentrations. Diesel transit buses with ground-level exhaust were associated with in-vehicle BC concentrations 6 times higher (92 μg m^{-3}) than high exhaust buses (16 μg m^{-3}). This effect was also observed comparing low exhaust delivery trucks to tractor-trailer rigs, which always had high exhaust locations.

13. 3.2. In-vehicle BC data re-sampling

To calculate population-average, in-vehicle BC exposures using the BC data collected in the field study, we needed to adjust for the purposeful following of diesel vehicles. We accomplished this through re-sampling of the data to reflect realistic driving. This required three steps: grouping the data into followed-vehicle scenarios; relating the Los Angeles and Sacramento data the rest of California; and determining the expected frequency of each followed-vehicle scenario.

14. 3.2.1. Followed-vehicle scenarios

We used the CPIEM model to re-sample from the log-normal concentration distributions shown in Table 3, based on the five followed-vehicle scenarios listed for each city. (Arithmetic means and standard deviations are also provided.) These followed-vehicle scenarios were selected based on three findings. First, as noted earlier, the vehicle followed was the most important predictor of in-vehicle BC. Second, available traffic volume data were generally grouped by axle number. Third, GPPCs were not distinguishable from not following any vehicle, and when GPPCs or no vehicle were being followed, arterial roads had significantly lower concentrations than freeways (Mann-Whitney, p<0.001, both cities).

Table 3. Characteristics of BC concentration distributions by vehicle type followed

Vehicle type followed	Los Angeles BC (μg m^{-3})	BC (μg m^{-3})		No. instances	Sacramento BC (μg m^{-3})	BC (
	AM±SD	GM	GSD		AM±SD	GM
GPPC/no target, arterial roads	4.0 ± 12	1.3	4.5	34	1.2 ± 2.4	0.5e
GPPC/no target, freeways	6.9 ± 8.2	4.4	2.6	50	5.5 ± 5.7	3.8
Diesel vehicles, 2-axle	19 ± 19	11	2.9	40	10 ± 15	5.9
Diesel vehicles, 3–5-axle	14 ± 11	11	2.0	63	11 ± 14	7.3
Transit buses, low exhaust	100 ± 210	43	3.7	16	20 ± 10	18

GPPC=gasoline-powered passenger car; AM=arithmetic mean; ASD=arithmetic standard deviation; GM=geometric mean; GSD=geometric standard deviation.

We found these five followed-vehicle scenarios (in Table 3) to be the distributions that captured most of the variability in the BC data, while still being linkable to the traffic volume data needed to accurately estimate expected frequencies of occurrence. The distributions of BC concentrations for these scenarios were found to be log-normal (Shapiro Wilks' *W* test for $n<50$ and the D'Agostino test for $n>50$, $p<0.05$) (Gilbert, 1987). For ease of modification, the distributions used in the model (Table 3) were the best-fit theoretical log-normal distributions for each scenario. These were derived from the linear regression equations of the probability plots of the natural logarithms of the BC concentrations (average Pearson's r^2=0.93).

15. 3.2.2. Relating Los Angeles and Sacramento data to California

Traffic volumes and congestion levels varied greatly between Sacramento and Los Angeles, and these differences were reflected in the different BC concentrations observed for the same followed-vehicle scenario. This was especially pronounced for the GPPC/no target scenarios, where BC concentrations reflected roadway "baseline" concentrations. We felt these congestion differences were the most important traffic-related difference between various regions of California, so to account for these differences, California counties were categorized into high, medium, or low congestion, based on the DVMT/road-mile ratio. As noted, we assumed high and moderate congestion areas were best represented by the BC measurements made in Los Angeles and Sacramento, respectively.

The distribution of California county DVMT/road-mile had three natural groupings, as shown in Fig. 2. The high-congestion counties of the Los Angeles Basin and the San Francisco Bay Area (DVMT/road-mile weighted average of 9300) accounted for the majority of the state's VMT, even though they were concentrated in two geographically small areas of the state. The remaining urban areas (weighted DVMT/road-mile average of 5700) include the Sacramento area. Rural roads (weighted average of 1900) were assumed to have low BC concentrations, with rural driving having little impact from following diesel vehicles, so rural roads were excluded from further analysis.

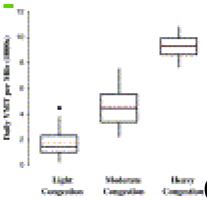(5K)

Fig. 2. Daily VMT per road mile, California county averages.

16. 3.2.3. Expected frequencies of followed-vehicle scenarios

We assumed the average time spent behind diesel vehicles during typical driving is proportional to the average fraction of diesel vehicles on the road, since vehicles mix freely and randomly together, and vehicles in moderately or highly congested conditions are usually behind other vehicles. The statewide average drive-time fractions by followed-vehicle scenario are presented in Table 4 and were derived by vehicle axle number fraction for the year 1997 (CalTrans, 1998). The GPPC/no target fractions for arterial and freeway VMT were adjusted for speed differences.

Table 4. Fractions of vehicle-type following-time fractions by congestion type and relative contributions to in-vehicle exposure

	Followed-vehicle scenario frequency of occurrence (%)	Relative contribution to in-vehicle exposure (%)
Counties with heavy congestion		
GPPC/no target, arterial roads	38±4.6	34
GPPC/no target, freeways	15±1.8	26
3–5-axle diesel	2.4±0.34	7.7
Other 2-axle diesel	1.9±0.17	10
2-axle, low exhaust bus	0.07±0.02	1.2
Heavy congestion county total	57	79
Counties with medium congestion		
GPPC/no target, arterial roads	15±1.6	4.4
GPPC/no target, freeways	6.9±0.8	8.8
3–5-axle diesel	0.97±0.13	2.6
Other 2-axle diesel	0.74±0.06	1.6
2-axle, low exhaust bus	0.03±0.01	0.0
Medium congestion county total	24	17
Non-urban counties/low congestion	19	4
Total frequency of occurrence	100	100
Total for all diesel vehicles	6.1	23

GPPC=gasoline-powered passenger car; VMT=vehicle miles traveled.

Over shorter time periods, or on a vehicle-by-vehicle basis, the fraction of diesel vehicles on the road will vary by time of day, but time-resolved vehicle count information is limited to relatively few weigh-in-motion locations on freeways. For freeways in and around Los Angeles, for example, Chinkin et al. (2003) compiled hourly changes in LD and HDD vehicle counts at eight weigh-in-motion stations. During the weekday rush-hour GPPC peaks at 7 a.m. and 5 p.m., HDD vehicles made up about 3% of the vehicles counted (Chinkin et al., 2003), while during weekday midday HDD traffic peaks, HDD vehicles made up about 6% of the vehicle counts (Chinkin et al., 2003). Other times of the day had HDD fractions between these extremes, except for early morning hours (midnight to 4 a.m.) when HDD fractions were about 10%, and during weekends when HDD fractions were less than a percent (Chinkin et al., 2003), but these two periods account for a small fraction of overall GPPC VMT. The HDD (3–5-axle) diesel VMT fractions for high and moderate congestion of 4.2% and 4.0%, respectively, (2.4÷57 and 0.97÷24 from Table 4) are quite close to midway between the 3% HDD volume fraction during rush hour and the 6% HDD volume fraction at peak midday truck traffic found by Chinkin et al. (2003).

17. 3.3. In-vehicle BC exposures

The results of the CPIEM re-sampling of the five followed-vehicle scenarios, weighted for their expected frequency of occurrence, are shown in Table 5 for high and moderate congestion areas. In-vehicle BC exposures in high congestion areas were nearly twice those of moderate congestion areas. The exposures were also highly skewed, with the arithmetic average four to five times higher than the 50th percentile. The importance of adjusting for the purposeful following of diesel vehicles can be seen by comparing the unadjusted field study measurements listed in the last column, which are three times larger than the calculated exposures.

Table 5. Adjusted versus unadjusted average BC concentrations in vehicles

	Adjusted in-vehicle BC concentration ($\mu g\, m^{-3}$) AM±ASD	In-vehicle BC concentration by percentile ($\mu g\, m^{-3}$)		Unadjuste tions from m^{-3}) AM±ASD
		50th	90th	
Los Angeles/high congestion	5.9±18	1.3	12.7	16±51
Sacramento/moderate congestion	3.1±9.3	0.64	7.4	9.0±11
Statewide average	4.1±12	0.88	9.3	

The last column of Table 4 shows the relative contribution of each driving scenario to overall in-vehicle exposures. Despite the high concentrations measured when following diesel vehicles, the relatively modest 6.1% of time spent directly behind diesel vehicles only resulted in 23% of the total in-vehicle BC population exposure. Sixty percent of statewide in-vehicle BC exposure occurred during the GPPC/no target scenario in areas of high congestion due to roadway "background" concentrations of BC.

18. 3.4. BC-to-DPM conversion

To convert the measured BC concentrations into DPM concentrations we derived conversion factors based on other published studies of the Aethalometer response to EC and the EC content of DPM. The relationship between BC and EC can depend on the optical characteristics of the aerosol being measured (Liousse et al., 1993), so a range of values was used based on the literature. The largest comparison study of BC and EC was conducted in six US cities by Babich et al. (2000). EC was analyzed using the thermal/optical reflectance (TOR) methods as discussed in Chow et al (1993) and Chow et al (2001). Babich et al. (2000) derived the regression equation:

$$\mathrm{BC}\ (\mu\mathrm{g\ m^{-3}})=\mathrm{EC}\ (\mu\mathrm{g\ m^{-3}})\times 0.765(\pm 0.014)-0.002(\pm 0.034)\ (r^2=0.94). \quad (3)$$

The slope ranged from 0.62 to 0.83 by city.

Another comparison between BC and EC via TOR was conducted by Allen et al. (1999) in semi-urban and suburban Pennsylvania, and yielded the regression equation:

$$\mathrm{BC}\ (\mu\mathrm{g\ m^{-3}})=\mathrm{EC}\ (\mu\mathrm{g\ m^{-3}})\times 0.95(\pm 0.04)-0.2(\pm 0.04)\ (r^2=0.93). \quad (4)$$

Park et al. (2002), comparing BC to EC determined by thermal manganese dioxide oxidation (Fung, 1990) found BC/EC slopes of 0.93 and 1.07 (r^2=0.99, 0.92, respectively) in Seoul and Kwangju, two large Korean cities.

In a dynamometer setting, BC and EC by TOR were compared by Moosmüller et al. (2001) using diluted diesel vehicle exhaust. The corresponding relationship (adapted from Moosmüller et al., 2001) was:

$$\mathrm{BC}\ (\mathrm{mg\ m^{-3}})=\mathrm{EC}\ (\mathrm{mg\ m^{-3}})\times 1.41+0.044\ (r^2=0.94). \quad (5)$$

Because roadway EC aerosols may share optical properties from both fresh diesel exhaust and aged urban aerosol, the full range of possible BC/EC ratios cited above was used, 0.62–1.4 or 1.0±40%.

The fraction of DPM that is EC is also known to vary. Shi et al. (2000) found the EC fraction of DPM to increase with both speed and load. They tested a 1995 diesel vehicle over a variety of loads and found the EC fraction of DPM to vary from about 25% at 25% load and 1600 rpm, to about 52% at 100% load and 2600 rpm. There is also limited evidence that the EC fraction of the diesel vehicle fleet may be decreasing over time. Pierson and Brachaczek (1983) measured a diesel fleet mass emission rate of 870 mg km^{-1}, 55% of which was EC, while Hildemann et al. (1991) found the EC fraction to be about 40% for 1986 year truck engines. Therefore, a range of DPM EC fraction of 25–50% was assumed, or 0.375±33%.

We assumed the BC to EC ratio and the EC to DPM ratio can vary independently, a conservative assumption to give the largest range of possible conversion factors. If the BC/EC ratio of 1.0±40% is combined with the EC fraction in DPM of 0.375±33% in a root-mean-square calculation, appropriate for independently varying uncertainties, a BC/DPM ratio range of 0.375±52% is obtained, or 0.18–0.57. The resulting BC-to-DPM conversion factors are then 1.8–5.6. These conversion factors are consistent with those derived in the 1997 tunnel study work of Kirchstetter et al. (1999) for measurements in the Caldecott Tunnels. Their BC/DPM ratio of 51±11% gave a BC-to-DPM conversion factor range of 1.6–2.5. Our conversion factors are 1.8–2.7 if the low load conditions of

Shi et al. (2000) are not used. Such low loads would not be expected in the Kirchstetter et al. (1999) study with the tunnel uphill grade of 4.2% and 40±7 mph average speeds.

19. 3.5. In-vehicle DPM concentrations

Table 6 presents in-vehicle BC concentrations converted to a range of DPM concentrations, using the above range of conversion factors. For comparison, ambient DPM concentrations estimated from Chemical Mass Balance (CMB) Model studies conducted in California are listed. These included CMB studies in San Joaquin Valley (DRI, 1990; Chow et al., 1992), San Jose (Chow et al., 1995), and the Los Angeles area (SCAQMD, 1991). The average in-vehicle DPM concentrations were consistently 5–14 times the ambient DPM concentrations throughout California.

Table 6. DPM concentrations in vehicles and ambient air

	In vehicle DPM concentration ($\mu g\ m^{-3}$) AM±ASD		Ambient DPM conc. ($\mu g\ m^{-3}$)	R...
	Low range	High range		
Los Angeles/high congestion	11 ± 32	33±99	2.4	4
Sacramento/moderate congestion	5.6 ± 17	17±52	1.2	4
Statewide average	7.3 ± 22	23±69	1.8	4

20. 3.6. In-vehicle DPM exposures

Table 7 compares 24-h DPM exposures as calculated by the Air Resources Board (ARB, 1998a) for the year 2000, which did not take in-vehicle exposures into account, with the resulting exposures including the in-vehicle microenvironment. The ARB 24-h average exposures were about two-thirds of the calculated ambient concentrations, lower because of particle losses in indoor environments. The in-vehicle exposure contribution to total DPM exposure ranged from about 30% to about 60%, (depending on the BC-to-DPM conversion factor), a very significant fraction of the total DPM exposure.

Table 7. Average 24-h DPM exposures, with and without in-vehicle concentrations

	Estimated CA 24-h DPM exposure, not including in-vehicle exposure, year 2000 ($\mu g\, m^{-3}$) AM±ASD	Range of CA 24-h mean DPM exposure, including contribution from vehicles, year 2000 ($\mu g\, m^{-3}$)		Range of pe[illegible] in-vehicle ex[illegible] sure (%)	
		Low	High	Low	High
Los Angeles/high congestion	1.6±0.58	2.3	3.7	31	
Sacramento/moderate congestion	0.80±0.29	1.2	1.9	31	
Statewide average	1.2±0.42	1.7	2.7	28	

21. 4. Conclusions

Aethalometer BC measurements, along with videotape analysis, allowed unambiguous determination of the sources of high in-vehicle BC concentrations while driving on arterial roads and freeways in Los Angeles and Sacramento. In-vehicle BC concentrations were highest when following diesel-powered vehicles, especially those with low exhaust pipes, such as most delivery trucks and about half of the transit buses in Los Angeles. The lowest BC concentrations were observed while following gasoline-powered passenger cars, whose impacts on BC concentration were, on average, no different than not following any vehicle. However, due to the small fraction of time diesel vehicles are followed in typical driving, about 6%, most in-vehicle exposure appeared to occur during the 94% of time drivers were not behind diesel vehicles, due to elevated roadway concentrations of BC. Statewide, most BC exposure occurred in Los Angeles and the Bay Area, the California regions with the highest congestion and the majority of the state's VMT.

Conversion factors necessary to convert the BC concentrations to DPM concentrations were calculated to range from 1.8 to 5.6, based on literature-cited studies of the Aethalometer response to EC, and dynamometer and tunnel studies of the EC fraction of DPM. Compared to CMB Model-derived estimates of ambient DPM concentrations and estimated indoor concentrations, in-vehicle DPM exposures appear to make very significant contributions to overall DPM exposures, ranging from approximately 30–55% of total DPM exposure on a statewide, population basis. Thus, the in-vehicle microenvironment may be the most important route of overall DPM exposure, although only 1.5 h day^{-1} is spent there, on average.

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