CHARACTERIZING THE RANGE OF CHILDREN'S POLLUTANT EXPOSURE DURING SCHOOL BUS COMMUTES

FINAL REPORT

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Principal Investigator

Dennis R. Fitz

College of Engineering
Center for Environmental Research and Technology
University of California
Riverside, CA 92521

Co-Principal Investigators

Arthur M. Winer, Ph.D. and Steven Colome, Sc.D.

Department of Environmental Health Sciences
School of Public Health
University of California
Los Angeles, CA 90095

Participating Researchers

Eduardo Behrentz, Lisa D. Sabin, Seong Jeong Lee, Kenneth Wong, Kathleen Kozawa

Department of Environmental Health Sciences
School of Public Health
University of California
Los Angeles, CA 90095

David Pankratz, Kurt Bumiller, David Gemmill, Matt Smith
College of Engineering
Center for Environmental Research and Technology
University of California
Riverside, CA 92521

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ABSTRACT

To determine the range of children's exposures during their bus commutes, especially those conditions leading to high exposures, real-time and integrated measurements of pollutant concentrations were conducted inside five conventional diesel school buses, as well as a diesel bus outfitted with a particulate trap and a bus powered by natural gas. Measurements were made during 20 bus commutes on a Los Angeles Unified School District bus route from South Central Los Angeles to the west side of LA, with additional runs on a second urban route, a rural/suburban route, and to test the effect of window position.

Children's school bus commutes in Los Angeles appear to expose them to significantly higher concentrations of vehicle-related pollutants than ambient air concentrations and frequently higher concentrations than those measured on roadways. Concentrations of diesel vehicle-related pollutants such as black carbon and particle-bound PAHs were significantly higher on board conventional diesel buses when windows were closed. This was due to the intrusion of the bus's own exhaust, as demonstrated through the use of a tracer gas added to each bus's exhaust. When windows were open, increased ventilation rates markedly reduced this effect, although high peak concentrations were then observed when following other diesel vehicles. On-board concentrations of vehicle-related pollutants were also significantly higher on the urban routes compared to the rural/suburban route, indicating the importance of surrounding traffic density.

Other related exposure scenarios such as bus loading and unloading, and time spent waiting at bus stops, were shown to make relatively insignificant contributions to children's exposure, due to the generally lower concentrations and the short times spent at those activities compared to bus commutes. Results from this study show that minimizing commute times, using the cleanest buses for the longest routes, and reducing bus caravanning and idling time will reduce children's exposure to bus-related pollutants.

1.0 EXECUTIVE SUMMARY

Background. Because children's lungs are still developing and children are more susceptible to adverse health effects from air pollution, potentially high pollutant exposures during school bus commutes are of concern. Studies of pollutant concentrations inside vehicles show high exposures are typical, but few studies have attempted to characterize concentrations on-board and near school buses. This study's main purpose was to determine the range of children's exposures during their bus commutes, with an emphasis on determining the specific factors and conditions leading to high exposures and comparing the effects of different bus and traffic characteristics and bus operating conditions.

Methods Real-time and integrated measurements of pollutant concentrations were conducted in the spring of 2002 while driving school bus routes in Los Angeles (LA) with five conventional diesel school buses, manufactured from 1975 to 1998, a 1998 diesel bus outfitted with a particulate trap, and a 2002 bus powered by natural gas. All diesel buses used low sulfur "green" diesel fuel. Runs included ten morning and ten afternoon bus commutes over an LA Unified School District (LAUSD) bus route from South Central LA to the west side of LA, with four additional runs on a second LAUSD urban route, seven additional runs on a rural/suburban route, and to three additional runs to test the effect of window position.

Results: Bus stop and bus loading/unloading activities were shown to make small contributions to overall commute-related exposures due to the low concentrations found there and the short lengths of time involved in those activities. Exposure factors calculated were as much as two orders of magnitude higher for bus commutes on urban routes than for the bus stop or loading/unloading microenvironments (Table 1.1).

Table 1.1 Average exposure factors (air concentration * time) in three microenvironments.

	Loading/Unloading[1]	Bus Stops[2]	Urban Commutes[2]
Black Carbon ($\mu g/m^3 * min$)	5	20	600
Particle-Bound PAH ($ng/m^3 * min$)	45	230	10000
NO_2 (ppb * min)	105	270	5500
Particle Counts (#/cm3 * min)	25	310	10000
PM2.5 ($\mu g/m^3 * min$)	N/A	130	3500

[1]Based on measurements taken during pilot study. [2]Based on five min. spent at bus stop and a 76 min. commute.

Overall, children's school bus commutes in Los Angeles appear to expose them to significantly higher concentrations of vehicle-related pollutants than ambient air concentrations and frequently higher concentrations than those measured on roadways. Self-pollution from the bus's own exhaust was found to play a significant role in on-board bus concentrations, especially when windows were closed, as was demonstrated by on-board measurements of an inert tracer gas, SF6, added to each bus's exhaust. Older buses showed higher rates of exhaust intrusion, but intrusion was detected in all buses. With closed windows, mean concentrations of diesel vehicle-related pollutants such as black carbon and particle-bound PAHs on board conventional diesel buses were more than double the mean concentrations with windows open. Under closed window conditions, diesel vehicle-related pollutants were also significantly higher on-board the conventional diesel buses as compared to the single CNG-powered bus, while the trap-equipped bus had concentrations between the two (although diesel-related pollutant concentrations on board this specific trap-outfitted bus appeared to be higher than expected based on

emissions data reported for other trap-equipped diesel vehicles). In contrast, natural gas-related pollutants such as formaldehyde were higher aboard the CNG bus. With closed windows, concentrations were also somewhat higher in the rear of the bus, but front/rear differences were generally smaller than the bus-to-bus concentration differences. When windows were open, the resulting high ventilation rates appeared to strongly reduce the amount of self-pollution, while the influence of following individual vehicles became more pronounced, and high transient concentrations of diesel vehicle-related pollutants were associated with proximity to other diesel vehicles.

On-board concentrations were also strongly influenced by other traffic sources. Table 1.2 presents the mean pollutant concentrations by run type for the urban and rural/suburban routes, under open and closed window conditions. Both window position and surrounding traffic density can be seen to have strongly affected on-board concentrations for vehicle-related pollutants, but for pollutants with a strong background component, such as $PM_{2.5}$ and fine particle counts (0.3-0.5 μm), differences between routes and differences due to window position were less distinct.

Table 1.2. Concentrations means and range of means by route and by window position[1].

Pollutant	Urban Route[1]				Rural/Suburban Route	
	Windows Closed (a.m.)[2]		Windows Open (p.m.)[2]		Windows Open (p.m.)[2]	
	Mean	Range of Means	Mean	Range of Means	Mean	Range of Means
Black Carbon (μg/m³)	10	2.5 – 19	5.2	2.9 – 9.1	2.7	0.9 – 4.8
PM-bound PAH (ng/m³)	198	64 – 400	96	33 – 147	36	14 – 66
NO_2 (ppb)	64	34 – 110	73	39 – 120	45	23 – 68
Formaldehyde (μg/m³)	2.1	0.89 – 4.8	1.1	0.55 – 2.1	0.93	0.34 – 2.0
Particle Counts (#/cm³)	113	51 – 235	96	19 – 276	159	29 – 253
Carbon Monoxide (ppm)	5.1	3.7 – 6.4	2.4	1.9 – 2.9	na[3]	na[3]

[1]Includes commutes on urban route one only. [2]Windows closed in morning and open in afternoon, similar to conditions uniformly observed in other buses in Los Angeles for April through June. [3]No measurements conducted.

Conclusions: Measurements made on-board school buses in Los Angeles indicated higher exposures are occurring during children's commutes than ambient air concentrations would indicate. These exposures resulted primarily from the commute itself, and not from loading, unloading, or waiting at bus stops. High commute exposures had several causes: the high concentrations of pollutants already present on roadways, especially if traffic was heavy; the direct influence of other vehicles being followed; and the contribution of the bus's own emissions. The extent of a bus's own contribution to these high concentrations appeared to be higher when windows were closed and for older buses. However bus-to-bus variability was relatively high and therefore with the small number of buses studied, and the limited number of routes covered, the findings of this study should not be viewed as inherently typical for all buses under all conditions. However, minimizing commute times, using the cleanest buses for the longest bus routes, and reducing bus "caravanning" and unnecessary idling time would reduce children's exposures to bus-related pollutants.

2.0 INTRODUCTION AND BACKGROUND

2.1 Introduction

The health of children in California is the focus of intense interest by the Governor, the State Legislature, and local, state and federal health agencies, as well as by academic researchers. Growing evidence indicates children's health may be affected by environmental influences, including air pollutants. Although ambient (outdoor) air pollution may contribute to adverse health effects, the highest exposures to a range of air contaminants may occur in other microenvironments, especially in vehicles. Of particular concern to the California Air Resources Board (ARB) is the time spent by children during school bus commutes. Out of the six million school children in California, one million are transported by public school buses (California Department of Education, 2002). About 70% of the 26,000 school buses operating in California are powered by diesel engines (Horie et al., 1994; Long, 2000), and the ARB recently declared particulate matter in diesel exhaust to be a toxic air contaminant.

A recent study of in-vehicle concentrations, conducted in Sacramento and Los Angeles using a passenger vehicle as a chase car (Rodes et al., 1998), found that proximity to diesel vehicles caused high concentrations of in-vehicle fine particles and black carbon. Moreover, children may be exposed to high concentrations of diesel particulate matter (DPM) and other associated vehicle emissions (e.g. nitrogen dioxide and carbon monoxide) while waiting at school bus stops, riding on buses (particularly where buses are "caravanning"), or during the time they are assembled at school for loading or unloading near buses that are idling.

2.2 Statement of the Problem

Because children's lungs are still developing and because they are more susceptible to the health effects of certain pollutants, there is concern regarding potentially high pollutant exposures during commutes to and from school. However, no comprehensive studies of children's exposure while traveling to and from school have been performed to date in California, even though roadways and sidewalks have been shown to have the highest outdoor concentrations for many air pollutants and in-vehicle concentrations have been shown to be higher than those measured at fixed site monitors and in some cases higher than measured along roadways (Shikiya et al., 1989, Rodes et al. 1998).

Children are especially susceptible to air pollution because of their high inhalation rates relative to body mass, high activity concentrations, greater time spent outdoors, narrower lung airways, immature immune systems and rapid growth (Lipsett, 1989; Pope, 1989; Phillips et al., 1991; Wiley et. al., 1991; U.S. EPA, 1996). In spite of this increased risk, there is a lack of data concerning children's in-vehicle exposure, especially involving diesel school buses. Because of the potential for high pollutant concentrations, school bus cabins, roadways, bus stops, and school yards near idling buses are critical microenvironments that must be included in accurate assessments of children's exposures.

2.3 Background

California Health and Safety Code Section 39660.5 requires the ARB to assess Californians' indoor exposures to toxic air pollutants as part of the ARB Toxic Air Contaminants Program. The ARB is also required to identify the relative contribution of indoor concentrations

to total exposure, taking into account both ambient and enclosed (e.g. indoor and in-vehicle) environments. Senate Bill 25 (Escutia 1999), requires the ARB to identify areas where exposure of infants and children to air pollutants is not adequately measured by the current monitoring network and to conduct enhanced monitoring. Because of the potential for high pollutant concentrations, school bus cabins, roadways, bus stops, and school yards near idling buses are critical microenvironments that must be included in assessments of children's exposures.

It is well understood that gasoline-powered vehicles create significant concentrations of fine particles, carbon monoxide (CO), oxides of nitrogen (NO_x) and volatile organic compounds (VOC), including aromatic VOC. In addition, diesel-powered vehicles emit greater mass per mile of fine particles, NO_x, sulfur compounds, semi-volatile organic compounds (SVOC) and elemental carbon (EC), and may emit higher quantities of polycyclic aromatic hydrocarbons (PAH), some of which are mutagenic and/or carcinogenic. Consistent with this, elevated concentrations of elemental carbon and PAHs have been recorded in traffic tunnels and on heavily traveled roadways (Hering et al., 1984; Benner et al., 1989; Venkataraman et al., 1994a,b; Kirchstetter and Harley 1999).

In traffic, vehicle occupants are primarily exposed to the exhaust of neighboring vehicles, particularly those directly ahead of the occupant's vehicle (Rodes et al., 1998). In-vehicle concentrations of CO and fuel-related VOCs have been found to be significantly higher than those in ambient air (Shikiya et al., 1989; Ptak and Fallon, 1994; Lawryk and Weisel, 1995; Rodes et al., 1998; Jo and Park, 1999, Alm et al., 1999). Of all these in-vehicle studies, only the two (by Shikiya et al., 1989 and Rodes et al., 1998) were performed in California. Other variables that have been shown by previous researchers to be important influences on in-vehicle concentrations include traffic density, driving lane, inter-vehicle spacing, traffic speed, and the number of stops. Secondary factors that may also be significant under certain conditions include wind speed and direction, inverse height, time of day, vehicle size, vehicle age, vehicle ventilation settings, and engine type.

According to a recent Natural Resources Defense Council (NRDC) study (Solomon et. al., 2002), lack of explicit idling policies is common in Los Angeles area district schools. However, the ARB has recently approved an airborne toxic control measure (ATCM) that would limit school bus idling, and idling at or near schools, to only when necessary for safety or operational concerns (ARB, 2002). The ATCM to limit idling is intended to reduce children's exposure to diesel exhaust particulate matter and other Toxic Air Contaminants (TAC) from heavy-duty motor vehicle exhaust.

2.4 Previous In-Vehicle Studies

In traffic, vehicle occupants are primarily exposed to the exhaust of neighboring vehicles, particularly those directly ahead of the occupant's vehicle (Rodes et al., 1998), and in-vehicle concentrations of CO and fuel-related VOCs have been found to be significantly higher than those in ambient air (Shikiya et al., 1989; Ptak and Fallon, 1994; Lawryk and Weisel, 1995; Rodes et al., 1998; Jo and Park, 1999; Alm et al., 1999; Solomon et al., 2001; Wargo et al., 2002), including in the three studies performed in California (Shikiya et al., 1989; Rodes et al., 1998; Solomon et al., 2001).

Children typically spend over an hour a day in transit (Phillips et al., 1991) and, as noted, in-transit microenvironments can exhibit high concentrations of pollutants. For example, for aromatic VOCs, such as benzene, the in-transit microenvironment is estimated to contribute 10 to 60 percent of a nonsmoker's total exposure (Chan et al., 1991a,b; Weisel et al., 1992; Lawryk and Weisel, 1995; Fruin et al., 2001).

In the Rodes et al. (1998) study, for most pollutants 2-hr integrated samples were collected inside and outside a passenger car, along the roadway where the vehicle traveled, and at ambient monitoring sites. Pollutants measured included PM_{10} and $PM_{2.5}$, metals, and thirteen VOCs. Continuous measurements were made of fine particles, CO and black carbon (BC). The driving scenarios were chosen to evaluate in-vehicle pollutant concentrations as a function of traffic congestion, vehicle type, roadway type, time of day, and ventilation setting.

Rodes et al. (1998) found in-vehicle pollutant concentrations were generally significantly higher in Los Angeles than in Sacramento, supporting the choice of Los Angeles for the present school bus study, which sought in part to identify high-end exposures. In general, VOC and CO concentrations inside or just outside the vehicles were four to ten times higher than those measured at the roadside or at ambient air stations (Rodes et al., 1998). Interestingly, however, in-vehicle $PM_{2.5}$ concentrations were consistently lower than $PM_{2.5}$ concentrations just outside the vehicles, and in some cases were also lower than roadside concentrations. $PM_{2.5}$ concentrations inside or just outside the vehicles, however, were usually higher than concentrations measured at the nearest ambient site, however.

Also interesting was the finding by these investigators that under their study conditions, factors such as ventilation settings and vehicle type had little effect on in-vehicle pollutant concentrations. In contrast, factors such as driving lane (e.g. carpool lane versus right lane), roadway type, congestion level, and time of day all influenced in-vehicle concentrations. For example, substantially higher pollutant concentrations were measured in the far right hand lane compared to the carpool lane and when following high-emitting "lead vehicles."

Further analysis of the Rodes et al. (1998) data was conducted by Fruin (2003). To determine which parameters had the most significant impacts on in-vehicle DPM concentrations, video tapes recorded during the driving commutes were used to create more refined groupings of the vehicles being followed, (e.g. 2, 3 or 5-axle diesel vehicles and alternate-fuel buses). Other factors such as exhaust location and following distance of the vehicle ahead were also included in the analysis. Results of these analyses confirmed the Rodes et. al. (1998) conclusion that following a diesel-powered vehicle was associated with a significant increase in both black carbon and fine particle counts. In addition, Fruin (2003) found that exhaust location, following distance, and sometimes road type also influenced in-vehicle concentrations, while following a gasoline-powered passenger vehicle or alternate fuel bus had weak or no associations with in-vehicle DPM concentrations.

An earlier in-vehicle study conducted in California was the pioneering study by Shikiya et al. (1989). They demonstrated that in-vehicle concentrations of emitted criteria pollutants such as CO and NO_x could be two to four times those measured at fixed site monitors. Of the

remaining studies discussed in the Introduction given above, only those by Ptak and Fallon (1994), Alm et al. (1999), and Solomon et al. (2001) measured particulate matter.

In the recent study by Solomon et al. (2001), a team of researchers from the Natural Resources Defense Council and the University of California, Berkeley, investigated the concentrations of diesel exhaust constituents inside school buses in the Los Angeles area. They employed continuous measurements of $PM_{2.5}$ and black carbon concentrations inside four school buses. For comparison, they also included measurements outside the buses and in a passenger car traveling ahead of the buses. They found the level of black carbon was higher in the back of the buses compared to the front. Furthermore, the level of black carbon increased when all windows were closed, while it decreased with windows open. However, the level of black carbon in the cabin did not change, or decreased, during idling. Solomon et al. (2001) concluded the level of black carbon in the back of a school bus with windows closed could be up to four times higher than in a passenger car ahead of the bus.

In a recent school bus study in Connecticut, Wargo et al. (2002) found concentrations of black carbon and $PM_{2.5}$ inside commuting school buses were often 5-10 times higher than rural background concentrations. They found several important variables affected the concentrations of pollutants inside school buses, including bus ventilation via windows, bus idling behavior, and outdoor concentrations on bus routes. They observed the concentrations of black carbon and $PM_{2.5}$ in moving buses were higher when windows were closed compared to when windows were open. However, mean concentrations of both black carbon and particulates were higher in idling buses when windows were open. Furthermore, idling buses had higher concentrations of particles and black carbon than moving buses. They also found concentrations increased when traveling behind other buses including diesel buses (e.g., caravanning). On the other hand, they found black carbon and particulate concentrations did not vary by sampling location within their buses (i.e., front versus rear). They also employed school buses fueled with natural gas in their study and observed CNG buses emitted 60–98% less black carbon than diesel-powered buses.

Wargo et al. (2002) provided several recommendations for reducing children's exposure to diesel exhaust particulate: prohibit bus idling; retrofit with particle traps and catalytic converters; use ultra-low sulfur fuels; allocate the cleanest buses to the longest route; and limit ride duration. Solomon et al. (2001) also addressed similar recommendations, including retrofitting with particle traps, purchasing alternative fuel school buses (e.g., CNG), and keeping windows open on school buses. The South Coast Air Quality Management District (AQMD) has approved $10.4 million to install particulate trap filters on nearly 1,500 diesel-powered school buses in the summer of 2002 (AQMD, 2002).

Of the peer-reviewed articles we found concerning exposures to pollutants in buses where human subjects were involved, only two specifically concerned school children. Wu et al. (1998) reported the use of an iridium tracer to determine soot exposure of high school students commuting to and from school on public diesel transit buses in Baltimore. A portion of the Baltimore municipal fuel supply was tagged with an iridium tracer and exposure during commutes was monitored with personal aerosol monitors. Twenty samples were collected over 10 days while four students commuted on regularly scheduled transit buses and a fifth student commuted by private car. Exposures were greatest for students commuting through the

congested central business district of Baltimore. The tracer was undetectable in samples collected by the student commuting by car when the windows were closed, but comparable to the other students (on transit buses) when the car windows were open during the commute.

Wargo et al (2002) outfitted children with personal monitors that measured PM_{10}, $PM_{2.5}$, and VOCs. Measurements were made throughout the school day, including time spent traveling to and from school on diesel school buses. Average personal concentrations of $PM_{2.5}$ were nearly three times higher than average concentrations for outdoor air in that community, while concentrations measured while the children were riding inside school buses were up to ten times greater than background.

2.4.1 Related Traffic Exposure Studies

Adams et al. (2001) studied determinants of personal exposure to $PM_{2.5}$ in transport microenvironments in London, UK. Four transport modes, bicycle, bus, car and underground rail, were involved. Meteorological variables (wind speed, wind direction, precipitation, temperature, atmospheric pressure and relative humidity), traffic density and route were also considered. Except for the underground rail, they found wind speed and route were significant factors determining personal exposure to $PM_{2.5}$, while transport mode was not significant.

Chan and Wu (1993) studied bus commuter and pedestrian exposure to traffic pollution in Hong Kong. Kinney et al. (2000) measured concentrations of $PM_{2.5}$ and elemental carbon on sidewalks in Harlem, New York City. They selected four monitoring sites ranging from heavily traveled roadways to a quiet residential sidewalk. $PM_{2.5}$ concentration showed little association with the proximity to local diesel traffic, while elemental carbon concentration exhibited a strong spatial gradient across sites, consistent with recent studies by Zhu et al. (2002 a, b). Average elemental carbon concentration ranged from 1.5 ug/m^3 to 6.2 ug/m^3 (a four-fold difference between two sites with the largest contrast in diesel traffic counts).

Gee and Raper (1999) investigated commuter exposure to respirable particles inside buses in Manchester, England. This study involved using personal sampling pumps installed in the bus cabs to obtain an estimate of the average commuter exposure to fine particles during congested commutes. Conceicao et al. (1997) installed a "removal" duct in a commuter bus to improve ventilation rate and modeled the airflow with a simple, uni-dimensional flow model, predicting the air exchange rate as a function of the vehicle velocity. Tracer gas experiments were performed to test the model and the efficacy of the air removal duct. Scheirl and Fruhmann (1996) measured airborne platinum concentrations in city buses in Munich, Germany.

Several studies in Denmark measured relationships between DNA adducts found in urine samples from Copenhagen bus drivers and pollutant concentrations in their buses. Loft et al. (1999) found increased urinary excretion of a biomarker of oxidative DNA damage in urban bus drivers. Autrup et al. (1999) and Nielson et al. (1996) used selected biomarkers in nonsmoking Danish bus drivers and postal workers and found significantly higher concentrations of carcinogen-DNA adducts in bus drivers working in the central part of Copenhagen compared with rural bus drivers in a control group.

Soll-Johanning et al. (1998) conducted a retrospective cohort study of cancer incidence in urban bus drivers in Copenhagen covering the period 1900 to 1994 and found an increased risk of developing several types of cancers, although risk factors other than exposure to air pollutants during working hours (e.g., smoking) could not be ruled out as causal factors. Fernandez-Bremauntz and Ashmore (1995a) studied the exposure of commuters to carbon monoxide in a range of vehicles, including transit buses, and in general found much higher in-vehicle CO concentrations in Mexico City than those reported for previous studies in the United States. This study also reported mean in-vehicle CO concentrations were three times higher than fixed-site monitoring concentrations.

2.5 Objectives

2.5.1 Overall Objectives

The overall objective of this study was to characterize the range of exposures experienced by children during their school bus commutes, especially in potentially high exposure conditions. By identifying factors that lead to higher exposure, this study may provide guidance for minimizing children's exposures. The results may also facilitate evaluations of the direct health benefits of alternative fuel types and improved bus emission control technologies.

2.5.2 Specific Objectives

The over-arching specific objective of this project was to obtain measurements of in-bus and near-bus pollutant concentrations during normal school bus operations across the full range of anticipated conditions, but with special emphasis on obtaining measurements during operations expected to lead to realistic high-end exposures. Diesel buses were of highest priority because they comprise the majority of buses used by California schools.

2.5.2.1 Pilot Study

The specific objectives of the pilot study were to test and verify the feasibility and utility of the study design and proposed sampling methods prior to the main study, and to gain experience in all facets of conducting the experiments. Specifically, the pilot study gave us experience in deploying an extensive array of instruments within the demanding confines of a bus and a van, and allowed us to evaluate how well the various instruments performed on a moving platform, as well as the utility of the collected data. Another objective of the pilot study was to evaluate the relative importance of three microenvironments, commutes, bus stops and loading/unloading zones to children's total school bus related exposures.

2.5.2.2 Main Study

The specific objectives of the main study were to obtain measurements that can be used (in combination with the pilot study data) to address the following questions:

1. What are children's pollutant exposures during bus transit, especially for those who spend a large portion of their transit time in high-concentration conditions?
2. How do different bus fuel types (i.e. diesel - and natural gas-powered) affect children's exposure?
3. What are the most important factors governing children's exposure associated with commuting on school buses?

3.0 PILOT STUDY FINDINGS AND RECOMMENDATIONS

3.1 Introduction

The principal purpose of the pilot study was to prove methods and refine protocols. Thus, the pilot study was conducted to verify the feasibility of the study design and the utility of the instruments and sampling methods on a mobile platform. In addition, the pilot study was designed to provide range-finding measurements of conditions influencing high exposures to mobile source air pollutants in the context of school bus-related microenvironments. Three locations identified as potentially high exposure scenarios were explored in the pilot study with an emphasis on identification of conditions leading to highest exposures. These included exposures while waiting for the bus at a bus stop near home, while commuting on the bus in traffic, and/or while waiting at the school bus loading zone for the return ride home. Another objective was to evaluate the importance of these three microenvironments to children's total school bus-related exposures.

In the pilot study, we evaluated a single diesel school bus traveling along a single bus route. A suite of real-time and integrated instruments were used to measure concentrations of gaseous pollutants and fine particles inside the cabin of the bus while traveling along a designated route, at a selected bus stop, and in front of a school in the bus loading/unloading zone. The importance of variables such as location inside the bus cabin, differences between concentrations inside and immediately outside the bus, and the effect of window position were also evaluated.

The highest exposures were expected in areas with heavy traffic congestion and during periods of meteorological stagnation. For this reason, the pilot study was conducted in the service area of the Los Angeles Unified School District(LAUSD) in November and December 2001. We used contacts within LAUSD to identify bus-commuting routes and to gain access to detailed routes and stops. We replicated the characteristic patterns of loading/unloading, bus stops and commutes observed for a specific bus route for an appropriate school, with emphasis on conditions where adjacent traffic was greatest, bus idle time was longest, and where children were waiting for substantial periods of time to be picked up.

3.2 Summary of Pilot Study Findings

3.2.1 Exposure in Three School Bus Commute Microenvironments

As discussed extensively in the Pilot Study Final Report (Fitz et al., 2002), three microenvironments were investigated concerning children's pollutant exposure due to school bus-related activities. "Bus commutes" refers to measurements made using the bus as a platform for the various instruments. For this type of run a typical route transporting children from the inner city to BSMS (and vice versa) was followed. "Bus stops" refers to sampling at one of the stops along the selected route. Generally, bus stops may be near children’s residences or often, for reasons of safety and efficiency, at a nearby school. An instrumented van was parked in front of Vermont Elementary School at the corner of Vermont Avenue and 97th Street, where diesel buses arrived to pick up students traveling to other schools (morning) or to drop off student coming from other schools (afternoon). In addition, a large number of parents dropped off or picked up children at this "bus stop” using their personal vehicles.

"Loading/unloading" runs involved measurements with a sampling van parked next to the sidewalk of BSMS, in a portion of the staff parking lot, about five feet from the curb and near the location where children congregated briefly when leaving or boarding the buses. Typically each morning and afternoon, nineteen school buses lined up along this sidewalk to unload or load children, respectively.

After analyzing the pollutant concentrations measured in the pilot study, as well as the time spent by children in these three microenvironments, we concluded the relative importance of commutes versus stops versus loading/unloading zones was quite different. To better understand and quantify these differences, we estimated pollutant-specific exposure factors for the three microenvironments, defined as the product of the average concentration for a specific pollutant times the time spent by children in the particular microenvironment. Finally, we calculated the average ratios of exposure factors for each pair of microenvironments for each pollutant. The average over all pollutants was taken to be a reasonably quantitative measure of how important one microenvironment was in relation to the other two.

Table 3.2.1 summarizes the results for the calculation of exposure factors, exposure ratios, and exposure ratios averaged for the set of analyzed pollutants. Note the estimated average exposure ratio for bus commutes versus bus stops was 35. This indicates the contribution of the bus stop was about 3% of that observed for bus commutes. Similarly, the estimated average exposure ratio for bus commutes versu the loading/unloading zone was 90. Thus, the exposure contribution of the loading/unloading zone was only approximately 1% of that observed for bus commutes.

From these results we concluded the loading/unloading zone microenvironment was not as important in terms of exposure as the other two microenvironments. There are two principal reasons for this. First, we observed that children generally spend a very short time on the sidewalk (typically as little as a minute, but on occasion as much as five minutes) before boarding the buses in the afternoon, or before entering the school in the morning. Second, bus drivers were required by school district policy to turn off their engines as soon as they arrived in the morning, before the children leave the bus. Similarly, in the afternoon, drivers were instructed to not turn on their engines before all children were aboard the buses and the entire fleet was prepared to depart. During the pilot study we observed nearly 100% compliance with these regulations (although we have anecdotal information from a BSMS teacher that this policy was not always rigorously enforced--J. Fogel, private communication. 2002).

3.2.2 Conditions Leading to Highest Exposures Inside the Bus Cabin

During the pilot study, conditions associated with the highest pollutant concentrations inside the bus were proximity to another diesel vehicle, idling (either at bus stops or traffic lights, or due to traffic congestion) and closed windows. Also, the difference between pollutant concentrations inside the bus while traveling on the freeway versus surface streets was not found to be significant during the pilot study. However, this may have been due to the experimental design, which involved alternating between open and closed windows during the course of each run. Windows were typically more often open on the freeway portion of the run and more often closed while traveling on surface streets. Because opening windows was found to reduce

Table 3.2.1 Calculated exposure factors for three school bus commute-related microenvironments.

	Approximate Average Concentration			Exposure Factor[2,3]				
	Bus Stop	Loading/Unloading	Bus Routes	Bus Stop	Loading/Unloading	Bus Routes	Exposure Ratio A[4]	Exposure Ratio B[5]
BC (mg/m^3)	4	2	14	20	10	1400	70	140
PAH (ug/m^3)	65	15	100	325	75	10000	31	133
NO_2 (ppb)	45	35	55	225	175	5500	24	31
VOC (ppb)	120	70	175	600	350	17500	29	50
CO (ppm)	3	1	3	15	5	300	20	60
Particle Counts ($\#/cm^3$)[1]	30	8	43	150	40	4300	29	108
PM-2.5 (ug/m^3)	35	13	56	175	65	5600	32	86
						Average Exposure Ratio:	35	90

[1]Between 0.3 and 0.5 um diameter

[2]Defined as the product of average concentration and time, in minutes, spent in that particular microenvironment

[3]Average time spent on bus runs was 100 minutes. Upper limit to time spent at bus stops was 5 minutes (approximate values)

[4]Defined as the ratio of exposure factor for bus routes to exposure factor for bus stops

[5]Defined as the ratio of exposure factor for bus routes to exposure factor for loading/unloading zones

concentrations inside the bus, differences in pollutant concentrations while traveling on freeways versus surface streets may have been obscured during the pilot study.

3.3 Modifications of Experimental Design for Main Study

3.3.1 Relative Importance of School Bus Commute Microenvironments

As discussed above, in the pilot study, concentrations observed during bus commutes were significantly higher than those at bus stops or at the loading/unloading zone. In addition, we observed the time children spent in these microenvironments was substantially different, typically a few minutes at the bus stop or loading/unloading zone versus up to 100 minutes for bus commutes each way. We concluded bus commutes were the most important microenvironment in terms of exposure, and recommended the main study focus primarily on that microenvironment. However, while we found no justification for expending additional resources in further characterization of loading/unloading zones, we determined a limited number of additional measurements at an additional bus stop during the main study would allow us to better define potential exposures associated with this microenvironment since sporadic and short-lived high concentrations were sometimes observed due to the arrival and departure of other buses.

Specifically, although we found the bus stop contribution to exposure was minor compared with the bus runs (about 3%), additional runs were recommended in part because a fairly high emitting bus was used in the pilot study. A less polluting bus may exhibit lower average exposure ratios, leading to a greater contribution from bus stops to total exposure.

A second reason for a limited number of additional bus stop runs was that in the pilot study we observed the highest values for fine particle counts in the 0.18 um size range during the bus stop measurements. This was due in part to the use of the Vermont Avenue School as a bus stop by a number of schools other than BSMS. We observed in the neighborhoods in which our pilot study took place almost all the bus stops have been placed at schools (presumably in part out of safety considerations for the children). Thus, a substantial number of buses may stop to pick up students, destined for a variety of different schools. For example, during our monitoring periods at the Vermont Avenue School bus stop at various times as many as half a dozen buses would pull up in front of the school (adjacent to our instrumented van) and wait with the engine idling until children boarded. In some cases buses were early and waited with their engines idling for several minutes, while students waiting for other buses stood nearby. Buses would then accelerate away from the curb, often releasing an exhaust cloud of black smoke. Several of the highest fine particle counts we observed occurred at such times. Similarly, parents driving children to and from the Vermont Elementary School, in some cases in "gross emitter" vehicles (as documented in our field-notebooks), contributed to the sporadic occurrence of high particle counts.

3.3.2 Precision Data

Paired instrument data were important in determining the reliability, and precision of the deployed instruments. The lack of adequate precision estimates for a number of the instruments from the pilot study data (due to insufficient pollutant concentrations) made it difficult in some cases to interpret differences observed in pollutant concentrations between the front and back of the bus, and between inside and outside the bus, with a high level of confidence. The relatively

unpolluted background conditions in West Los Angeles, where all but the NO_2 precision data were collected, often resulted in measured pollutant concentrations that were not sufficiently above the detection limit. One recommendation for the main study was to perform precision measurements in a location where background concentrations for many of these pollutants would be expected to be sufficiently higher than the detection limit. In addition, the importance of collecting sufficient precision data before and after runs to establish comparability of measurements from pairs of instruments was emphasized.

3.3.3 Gradient Tests

Based on analysis of the pilot study data, we recommended simultaneous measurements inside and immediately outside the bus continue during the main study. The results from the pilot study indicated children were exposed to higher pollutant concentrations on the bus compared with outside, emphasizing the importance of the school bus microenvironment to children's total exposure.

Although the pilot study found few differences between concentrations at the front and rear of the bus cabin, our recommendation for the main study was to continue front and rear sampling in a limited number of cleaner buses to better determine if there was a concentration gradient between the front and rear of school bus cabins across different bus and fuel types.

3.3.4 Conditions Leading to Highest Concentrations Inside the Bus Cabin

The data from the pilot study concerning window position were complex and inconclusive regarding the importance of this variable to children's exposure. We recommended further assessment of this variable in the main study to obtain a more robust data set.

In addition, although the influence of the bus's own exhaust was not directly measured during the pilot study, the importance of this variable in estimating children's exposure during bus commutes was potentially high. For the main study, we recommended the use of an SF_6 tracer added to the bus exhaust as a means to evaluate the importance of a bus' own exhaust in children's exposure during bus commutes. We also recommended a snap and idle test for each bus for the same reason.

3.3.5 Routes

Based on the pilot study results, we elected to use the same urban route for the majority of commute runs, except that the route began and ended at the fifth bus stop. This shortened the route to about one hour duration, which was judged optimum to meet the pollutant sampling objectives of the main study while maintaining the diversity of roadway types and congestion scenarios.

3.3.6 Summary of Pilot Study Findings

The pilot study demonstrated the proposed instrumentation and measurements were feasible on a moving platform such as a school bus and that the proposed routes provided both high concentration conditions as well as a wide range of concentrations. However, locations of higher concentrations for the collocated instrument comparisons were needed to ensure adequate signal-to-noise conditions for direct instrument comparisons. Bus stop and bus loading/ unloading activities were demonstrated to make small contributions to overall commute

exposures, so further characterizing these particular activities was de-emphasized in the main study to allow greater resources to be devoted to characterizing the actual drive portion of the commutes and the differences between buses.

4.0 EXPERIMENTAL METHODS AND STUDY DESIGN

4.1 Introduction

In this study, we obtained measurements that were used to determine children's pollutant exposures during bus transit, especially for those who spend a large portion of their transit time in high-concentration conditions. We measured a wide range of gaseous and particulate matter species inside school buses while traveling along in-use school bus routes. The great majority of these measurements were made using sophisticated real-time instruments, and successfully operating these instruments on mobile platforms was a major experimental challenge.

A total of 36 runs were completed, including six runs on a simulated rural/suburban route, 20 runs on an in-use urban route, four runs on a second in-use urban route, three window position tests, and two measurement periods at a bus stop along the primary urban route. Seven different buses were used, including three high emitting diesel buses, two typical diesel buses, one particle trap-outfitted diesel bus, and one CNG bus. The data collected were used to determine the most important factors governing children's exposure associated with commuting on school buses. All data collected during this study will be submitted to the ARB for future additional analyses.

4.1.1 Characterization and Justification for the Selected Buses

The primary criterion for selecting buses for testing was that they be representative of the school bus fleet used in California with regard to age, mileage, and manufacturer, and include several older, dirtier buses for testing worst-case conditions. Toward this goal, we included the testing of three buses representative of high exhaust emissions. To evaluate state-of-the-art control technology, we evaluated exposures on-board a newer-model bus equipped with a particle trap catalyst. Finally, in order to compare children's exposure during bus commutes for different bus fuel types, we included a CNG-fueled bus. A decision was also made to use buses that were in current use transporting children to school. However, the extensive suite of instruments employed in this study, the batteries needed to power them, and the researchers needed to operate them occupied almost the entire working and seating area inside the bus. Because of this, and for safety reasons, no children were aboard any of the buses in this study. We note the batteries and equipment served as approximate proxies for the children's weight corresponding to a fully occupied bus.

Figure 4.1.1.1 shows the distribution by model year of school buses in California. These data are from the California Department of Motor Vehicles 1998 vehicle registration database. From this table, we see most buses in use were 1985 model year or newer. Table 4.1.1.1 gives the distribution by bus make and use in 1998 in southern California (Los Angeles, Riverside, Orange and San Bernardino counties). Crown buses have not been sold since approximately 1994, and International made chasses for Thomas. Based on informal surveys with bus operators, Thomas appeared to be the dominant seller in southern California. Except for a few older gasoline buses, almost all school buses in the 1998 fleet had diesel-powered engines.

Selection of buses also depended on the difficulty in securing them. The major lease operators were unwilling to cooperate with us for this study, due to a perceived risk of adverse

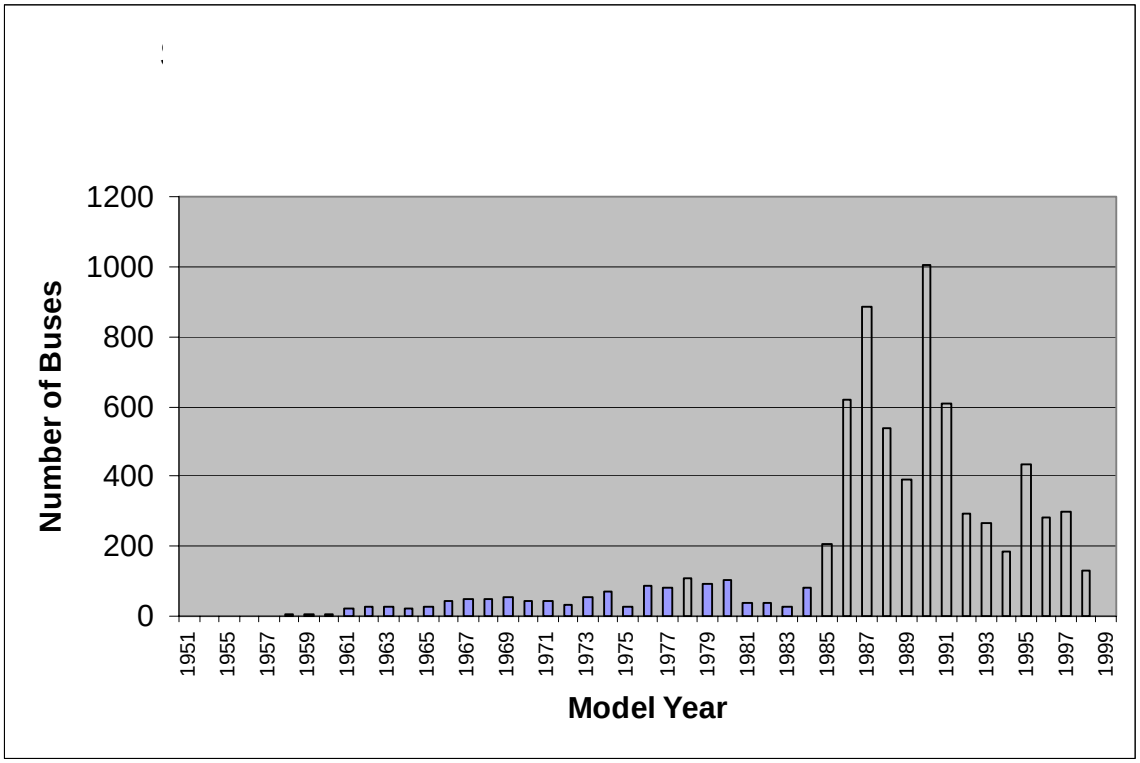


Figure 4.1.1.1 California statewide distribution of school buses by model year.

Table 4.1.1.1 Distribution of buses in southern California by manufacturer.

Make	Number	% Total
Bluebird	1103	14.9
Carpenter	93	1.3
Collins	937	12.6
CROWN	1416	19.1
Gillig	576	7.8
Goshen	5	0.1
International	1731	23.3
Superior	3	0
Thomas	1464	19.7
Ward	26	0.4
Wayne	60	0.8
Total	**7414**	**100**

publicity based on the experience with a study by the Natural Resources Defense Council, which generated negative publicity concerning diesel school bus pollution (Solomon, 2001). These leasing companies also tended to have fleets composed primarily of relatively new buses. Because of these issues, we felt more appropriate sources for school buses were individual school districts. However, we found that many districts were also unwilling to participate for the same reason as the leasing companies (i.e., perceived risk of negative publicity).

Despite these problems, we were able to find two local districts willing to participate in our study by providing buses from their in-use fleet of approximately 150 buses. Although the number of buses was limited, these districts had a wide variety of buses in terms of age and make, and had a small number of CNG buses as well. A final selection constraint was that the buses be large enough to carry all of the test equipment. Most of the larger buses were 10-14 rows, which was sufficient. The smaller buses tended to be van-based with 3-4 rows. These were too small for our equipment and not representative since school districts tend to have only a small number of these for special purposes in any case.

Table 4.1.1.2 lists the buses selected for testing. As noted earlier, the primary criterion for selecting buses for testing was that they be representative of the school bus fleet used in California. Two buses were selected to meet this criterion. The first, representative diesel school bus number one (RE1) was a 1998 Thomas Saf-T-Liner, while the second, representative diesel school bus number two (RE2) was a 1993 Thomas Saf-T-Liner. The three buses selected to be representative of high exhaust emissions were expected to have relatively high emissions based on their age and in the opinion of bus service personnel. These buses included high emitter diesel school bus number one (HE1), a 1985 Thomas Coach; high emitter diesel school bus number two (HE2), a 1985 Crown Supercoach, and high emitter diesel school bus number three (HE3), a 1975 Crown Supercoach. This last bus also satisfied ARB's interest in testing a pre-1977 bus. Throughout this report, we refer to the two representative diesel buses (RE1 and RE2) and the three high emitter diesel buses (HE1, HE2 and HE3) as "conventional" diesel buses. We are using this term to refer to buses powered by typical diesel engines, which have not been modified in any way to reduce emissions beyond standard practices and as required by current regulations. We are not using the term "conventional" in the sense of meaning the engine is in the front of the bus, although this designation is an industry practice.

To test a diesel bus equipped with a particle trap catalyst, we selected a 1998 Thomas Saf-T-Liner equipped with a Johnson Matthey Continuously Regenerating Technology (CRT®) particulate filter (trap-outfitted diesel school bus number one, TO1). The particle trap combined a platinum catalyst and a filter element to "trap" particulate matter, and operated as a passive emissions control system. Finally, in order to compare children's exposure during bus commutes for a different bus fuel type, we tested a 2002 Thomas Saf-T-Liner Compressed Natural Gas school bus (CNG). Photograph 1 shows two of the buses employed in the main study.

Snap and idle opacity tests were performed on all buses. In this test the opacity of the exhaust was measured while the engine was brought from idle to its governed maximum rpm by immediately applying full throttle. The monitoring device recorded the maximum opacity. This test was performed a minimum of three consecutive times and the results were within 5% for a valid test. These results are also shown in Table 4.1.1.2. As expected, the 1985 Crown Supercoach (HE2), which was identified by maintenance personnel as the "smokiest" in their fleet, clearly gave the highest value. It should be noted that it was difficult to conduct a snap and idle test for the CNG bus because the relatively high fraction of water in the exhaust compared to a diesel bus resulted in fogging of the optics in the test device, and inconsistent, but low, measurements resulted.

Table 4.1.1.2 School buses selected for testing.

Bus Number	Bus Type	Year	Make	Model	Rows	Engine	Mileage	Displacement	Snap/Idle	Cycles
854	HE1	1985	Thomas	Coach	14	Cat 3208	293000	10.4 L.	8%	4
851	HE2	1985	Crown	Supercoach	15	Detroit Diesel 671	315000	6 L.	57%	2
752	HE3	1975	Crown	Supercoach	15	Cummins 290	316000	6 L.	18%	2
986	RE1	1998	Thomas	Saf-T-Liner	14	Cummins 250 HP 8.3	111000	8.3 L.	2%	4
921	RE2	1993	Thomas	Saf-T-Liner	13	Cat 3116	177000	6.6 L.	11%	4
982	TO1	1998	Thomas	Saf-T-Liner	14	Cummins 250 HP 8.3	78000	8.3 L.	1%	4
8	CNG	2002	Thomas	Saf-T-Liner	14	John Deere 8.1	1000	8.1 L.	N/A[1]	4

[1]Snap and idle tests not possible due to too much water vapor.
HE1 – HE3: High emitter conventional diesel school buses.
RE1 – RE2: Representative conventional diesel school buses.
TO1: Particle-Trap outfitted diesel school bus.
CNG: Compressed natural gas school bus.

(a) (b)

Photograph 1. a) High emitter conventional diesel test bus parked at UCLA
b) CNG Bus parked in front of a bus stop during an afternoon run

4.1.1.1 Fuel Used in the Test Buses

The fuel used in all diesel buses we tested was Arco Emission Control Diesel (ECD-1). This fuel, also sometimes called "green" diesel, has ultra-low sulfur content (<15ppm), low aromatics, and a high cetane number. Ultra-low sulfur fuel must be used for after-treatment emissions control technologies (e.g. particle trap catalyst) to function properly. However, we used this "green" diesel on all the diesel buses we tested, with or without a particle trap catalyst.

4.1.2 Field Sampling Procedures

In each bus tested, two identical sets of instruments used to measure a variety of gaseous and PM species were mounted inside the bus cabin. One set was used to measure concentrations in the rear of the bus cabin, while the second set was used to measure concentrations either at the front of the bus cabin, or just outside the bus. Measurements were taken while the bus traveled along one of three designated school bus routes (described in Section 4.1.6). Each bus run lasted approximately one hour. A summary of the measurement methods used for this study is presented in Table 4.1.2.1. The instrumentation included both real-time and integrated measurements. Although two sets of instruments were in each bus, the majority of the analyses in this report (including the comparison between bus types) were done using the data from a single set of instruments, located at the rear of the bus cabin.

4.1.2.1 Instrument Packaging and Power Supply

Figure 4.1.2.1 presents the plumbing diagram for the sampling equipment inside each bus. The instruments were mounted onto three plywood platform with elastic cords. In order to minimize the effects of vibration, foam pads were placed between the instruments and the plywood. Instruments that sampled from the front of the bus, or outside the bus, depending on the run, were mounted on one platform and instruments that sampled from the rear of the bus were placed on a second platform. The data logging portable computer (PC), gaseous analyzer calibration

Table 4.1.2.1 Measurement parameters and methods used in the main study.

Code	Species/Parameter	Instrument	Detection Limit	Flow Rate, L/min	Amps 110v
a	$PM_{2.5}$ Integrated Mass	Custom/Harvard Impactor	5 $\mu g/m^3$**	20.0	*
b	$PM_{2.5}$ Real-Time Mass	TSI Dust Trak	1 $\mu g/m^3$	1.7	0.1
c	PM_{10} Integrated Mass	Custom/Harvard Impactor	5 $\mu g/m^3$**	20.0	*
d	PM_{10} Real-Time Mass	TSI Dust Trak	1 $\mu g/m^3$	1.7	0.1
e	Fine Particle Counts	Climet Spectro .3 OPC	Single particle	5.0	0.1
f	Ultra-Fine Particle Counts	CE-CERT SEMS	Single particle	5.5	5
g	Elemental Carbon	Magee Aethalometer	1 $\mu g/m^3$	5.0	0.1
i	Gaseous HC (Total and Speciated)	Gas Chromatographic Methods	Varies	Varies	*
j	CO	Langan electrochemical cell Dasibi 3003	2 ppm 1 ppm	Passive 2 lpm	Internal Battery 1.0
k	NO_2	CE-CERT NO_2/PAN Luminal GC TEI 42 NO/NO_2/NO_x	1 ppb 1 ppb	--- ---	1.0 2.0
m	Aldehydes & Ketones	DNPH collection/HPLC analysis	1 ppb	0.8	*
n	Particle PAH	EcoChem PAS 2000	3 ng/m^3	4	0.25
o	Temp, RH (in-bus)	Rotronics MP101A	2.0°C, 5%	NA	0.1
p	Location and Speed	Garmin GPS Map76	2-5 m	NA	0.1
q	SF_6	AeroVironment CTA 1000	0.010 ppb	0.1	2
r	Traffic Documentation	Sony CXC-390 Video Camera	NA	NA	Internal Battery

* Supplied by common vacuum pump (10 amps) ** 1 hour sampling time

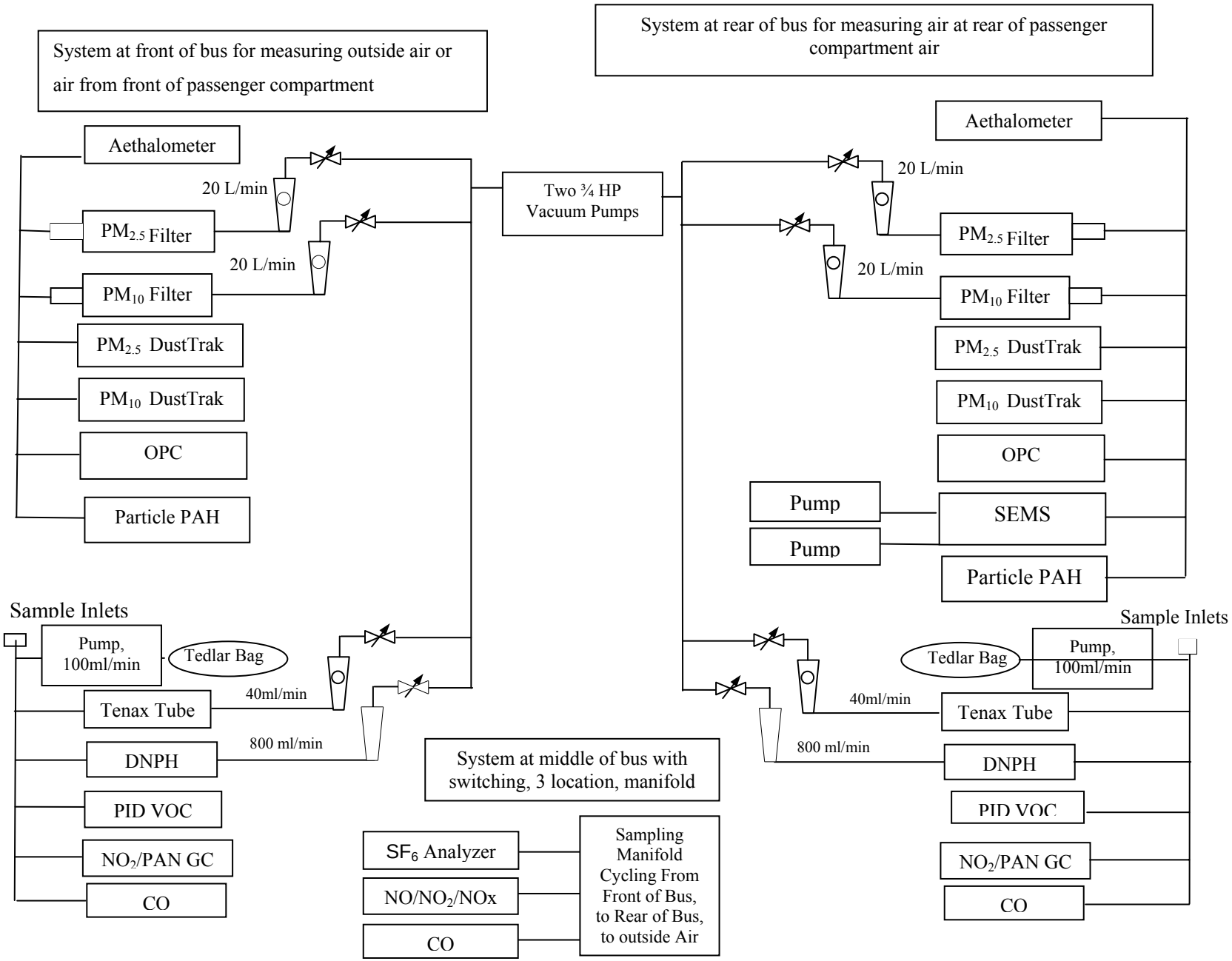


Figure 4.1.2.1 Plumbing schematic for school bus sampling instrumentation.

system and the instruments that were part of the "switching system" that monitored from the front, back and outside were mounted onto a third platform in the middle of the bus cabin. In this way the platforms were easily installed in the buses. The platforms were installed over the seat backs of the buses and tied down with ratcheting straps. All three platforms were mounted along the right side of the bus (facing forwards). The total flow rate for all samplers and the bypass for the isokinetic probe on the buses was about 150 L/min.

A trailer mounted North Star 13000 PPG propane-powered generator with an oxidation catalyst was used to supply AC power for the instruments during the first week of the main study. A hitch was installed at the back of the bus for the generator trailer. The decision was made in the main study to convert to the use of on-board batteries and invertors for all subsequent buses (Runs 4-36) in order to eliminate the noise, complexities and potential pollution from towing a generator behind the bus.

As discussed in more detail in Section 5.3, because of various difficulties and confounders encountered during the runs with the propane generator, the data for bus HE1 were not included in the majority of comparisons between buses.

The onboard inverter and battery system was a 7kVA (kilovolt Amp) Best Power uninterruptible power supply (UPS), but used only for its "inverter" capabilities on this program. Thirty-six deep cycle marine batteries were used for inverter input. Thirty-two of the batteries powered the UPS directly while the remaining four batteries were used to power four DC-powered vacuum pumps. The 200 Kg UPS was placed on the floor of the bus next to a side emergency exit. (One row of seats needed to be removed from most buses to make room for the UPS.) Nine wooden crates with four batteries each were placed in nine seats along the left side of the bus. Conventional "ground power" was used to power the instrumentation onboard the buses when the buses were "parked" at UCLA and CE-CERT.

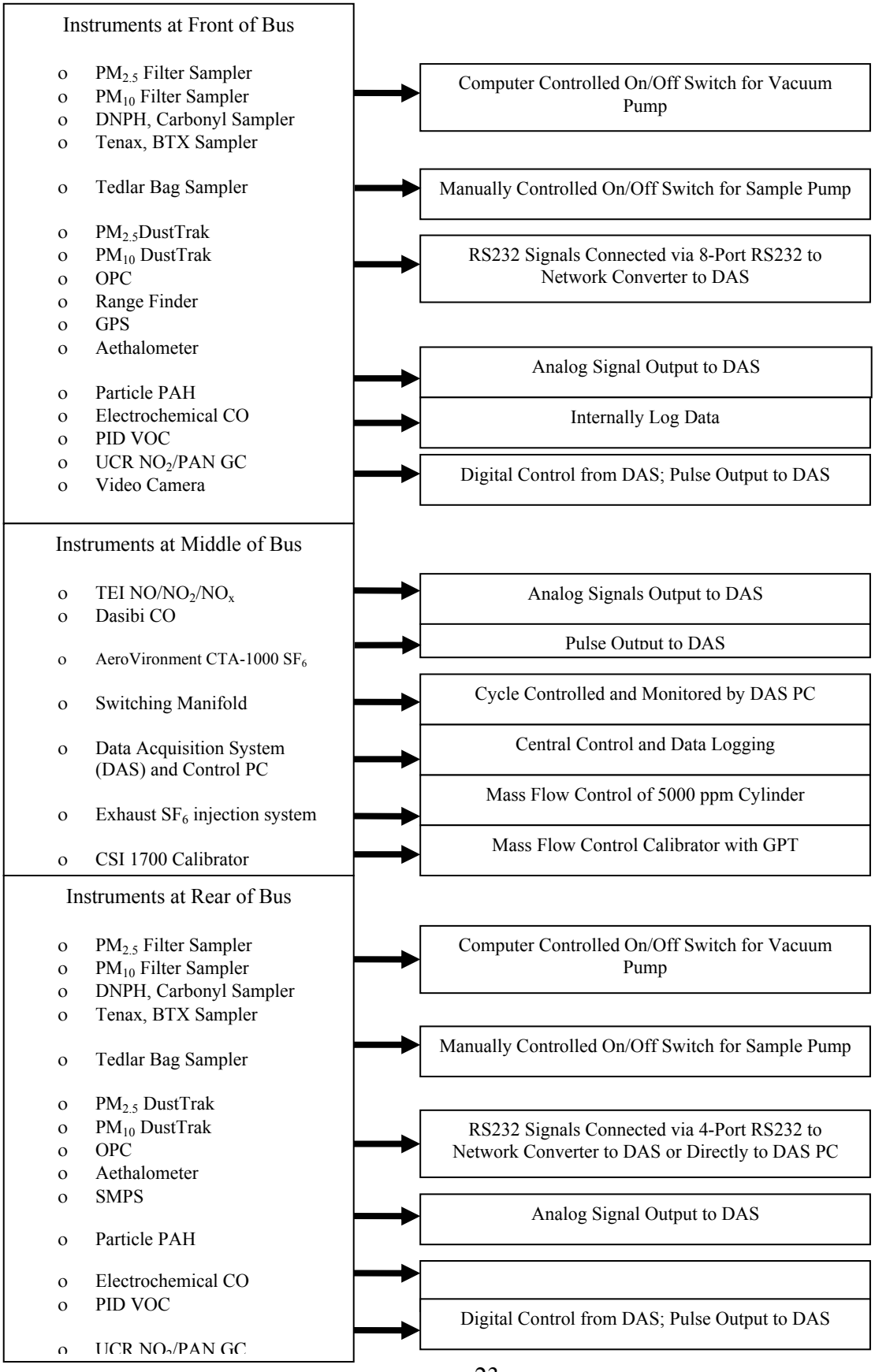
4.1.2.2 Data Collection

A schematic of the data logging design is presented in Figure 4.1.2.2. Instruments that internally logged data were downloaded via a PC. The clocks for all of these instruments were synchronized at the beginning of each run using the GPS time as a reference. All of the other instruments had either analog or digital inputs that were connected to a PC that collected data using LabVIEW® software. The data logging and control PC polled and logged data from all instruments once per second (except for the electrochemical cell CO analyzers and portable PID organic compound analyzers which internally logged data). All data were downloaded and backed up from the data logging and control PC, the electrochemical cell CO and organic compound analyzers on a daily basis. For data analysis, we obtained the ten-second median from our one-second data for all real-time instruments. This level of resolution was justified as we observed rapid instrument response to changes in concentrations inside the bus on the order of seconds.

4.1.2.2.1 Checkout and Installation of Instruments

Concurrent with the procurement and preparation of the school buses, the measurement instruments were assembled, configured, and tested at the CE-CERT laboratory. This assembly task included all necessary calibration and data logging equipment. The instruments were also

Figure 4.1.2.2 Data logging design inside school bus platform.



tested for proper operation and proper interfacing with their respective calibration systems and the data-logging and control computer.

Several shakedown runs of the measurement systems in the school bus were performed at the CE-CERT facility and in the Riverside area in order to assure proper operation of the instruments while in motion. During these shakedown runs tracer test was performed in the bus in order to measure the ventilation rates within the bus under different operating conditions.

4.1.2.2.2 Sample Probe Positions

The following descriptions apply to all of the buses employed in the main study. Sampling inside the bus primarily took place at two fixed locations at the front and rear of the cabin. The equipment at the rear of the cabin always measured concentrations at the rear of the cabin. However the equipment at the front of the cabin measured concentration at the front of the cabin during 20 runs, and just outside the cabin (with the addition of longer sampling lines with the inlets placed outside the bus as described below) during 15 runs. The sampling probes were positioned to sample at the height of the breathing zone of a child seated in the bus. Large diameter inlets were used for the interior sampling probes in order to minimize velocity changes and particle losses. Sampling outside the bus while in motion was performed using a specially fabricated isokinetic probe. Separate sampling systems were used for the gaseous and particulate matter (PM) sampling trains.

The instruments were placed on platforms, consisting of three plywood sheets, which were strapped to the tops of the bus seats along the right side of each bus (see Photograph 2). The tops of the seats were approximately one meter above the bus floor. The sample inlets for the gaseous and particulate samplers were located on top of the instrument platforms, such that

(a) (b)

Photograph 2. a) Array of instruments inside a test bus
b) Preparing the instruments before an afternoon run

the inlet heights were approximately 1.3 and 1.1 meters, respectively, above the bus floor. When sampling the air outside the bus, the instruments at the front of the bus were used, and the sampling probes were located at the base of the right side exterior driver's mirror, approximately 2.5 meters above the ground.

Gaseous samples were drawn from outside, and from the front and rear of the bus, using a variety of inlets, depending on the instrument. The PID organic vapor analyzers and electrochemical cell CO analyzers located on top of the sampling platforms at the front and rear of the bus sampled directly from their inlets located on each instrument. The Tedlar bag samplers at the front and rear of the bus had their own one meter long sample lines (0.08 cm inner diameter (ID), PFA Teflon) placed at a height of about 1.3 meters above the floor of the bus. The Tenax and DNPH cartridge samplers drew their air from three common sample lines (outside, and inside front and rear) through Teflon particulate matter filters and then through 0.65 cm inner diameter PFA tubing to a manifold where air was directed to each of the instruments through a series of T-fittings. The NO_2 GC's, GFC CO, TEI $NO/NO_2/NO_x$ and SF_6 analyzers also drew their air through those sample lines. However for these instruments, the manifold had sample lines connected to three-way solenoids, which rotated the sample path from outside the bus, to the front of the bus, then to the rear of the bus. The sample was drawn from each location for four minutes. This cycle repeated every 12 minutes. This was done to accommodate the sampling requirements for the SF_6 tracer gas analyzer. All sample inlets for those instruments attached to the switching manifold were located approximately 1.3 meters above the floor of the bus (front and rear), or 2 meters above the ground (outside).

PM sampling equipment was located at the front and rear of the bus. The equipment located at the rear of the bus was used to sample air from inside the bus near the rear only. Equipment located near the front of the bus alternated (depending on sampling requirements for specific tests) between sampling PM from inside the bus near the front and from outside the bus. The inlets for the PM sampling equipment located at the front of the bus all drew their samples from a common manifold.

For sampling outside the bus, a specially designed isokinetic probe was used (Figure 4.1.2.3). The air entered a 1 cm ID copper tube approximately one meter long. The inlet protruded out the front right window of the bus about 0.2 meters (see Photograph 3). The inlet was filed to a knife-edge and pointed directly forward to sample into the wind with minimum particle loss as the bus traveled down the road. The air flow rate was set to provide isokinetic sampling at a nominal bus speed of 20 m/s (40 mph). After the air passed through the inlet tube, it entered a 13 cm long section of tubing, 4 cm in diameter, and then passed into a 45 cm long "t-section" where additional bypass air, drawn to maintain isokinetic conditions, was removed. Sample air exited the "t-section" through a 23 cm long section of 4 cm diameter tubing into a 13 cm long by 10 cm diameter "stilling chamber" which had seven outlet ports. Two of the outlet ports were approximately 30 cm long, 1 cm ID copper tubing which ran from the stilling chamber to the Harvard Impactors. One of the outlet ports had 0.6 cm ID "nonstatic" plastic tubing (approximately 1 meter long) running to the front Aethalometer. The four remaining outlet ports had 0.4 cm ID "nonstatic" plastic tubing (each approximately one meter long) running form the stilling chamber to the DustTrak, OPC and PAH instruments. A tap on the side of the stilling chamber ran to a transducer used to monitor the pressure in the stilling chamber.

When the front system was used to sample air from inside the bus, the copper tube was removed from the system, allowing air to enter directly into the 4 cm diameter section and the bypass flow was turned off.

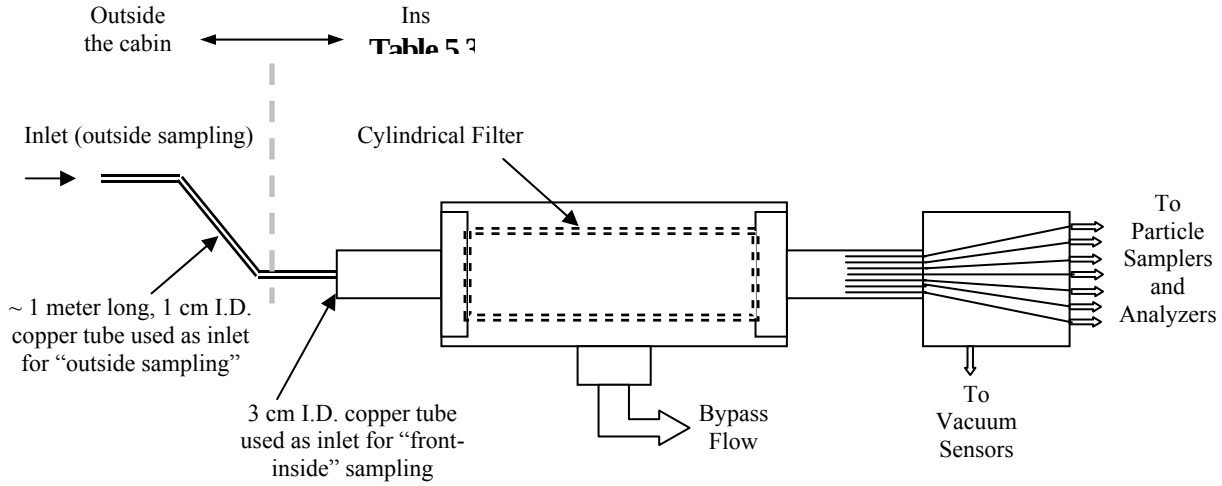


Figure 4.1.2.3 Sampling system (including the isokinetic inlet) used at the front of the bus to sample inside and outside aerosol.

Photograph 3. Inlet of the sampling line for outside measurements of gaseous species.

The rear PM instruments all drew air from a common location next to the rear gaseous sample inlets. No additional lines or inlets were used for the Harvard Impactor filer samplers. A 0.5 meter long, 0.6 cm ID nonstatic plastic tube ran from the common sampling location to the Aethalometer. Lengths of 0.4 cm ID nonstatic tubing between 0.5 and 1 m long ran from the common sampling location to the DustTrak, OPC, PAH and SMPS instruments. Table 4.1.2.2 summarizes the total length of the sampling lines used with these instruments. For the majority of our analyses, we focused on the rear instruments. PM2.5, black carbon, fine particle counts

and PAH (the particulate species we analyzed for this report), had sampling lines in the rear of 0, 0.5, 0.7 and 1.2 meters, respectively. See section 5.1.4 for a discussion of potential sampling line losses for particulate species measured in this study.

In the main study, bus stop measurements were conducted using an equipped bus parked at the bus stop. The sampling inlets for these measurements were located about 1.5 meters above the ground.

Table 4.1.2.2 Length of sampling lines

	Sampling line length (meters)		
Instrument	Rear	Front/inside	Front/outside[2]
Harvard Impactor - PM10	0[1]	0	3
Harvard Impactor - PM2.5	0	0	3
DustTrak - PM10	1.6	0.3	3
DustTrak - PM2.5	1.5	0.2	3
OPC	0.7	1	3.7
Aethelometer	0.5	1	1
PAH Analyzer	1.2	1	3.7

[1]Harvard Impactors were used with an open face inlet (no sampling line or adapter) for several experiments.

4.1.2.3 Instrumentation

4.1.2.3.1 Integrated PM_{10} and $PM_{2.5}$ Mass Concentration

Filter samples were collected using custom sampling systems designed for portable use. The inlets were of the Harvard design (Turner et al., 2000), which have been shown to have effective cuts at 2.5 and 10 μm while sampling at 20 L/min. The flow rates were controlled by a needle valve and measured with a rotameter and calibrated against a volumetric flow rate sensor. The samples were collected on 37 mm Gelman Teflon filters with a 2.0 μm pore size. A Cahn Model 34 microbalance at the CE-CERT laboratory was used to determine the weight of the filters to within ±2 μg before and after sampling. All filters were equilibrated at 23°C and 40% RH for at least 24 hours prior to weighing. Filters were weighed a minimum of three times before and after sample collection. If all three weighings were not constant to within 3 μg, the filters were reweighed until they were.

4.1.2.3.2 Real-Time PM_{10} and $PM_{2.5}$ Mass Concentration

Real-time PM_{10} and $PM_{2.5}$ measurements were made using Thermo Systems Inc. Model 8520 DustTrak Aerosol Monitors. Impactors were used to perform the necessary size cuts. The PM concentration that made it past the impactor was then determined by measuring the intensity of the 90° scattering of light from a laser diode. The instrument sample flow rate was 1.7 L/min. The averaging time was adjustable from 1 to 60 seconds, and an averaging time of one-second was used. The instruments were calibrated at the factory with Arizona road dust (NIST SRM 8632). Collocated sampling was performed each week to insure comparability.

Throughout this report, all PM2.5 data are from the Harvard Impactor integrated samples unless stated otherwise (i.e., as data from the DustTrak instruments).

4.1.2.3.3 Real-Time Particle Counts

4.1.2.3.3.1 Optical Particle Counter

Climet Spectro .3 Optical Particle Counters were used for particle count concentration measurements in sixteen size bins from 0.3 to 10 μm. Only the size range between 0.3 – 0.5 μm was used for data analysis in this report. This size range was selected because the highest number concentrations are found in the smaller size bins. Thus, this size range represents the majority of fine particle counts, while simplifying the analysis to just two size bins. Sample air was drawn through a laser beam inside the analyzer where particulate matter passes through the beam one particle at a time. As a particle passes through the beam it scatters light roughly in proportion to particle size. Optical detectors detect the scattered light and send an electrical signal proportional to the scattered light to the digital signal processor (DSP). The DSP performs a pulse height analysis of the signal to determine the size of the particle. The DSP then sends the particle size results to one of sixteen counters ("size bins") where the total number of particles in the size bins were accumulated. The instruments flow rates were set at 1.0 L/min. The instruments counting periods were set to fifteen seconds with a ten second "wait" period prior to starting a new counting period.

4.1.2.3.3.2 SEMS

A CE-CERT Scanning Electrical Mobility Spectrometer (SEMS) was used to measure particles in the range of 0.03 μm to 0.8 μm electrical mobility diameter. The instrument consisted of three major components: a Thermo Systems Inc. (TSI) model 3077 ^{85}Kr neutralizer which generates a known charge distribution on the aerosol; a TSI model 3081 differential mobility analyzer long column which selects for particle sizes based on the voltage applied and the particles' electrical mobility; and a TSI model 3760A condensation particle counter used to detect particles. The analyzer's four gas flows were calibrated at CE-CERT using a primary flow calibrator and the particle size was calibrated using aerosolized polystyrene latex spheres. The analyzer operated in a ten-minute cycle mode. It spent the first 75 seconds of each cycle scanning the 0.03 to 0.8 μm size range in seventy-five, one-second-long increments. It then spent the next 525 seconds fixed ("fixed period") on a single particle size that was expected to be the best indicator of the presence of diesel exhaust particulate matter. For all runs except the last run, the fixed-period particle diameter was 180 nm. For the last run, the fixed-period particle size was 50 nm. The instrument output particle count data were output and processed at a once per second rate.

4.1.2.3.4 Real-Time Black Carbon

The black carbon concentrations were measured using Magee Scientific Aethalometers. These instruments drew sample air through a 0.5 cm^2 spot on a quartz fiber filter tape. Infrared light at 880 nm was transmitted through the quartz tape and detected on the back side of the tape using photodetectors. (One detector sensed the light transmitted through the spot where the air was drawn through and the second detected light transmitted though an unused section of tape in order to correct for changes in the light source intensity and changes in the tape characteristics.) Decreases in the amount of light transmitted through the spot on the quartz tape were proportional to the amount of elemental carbon and "heavy" organic molecules collected. The instrument's response to the change in light transmittance was reported as "black carbon" (BC). The instrument's sample flow rate was maintained using mass flow controllers. The concentration of BC in units of mass of BC per volume of air (e.g. "μg/m^3") was determined by

the instrument from the flow rate and change in light transmittance data. When the light transmittance through the collection spot on the quartz filter had decreased by seventy-five percent, the quartz tape automatically advanced to a fresh section of filter. Each time the filter tape automatically advanced, the instrument recalibrated for approximately one minute prior to restarting sampling.

4.1.2.3.5 Real-Time Particle Phase PAH

EcoChem Model PAS 2000 analyzers were used to measure the concentrations of particle-bound polycyclic aromatic hydrocarbons (PAH). The instrument utilized the principle of photoionization of total particle-bound PAH by means of an ultraviolet lamp. The wavelength of the light was selected such that only the PAH absorbed on aerosols were ionized, while gas molecules and non-carbon aerosols remained neutral. The aerosol particles that had PAH molecules adsorbed on the surface emitted electrons, which were subsequently removed when an electric field was applied. Remaining positively charged particles were collected on a filter inside an electrometer where the charge was measured. The resulting electric current established a signal which was proportional to the concentration of total particle-bound PAH. The lower threshold of this method was about 3 ng/m^3 total particle-bound PAH.

The PAH instrument's full scale was set to read a maximum of 500 ng/m^3 in the front, and 1000 ng/m^3 in the rear, during the first 17 runs of this study. These runs included buses HE1, HE2, HE2, RE1 and RE2 (all diesel buses). Starting with Run 18, and on all subsequent runs (including TO1 and CNG), the scale was increased to read a maximum of 2000 ng/m^3 for both front and rear instruments. The implications of this change for our inter-bus comparisons of PAH concentrations are discussed in Section 5.3.3.

The location of the Aethalometer, particle count analyzer, PAH analyzer and the Harvard Impactors in a test bus with associated power cables and sampling lines is shown in Photograph 4.

(a)

(b)

Photograph 4. a) Data and power cables and sampling lines
b) Aethalometer, PC analyzer, PAH analyzer, and Harvard Impactors

4.1.2.3.6 Gaseous Hydrocarbons

A multifaceted approach using both rapid response "survey" real-time monitors and integrated sample collection followed by subsequent laboratory analysis was used. This allowed for full characterization of the gaseous hydrocarbons by "time-consuming" (but highly sensitive) laboratory chromatographic methods and the rapid response of a real time analyzer that could characterize short episodes of high concentrations.

4.1.2.3.6.1 Tenax Cartridges

Samples for aromatic hydrocarbons (benzene, toluene, and xylenes) were collected on a Tenax cartridge and analyzed by GC analysis, per the U.S. EPA TO-1 Method (U.S. EPA, 1988). The cartridges contained 0.1 gm of Tenax. The maximum target sample volume for these cartridges had been determined to be at least 3.0 liters before benzene breakthrough would occur; with the other aromatics having a longer retention time. The sample flow rate was set to 40 cc/min in order to stay well below the breakthrough volume for the study's projected 60 minute sample periods. A Hewlett Packard HP5890 II gas chromatograph with a thermal desorber, capillary column and flame ionization detector (FID) was used for the analysis

4.1.2.3.6.2 Tedlar Bags

Samples for 1,3-butadiene were collected in 8L Tedlar bags at a nominal sample flow rate of 100 cc/min. A potassium iodide (KI) trap was used to remove the ambient O_3 before the sample entered the bag. The samples were hand-carried to the CE-CERT analytical laboratory for GC analysis after the end of sampling. An aliquot from the bags were drawn through a sample loop of known volume on the GC. A Hewlett Packard HP5890 II gas chromatograph with a fixed volume sample loop, capillary column and flame ionization detector (FID) was used for the analysis.

4.1.2.3.6.3 Portable PID

The total amount of aromatic and other highly conjugated aromatics were continuously monitored with a ppbRAE Model PGM-7240 portable photo ionization detector. Sample air was drawn through the instrument's reaction chamber where it was continuously irradiated with high energy ultraviolet light. Compounds present that had a lower ionization potential than that of the irradiation energy (10.6 electron volts) were ionized. The ions formed were collected in an electrical field, producing an ion current that was proportional to total compound concentration.

4.1.2.3.7 Gaseous Aldehydes and Ketones

The measurement technique used was a variant of U.S. EPA Method TO-11 (U.S. EPA, 1988) for carbonyls in which 2,4-dinitrophenylhydrazine (DNPH) was impregnated on silica Sep-Pak cartridges. This procedure used C18 Sep-Pak cartridges (Waters/Millipore Corp., Milford, MA), which had been impregnated with acidified DNPH reagent for ambient sampling. The amounts of both the hydrazine and acid were optimized to achieve efficient collection of the carbonyl compounds and protection from ozone destruction of the captured carbonyl derivative. When ambient air was drawn through the cartridge at 1 L/min, carbonyls in the air sample were captured by reacting with DNPH to form hydrazones, which were extracted and then separated and quantified using HPLC (Fung and Grosjean, 1981). It had been shown that the silica cartridge when used alone had a significant negative ozone artifact, and that this method measured carbonyls comparably to the silica method with an ozone removal device.

The method analyzed samples for nine individual species: formaldehyde, acetaldehyde, acetone, propanal, crotonaldehyde, methylethylketone, butyraldehyde, benzaldehyde, and m-tolualdehyde, plus C_5, C_6 and $>C_6$ aliphatic carbonyls. Aliphatic carbonyls (C_5, C_6, and $>C_6$) were usually measurable but lower in concentrations compared with the C_1-C_4 carbonyls. Except for the straight chain aldehydes (e.g. pentanal, hexanal, etc.), resolution of the other isomers of C_5 and higher carbonyls was incomplete, and thus inaccurate. The latter were more appropriately reported as a group by carbon number. Pure DNPH derivatives of the aldehydes and ketones were used to prepare calibration standard for the high performance liquid chromatography (HPLC).

Field blank variability established the LQL (lower quantifiable limit), which for C_1-C_7 carbonyls was typically 0.5 ppb or lower (at 3 times the standard deviation of the blank variability).

Accuracy of this method was approximately $\pm$15% for formaldehyde based on comparison studies with long path spectroscopic techniques in an ambient air setting (Fung and Wright, 1990; Lawson et al., 1990). Since the basic chemistry of the DNPH method was the same for all carbonyls, the accuracy for higher carbonyls was expected to be in the same range.

4.1.2.3.8 Real-Time Carbon Monoxide

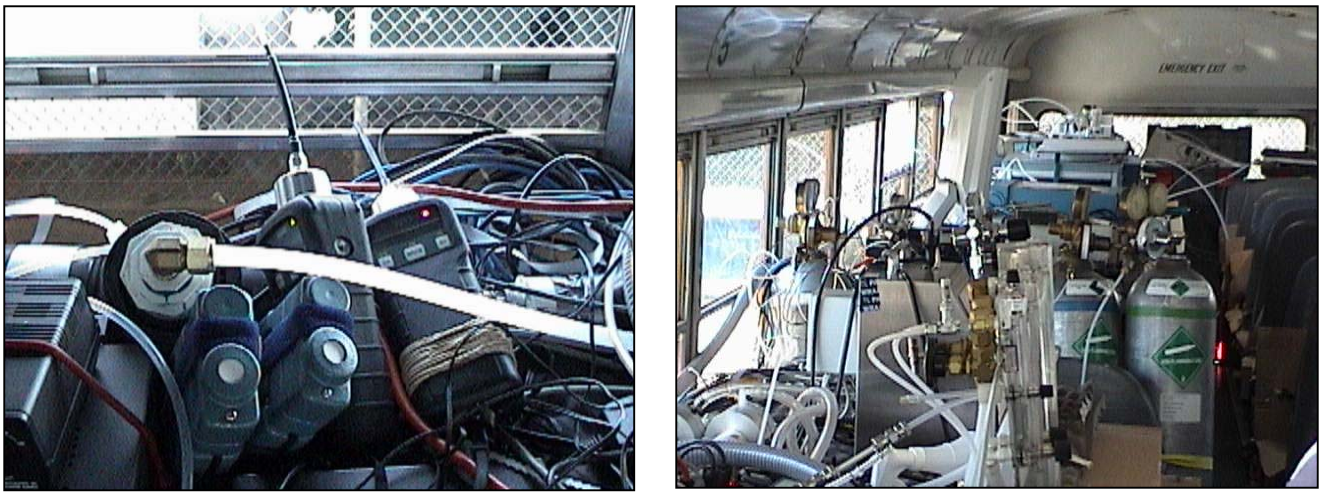
The carbon monoxide (CO) concentrations were measured using Langan electrochemical sensors. These instruments were designed to operate in a passive environment and have no sample flow. The instrument had a diffusion membrane that CO could pass through. The membrane was exposed to ambient air on one side and contained a solution that reacted with CO on the other. Ambient CO that diffused through the membrane reacted with the solution, creating an electric current that was proportional to the ambient CO concentration. The instrument was calibrated by sampling known concentrations of CO prepared in air.

A third CO analyzer was added for the main study, a Dasibi Model 3003. This analyzer measured CO using an infrared detector, and is an U.S. EPA Reference Method for CO. The analyzer was installed near the center of the bus and connected to the switching manifold described in Section 4.3.11.

The CO and VOC monitors along with associated cables, standard gas tanks, rotometers and sampling lines are shown in Photograph 5.

4.1.2.3.9 Real-Time Nitrogen Dioxide

The NO_2 concentrations were measured using two gas chromatographs (GCs) to measure NO_2 and peroxyacyl nitrates (PAN). Ambient air was brought into the analyzer and drawn through a sample loop. Once per cycle (one minute cycle time) a valve switched and the sample was pushed into a capillary column using air carrier gas. NO_2 and PAN were separated from each other in this column. The separated constituents eluted from the column onto a fabric wick wetted with luminol solution. NO_2 and PAN react with luminol, emitting photons of light proportional to their concentration. The central portion of the wick was viewed by a photomultiplier tube (PMT). Photons of light from the reaction with luminol entered the PMT



(a) (b)

Photograph 5. (a) Power and data cables, sampling lines, CO monitors, and VOCs analyzers; (b) Cables, standard gas tanks, rotometers and sampling lines

where the signal was multiplied through a series of electrified plates that successively multiplied and converted the signal from "light energy" to "electrical energy." This amplified electrical signal was output to the computer controller and data logger. Because NO_2 and PAN eluted from the instrument at different times, the data logging system could separately process and store the data from the PMT for these two compounds. These instruments did not detect HONO.

A third analyzer was added to the main study to measure oxides of nitrogen. A Thermo Environmental Instruments (TEI) Model 42 $NO/NO_2/NO_x$ analyzer was installed near the center of the bus. It sampled from of the same switching manifold described in Section 4.1.2.2.2. This analyzer was added to the program for several reasons: continuous sampling coverage from three locations, for quality control, and for ease of calibration. The preferred method for calibrating the NO_2/PAN GC's for NO_2 was by gas phase titration of nitric oxide. In order to perform this type of calibration, it was necessary to have an instrument that measured NO and NO_x. Thus, the data from this instrument was used to calibrate the NO2/PAN GC's and not for the analyses included in the remainder of this report.

The TEI Model 42 measures oxides of nitrogen by the chemiluminescent reaction of NO with ozone. The analyzer switches between two sample paths every ten seconds. The first sample path directs the sample directly into a reaction chamber where it was mixed with high concentrations of ozone (O_3). The NO and O_3 reaction generated photons of light proportional to the NO concentration that are detected and processed by the analyzer. The second sample path directed the sample through a converter that converted other oxides of nitrogen to NO. The sample was then directed to the reaction chamber. The results from the second sample path were reported as NO_x. NO_2 was calculated as the difference between the NO_x and NO responses. Since there may be additional non-NO oxides of nitrogen present (other than NO_2), this method can "over-report" NO_2 concentrations. However, the TEI 42 is an U.S. EPA Reference Method for NO_2.

4.1.2.3.10 SF_6 Tracer Gas

The sulfur hexafluoride (SF_6) concentrations were measured with an AeroVironment Model CTA 1000 analyzer. The primary task of this analyzer was to determine if a significant amount of the bus exhaust was entering the passenger compartment of the bus. For this task, SF_6 was injected into the bus exhaust systems as described in Section 4.1.4. A second use of the instrument was to determine the ventilation (or air exchange) rates in the passenger compartment as described in Section 4.1.3. This Aerovironment instrument uses electron capture detection after water and oxygen are removed from the sampled air. The instrument was developed for operation on a moving platform and has a sensitivity of approximately 0.010 ppb with a response time of about three seconds. In order to account for analyzer baseline drift, it was set up to frequently sample reference SF_6-free air.

4.1.2.3.11 Range Finder

In order to measure the distance from a vehicle being followed, a Laser Optronix DME 200 laser range finder was acquired. This device is palm-sized and operates with a pulsed 904 nm laser. If the reflectivity of the target is good, its range is 1-300 m, with an accuracy and resolution of about 1.0 m. The instrument outputs data once per second to the data logger. However, due to problems with trying to use the instrument on a "moving platform," as well as problems trying to measure distances to vehicles that were in the field of view for often considerably less than one second, no viable data were obtained from this instrument. Instead distances to vehicles was estimated when necessary from the video tape.

4.1.2.3.12 Bus Location

The bus location was determined with a Garmin GPS MAP76 global positioning system with a Wide Area Augmentation System (WAAS) corrections system. The system provides position accuracy of about 2-3 meters. The GPS has a 12 parallel channel receiver to continuously track and use data from up to twelve satellites. The WAAS system is a broadcasted "signal integrity" signal that is determined by fixed ground-based reference stations. The GPS uses the WAAS correction information to increase the accuracy of the positioning information. In addition to horizontal position (e.g. latitude and longitude or UTM coordinates), the corrected GPS system also provides elevation and velocity data. These data were displayed on a liquid crystal display on the GPS and were output digitally (RS232) for logging along with the air quality data on the data logger.

The GPS unit was used as a time reference during this study. The clocks on all other devices were set to the GPS time on a daily basis.

4.1.2.3.13 Meteorological Data

Temperature and relative humidity were measured inside the bus using a Rotronics Model MP101A sensor. The sensor determined temperature using a 100 ohm platinum resistance temperature device (RTD) and relative humidity using a Hygrometer C94 capacitive humidity sensor. The instrument included internal signal conditioning to process and output two voltage signals that were proportional to the temperature and relative humidity, respectively.

Hourly wind speed and direction data were obtained from the nearest AQMD fixed site monitoring stations for the dates of times of each bus run. These data provided an estimate of

the prevailing area-wide wind conditions during a bus run, but did not account for micrometeorology around the bus (e.g. street canyon effects) throughout the run. Therefore, tnterpretation of the data based on these measurements requires caution. In general, and as expected based on land/sea temperatures, there was little or no wind during the morning runs, while the afternoon runs typically occurred during on-shore flow conditions with significant wind speeds. Therefore, only the afternoon wind data were useful as a potential explanatory variable for differences in pollutant concentrations between afternoon runs. For the primary urban route, hourly wind speed data from the two nearest AQMD stations (West Los Angeles (WLA) and downtown Los Angeles (CELA)) for 15:00–16:00 and 16:00-17:00 (the hours covering all runs on this route) were averaged and this value assigned to the run. The same procedure was used for the second urban route using the Hawthorne (HAWT) and Lynnwood (LYNN) AQMD stations. Wind direction measured at the AQMD stations was consistently from the West or Southwest for all afternoon runs.

4.1.2.4 Use of Video Camera and Digital Camera

A Sony DXC-390 video camera was mounted at the front of the buses to record traffic conditions in the lane in which the bus was traveling, as well as the adjacent lanes during all measurement periods. This camera has an RS-232 Interface, a resolution of 800 TV lines, and a 10-bit digital signal processing system. In addition, the camera has a variable speed electronic shutter, which provides the capability to capture clear images of high speed moving objects. The camera was set to a wide angle to view as much of the scene as possible. The camera included a "time stamp" feature for adding date and time information to the video. The clock in the video camera was synchronized with the GPS master clock time prior to each run.

4.1.2.5 Documentation of Bus Commute, Traffic Conditions and Events During Each Run

In addition to the video camera, a software program developed and tested by the UCLA field team was used to record traffic conditions, bus movement and other observations during each bus run. Detailed observations during each bus run, such as when the bus was moving, idling, or at a bus stop, identification of vehicles in front of the bus, and traffic conditions were recorded and time stamped in a data log using a laptop computer. The clock used for the time-stamp was also synchronized with the master clock. Comparison of peak concentrations of pollutants and the observations recorded in the data log were used to identify conditions that led to the highest pollutant concentrations inside the bus during selected runs. All of the data collected during this study, will be submitted in electronic format to the ARB for further review and analysis.

4.1.3 Bus Ventilation Rate Measurements

We emphasize here that the experiments conducted with SF_6 to measure ventilation rates in each bus at several speeds and with windows opened or closed (as explained below), were all conducted at CE-CERT prior to each bus being driven to the Westside. Thus, the use of SF_6 for the ventilation rate measurement was completely separated in time and location from the SF_6 tracer studies designed to determine the contribution of a bus's own exhaust to within-cabin concentrations. In short, the SF_6 ventilation measurements did not affect the subsequent exhaust tracer measurements in any way.

Exchange of the air inside bus cabins is controlled by several factors, including window position (i.e. closed, opened or some position in between) and penetration of outside air through leaks around windows, the main door, roof vents, emergency exits and other nonspecific locations, depending on bus speed and wind speed. Air exchange rates may also be significantly affected by other characteristics of the bus including differences in construction specifications for older buses versus newer buses. Some of these characteristics may increase as a bus ages (e.g. leaks). For this study, air exchange rate or ventilation rate tests were performed with the windows open and the windows closed at bus speeds of 0, 20, and 40 mph. Air exchange rates inside the bus were measured by releasing a SF_6 tracer gas inside the bus and monitoring the concentration of the gas over time.

High concentration (5,000 or 10,000 ppm) SF_6 cylinder gas was used as the tracer. A plastic syringe was filled with 10-30 cc of the SF_6 cylinder gas. The amount used was varied as a function of window position, bus speed and cylinder concentration, to obtain interior SF_6 concentrations within the SF_6 analyzer operating range in the available experiment time. SF_6 was released from the syringe as a line source in the aisle of the cabin as the operator walked from the back to the front of the bus. The operator then walked back to the rear of the bus, using his body to help rapidly mix the SF_6 inside the bus.

The SF_6 concentration was continuously monitored from the middle of the bus at the top edge of the seats facing the aisle. All 0 mph tests were performed in the CE-CERT parking lot with the bus facing north. All the 20 and 40 mph tests were performed on Riverside Drive between Riverside and Colton (except the CNG bus tests). Riverside Drive was a four lane road with a posted speed limit of 45 mph, with more than two miles between traffic stops and light traffic during the period we conducted the ventilation tests. Although there were variations from test to test, typically the test began with windows closed traveling north on Riverside Drive at 20 mph, followed by traveling south on Riverside Drive at 40 mph. The windows remained closed for the duration of the test. A second test was performed in a similar manner, again traveling north on Riverside Drive at 20 mph, followed by traveling south on Riverside Drive at 40 mph, this time with the windows open for the duration of the test. Except for the tests on the CNG bus, ventilation tests were performed just prior to starting the rural/suburban run. Due to equipment problems, the ventilation tests at 20 and 40 mph for the CNG bus were performed at the end of the rural suburban run on Grand Avenue from Diamond Bar to Glendora and back. The speeds and window positions for these tests were similar to those used for previous tests on Riverside Drive. The 0 mph tests were performed over periods ranging from ten minutes to over an hour. The 20 and 40 mph tests were performed over periods of two to ten minutes.

For all tests, the amount of SF_6 released resulted in concentrations beyond the instrument's full scale. Typically, after a brief period beyond full scale (during which the SF_6 was also mixing throughout the bus), the concentration of SF_6 dropped to the monitor's full scale. At this point the SF_6 decay rate was measured and used to determine the air exchange rate in the bus.

To determine the SF_6 exchange rate, we used the following equation, which assumed bus ventilation rates are a first order process:

$$(SF_6)_t = (SF_6)_0 * [1 - \exp(-t/\tau)] \quad (4.1.3.1)$$

where $(SF_6)_0$ is the SF_6 concentration at time zero, $(SF_6)_t$ is the SF_6 concentration at time t, and τ is the time constant. The SF_6 removed from the bus (and the percentage of bus air exchanged) over time is 63%, 87%, 95%, 98%, and 99% for times equal to the first through fifth time constants, respectively. Although there were many factors that controlled mixing and exchange of air in the bus, inspection of the plots of SF_6 versus time indicated a first order decay provided a reasonably accurate fit of the data.

The results of the ventilation tests are presented in Table 4.1.3.1, which shows the time constant, or the time required for 63% of the bus air to be exchanged. The time for essentially complete exchange is three times (i.e. 95% exchange) to five times (i.e. 99% exchange) longer. The shorter the time for air to exchange, the higher the ventilation rate. The ambient wind speed for all stationary ventilation tests was approximately 1-3 m/sec. The ventilation rate when the bus was stationary with the windows closed varied from moderate for HE3 (under ten minutes for 63% of the bus air to be exchanged) to no detectable ventilation during the fifteen to forty-two minutes that the test was conducted (HE1, HE2, RE1, RE2, TO1, CNG). The ventilation rate was higher for windows open compared to windows closed (for the same bus speed). The ventilation rate was also higher when the bus was driven at 40 mph compared to 20 mph. The tests showed there were a significant number of "leaks" in all buses tested. As can be seen in Table 4.1.3.1 (excluding RE2, windows open at 0 mph), even with the windows closed, the ventilation rates for the buses traveling at 20 and 40 mph were higher than the ventilation rates for the same buses at 0 mph with the windows open.

Significant differences in the ventilation rates between buses were observed. Because only a single set of ventilation tests was performed for each bus, it is not possible to determine if these differences were definitely due to differences between the buses, or if they were due to other factors, such as different wind conditions when each bus was tested. One indication of the possible major effect that wind speed (as opposed to bus speed) may have on ventilation rate comes from the windows open at 0 mph test performed on the last bus. The CNG bus had a moderately low ventilation rate with windows open at 0 mph (seven minutes for 63% of the air to be exchanged), while the other buses had higher ventilation rates, (between two and four minutes for 63% of air to be exchanged) for 0 mph. We assume that when the bus windows are open at 0 mph, any major differences observed are due to wind speed and not to unique characteristics of one bus compared to another. Hence, the most probable explanation for the low ventilation rate in the CNG bus with windows open at 0 mph was the calm wind conditions during the test. For each series of 0, 20 and 40 mph tests, there was a factor of two to a factor of four difference between the bus with the highest ventilation rate to the bus with the lowest ventilation rate. Because of the likely significant role that wind contributed to these differences (especially for the 0 and 20 mph tests), it is, in general, difficult to draw conclusions about possible bus-to-bus differences in ventilation rates. However, air exchange rates were typically much higher with open windows and much higher at higher speeds, regardless of window position.

WEEK # BUS TEST CONDITION	1 HE1 Response Time (First Time Constant) (min:sec)	2 HE2 Response Time (First Time Constant) (min:sec)	3 HE3 Response Time (First Time Constant) (min:sec)	4 RE1 Response Time (First Time Constant) (min:sec)	5 RE2 Response Time (First Time Constant) (min:sec)	7 TO1 Response Time (First Time Constant) (min:sec)	8 CNG Response Time (First Time Constant) (min:sec)
Windows closed 0 mph	> 15 min	> 30 min	09:47	> 30 min	> 30 min	> 15 min	> 42 min
Windows open 0 mph	02:39	02:38	03:16	03:57	00:40	02:18	07:00
Windows closed 20 mph	NA	01:20	01:52	01:56	03:31	04:38	02:00
Windows closed 40 mph	NA	00:52	00:38	01:05	02:00	01:22	01:21
Windows open 20 mph	NA	00:31	00:58	00:48	00:47	00:23	00:26
Windows open 40 mph	NA	00:16	00:29	00:17	00:37	00:12	00:23
Number of windows opened for "window open" condition at 0 mph	every other window opened 7 cm	all windows and door opened	Four windows on both sides (8 total) down 4 inches	Three windows on both sides (6 total) down 4 inches	3/4 of windows opened 4 inches	6 of 12 of right side and 7 of 13 on left side opened 4 inches	Six windows on each side (12 total) opened 3 inches
Number of windows opened for "window open" condition at 20 and 40 mph		Four windows on left side down 4 inches	Three windows on both sides (6 total) down 4 inches	Three windows on both sides (6 total) down 4 inches	3/4 of windows opened 4 inches	6 of 12 of right side and 7 of 13 on left side opened 4 inches	Six windows on each side (12 total) opened 3 inches
Test notes for "windows closed" condition at 0 mph	one front window and one rear window down 3 inches						
Test notes for "windows closed" condition at 20 and 40 mph		Driver's window opened 1/2 way					

Table 4.1.3.1 Ventilation time constants measured inside buses at various speeds and for open and closed windows (see text).

4.1.4 Characterization and Justification for the Selected School

For this study we sought to identify a school with a diverse student population drawn from various parts of Los Angeles, and which offered a broad range of travel distance, roadway type, and traffic congestion scenarios associated with bus commutes. Due to resource constraints the present study was unable to test across a full range of possible commutes, meteorological conditions, bus manufacturers, model years, school districts, geographic locations in California, etc. Our results are "representative" only to the extent the commutes, buses, conditions, areas, etc. we studied are representative of southern California school districts.

The school selected for both the pilot and main studies was the Brentwood Science Magnet School (BSMS), located at 740 Gretna Green Way, in West Los Angeles. The BSMS was in Local District D of the LAUSD. It was a K-5 facility with a total enrollment of 1,209 students in the 1999-2000 school year. Because it was a magnet school, BSMS was racially and ethnically diverse with a student population demographic of approximately 20% Asian, 20% African-American, 25% Hispanic, and 35% Caucasian.

We felt the BSMS school provided high exposure scenarios and yet was located close to UCLA, allowing ready access and the opportunity to optimize time spent on acquiring data. Also, because BSMS was a magnet school, it provided a wide range of distance and congestion scenarios for its bus commutes.

Considerable initial research concerning the selected school, and its associated bus behavior, was conducted prior to the start of the pilot study, including a detailed analysis of routes, patterns, and travel times for all nineteen routes used by the BSMS. A relationship was established with the school through the school's principal and its science advisor, and also through the LAUSD Transportation Branch and its Director, Antonio Rodriguez. The school principal, Sharon Katz, provided approval for conducting the pilot study at BSMS, while the LAUSD Transportation Branch provided the detailed route sheets for all the bus commutes to and from this school. Additional support was provided by Dr. Angelo Bellomo, Director of the LAUSD Environmental Health and Safety Branch.

We expected the bus commutes from the BSMS were capable of producing high exposure scenarios on the bus during the commute due to the long distances of the commutes and the areas of Los Angeles they traveled through. Figure 4.1.4.1 shows the starting locations of each of the bus routes bringing children to BSMS. Most of these routes involved both freeways and residential arterials, including those expected to have heavy traffic congestion. Children were bused to BSMS from over 200 schools throughout the city, with an average commute time and distance of 85 minutes and 23 miles, respectively. Typically, 85% of the children attending BSMS were transported to and from this school each day on nineteen diesel school buses.

The school property was bounded by Gretna Green Way on the west, Montana Avenue on the south, Bundy Avenue on the east and San Vicente Boulevard on the north. We characterized non-freeway traffic densities for these four arterials as low, low-medium, medium, and medium-high, respectively, with the caveat that at the student drop-off and pick-up times

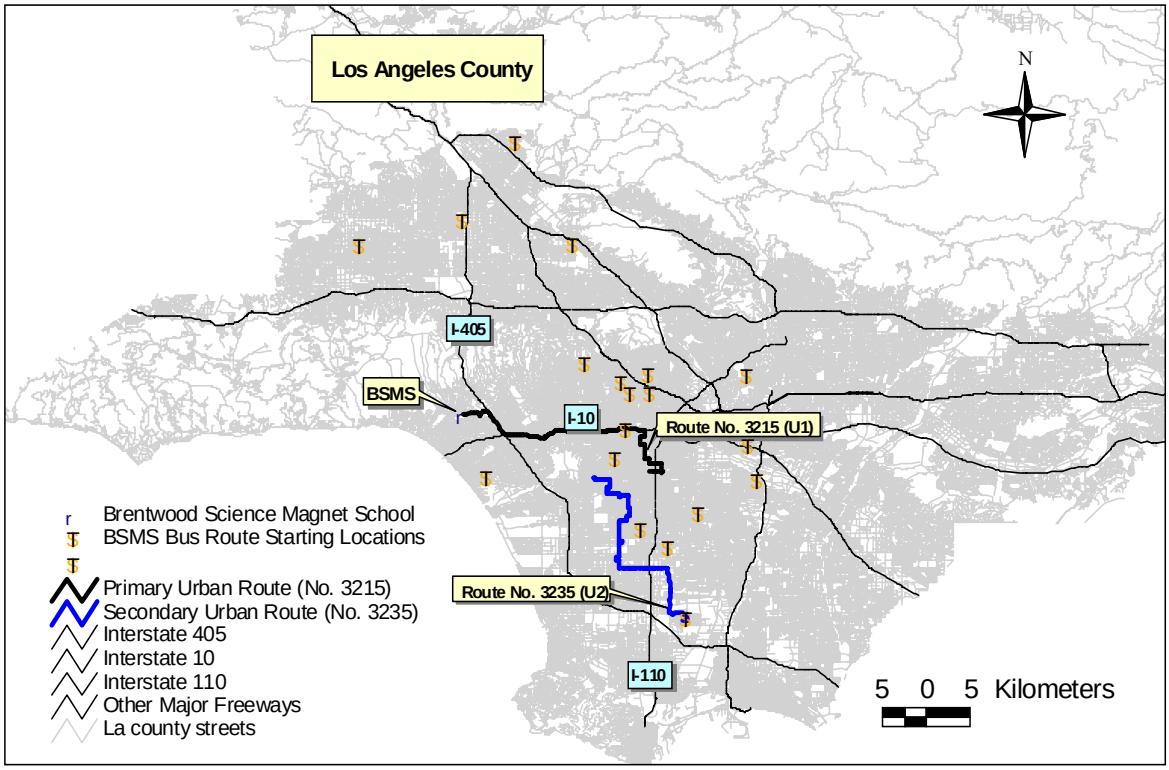


Figure 4.1.4.1 Brentwood Science Magnet School bus route starting locations.

there was additional traffic congestion on Gretna Green Way from passenger vehicles of parents, teachers and staff who were arriving or departing, as well as from the arrival or departure of school buses. Three of these streets were one lane each way, while San Vicente Blvd. was two lanes each way with a median strip of approximately 20 feet width.

A key physical aspect of the school was a block-length sidewalk, approximately ten feet wide, along the east side of Gretna Green Way. Typically, all nineteen buses bringing students to the school lined up along this sidewalk. Figure 4.1.4.2 shows the layout of the school and bus loading area. All school bus loading and unloading took place at this sidewalk. In addition, there was head-in parking for teachers and staff (and perhaps parents) on the west side of Gretna Green Way.

We acquired information about loading/unloading activities in both the morning and early afternoon. Direct observation of BSMS during five days in December 2000 and September and October 2001 was conducted and the following observations made. In the morning, buses arrived and parked on Gretna Green Way, along the sidewalk in front of the school, between 7:40 and 8:00 (see Photograph 6). Each bus turned off its engine as soon as it parked, a key observation and in apparent compliance with LAUSD regulations. Children quickly unloaded

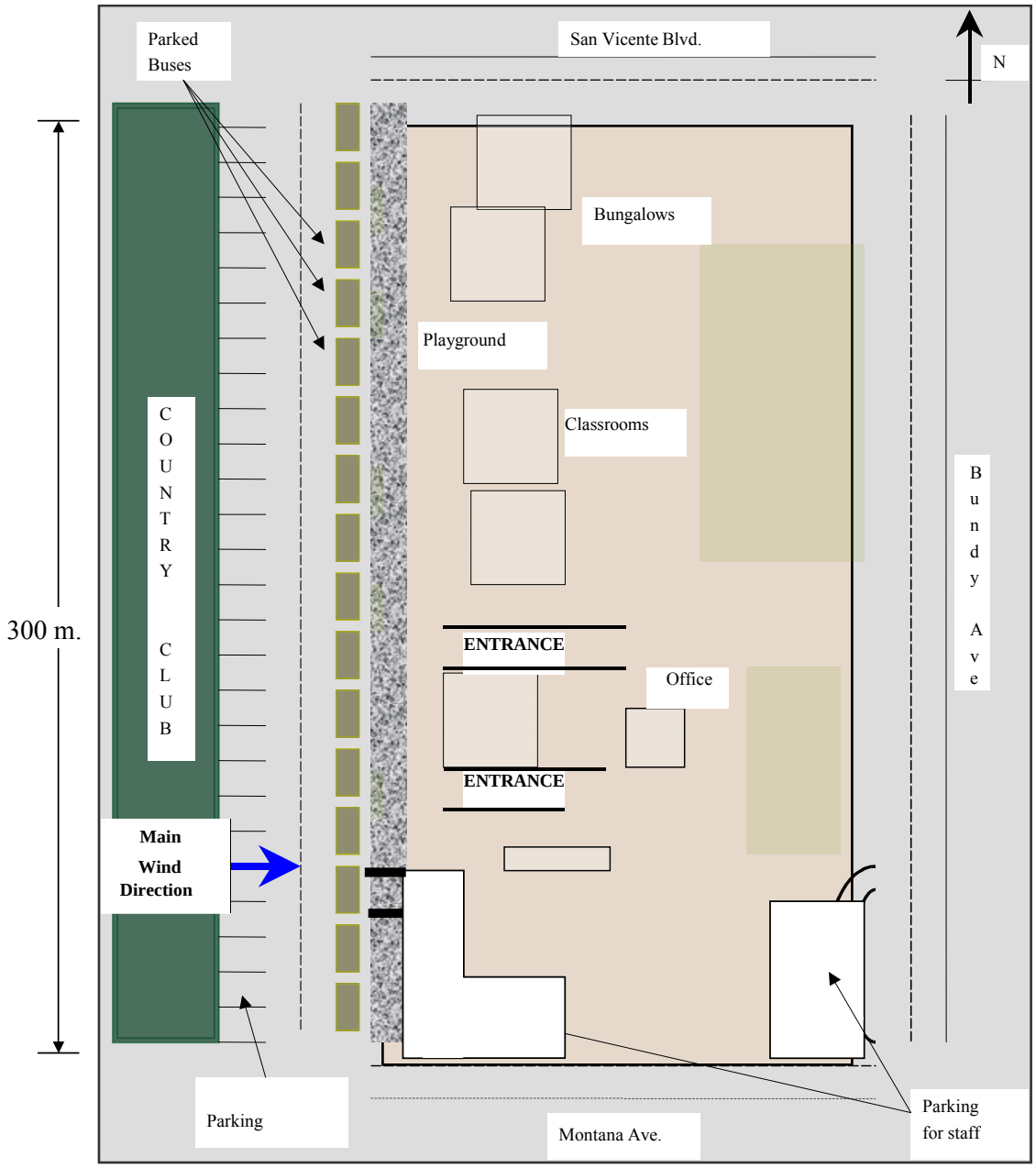


Figure 4.1.4.2 Diagram of Brentwood Science Magnet School.

from the buses onto the sidewalk in front of the school, where they waited until all children had unloaded from that bus. The children then walked as a group into the school through two entrance ways. Buses arrived at different times, so often children were waiting on the sidewalk when another bus pulled up to the curb. In general, morning commutes were conducted with bus windows up, in part due to cooler morning temperatures.

Photograph 6. School bus pulling up at BSMS, at the end of a morning run, with children in the unloading zone.

In the afternoon, buses arrived and parked on Gretna Green Way, along the sidewalk in front of the school, between 13:00 and 14:30 (Photograph 7). Again, each bus turned off its engine as soon as it parked. In the afternoon, all buses had the windows pulled down about one quarter of the way. School ended at 14:45, and children quickly boarded a total of nineteen buses. Of these, four were "small" and fifteen were "full size." Typically, children were on the sidewalk no more than about five minutes before they boarded, but on occasion groups of children were on the sidewalk considerably longer. At about 15:00, the bus engines were turned on simultaneously and generally all buses pulled away from the curb by about 15:05 (See Photograph 8). In general, no engines were turned on before all buses were boarded. The great majority of the buses turned right, or east, on San Vicente Boulevard.

Photograph 7. Line of buses in front of BSMS before an afternoon run.

(a) (b)

Photograph 8. a) Departing from BSMS during an afternoon run, school bus ahead.
b) Driving behind a school bus on an arterial street.

During the loading period, parents driving passenger vehicles dropped off (AM) or picked up (PM) approximately two dozen students on Gretna Green Way. However, no children other than those taking the buses stood on the sidewalk in front of the school. A few children were picked up by parents who appeared to walk to the school. Observation of the departing buses in the afternoon revealed several buses often "caravanned" at least part of the way on San Vicente, Wilshire and the San Diego Freeway (Interstate 405), which can lead to high exposures in the "following" buses (Rodes et. al. 1998).

4.1.5 Characterization and Justification for the Selected Bus Routes

Three different bus routes were used in this study. The first two were both in-use bus routes that traveled from highly urbanized areas of south-central Los Angeles to west Los Angeles. The third was a route that was fifty percent rural and fifty percent suburban, and traveled from rural Riverside to Diamond Bar in Los Angeles County. The principle criteria used to select these routes were that they be representative of bus routes in Southern California and that they also be representative of the wide range of roadway types and traffic congestion scenarios encountered in the South Coast Air Basin.

4.1.5.1 Urban Route One

During our investigation of BSMS bus commutes, the Director of the Transportation Branch at LAUSD provided the complete route sheets for all buses serving the BSMS. There were 19 different bus routes associated with the BSMS and these sheets provided the exact route followed, including the timing and duration of the stops, and the number of children picked up or dropped off at each stop. As can be seen in Figure 4.1.4.1 (triangles mark the starting locations of bus routes), students were bused from all parts of Los Angeles County to the BSMS, with bus routes covering extensive portions of South Central and East Los Angeles, Hollywood, the San

Fernando Valley, and reaching as far south as Carson. Combined, these bus routes provided comprehensive coverage of Los Angeles County west of the 710 Freeway.

In an effort to choose a bus route for the main study that provided an appropriate range of roadway types and traffic conditions representative of Los Angeles County commutes in particular, and dense urban traffic congestion in general, we investigated a selected number of BSMS bus routes. Four routes, including the route used during the pilot study, were evaluated. Since the pilot study route originated in south central Los Angeles, we selected one comparison route originating in north Los Angeles County (i.e. San Fernando Valley, Route No. 3222) and one originating in the south (i.e. Carson/Long Beach area, Route No. 3235). This was done to assess potential differences in route characteristics due to geographic location. The third comparison route selected (Route No. 3202) was one originating in an area closer to the pilot study route, to account for potential differences in route characteristics over similar geographic areas. Each of these three routes incorporated both freeway and surface streets, and included a variety of traffic congestion scenarios. Surface streets along each route were categorized into residential, minor arterial, and major arterials using Caltrans annual average daily traffic count data from 1998 and 1999.

Table 4.1.5.1 shows the comparison of these routes, and includes the percentage (in distance) for each of the roadway types listed above, as well as the percentage of freeway versus surface streets. As can be seen from this table, Route No. 3215, the route used in the pilot study, had a similar mix of roadway types and traffic conditions compared with the other three routes.

Table 4.1.5.1 Comparison of the distribution of roadway types on selected BSMS bus routes.

		Percent of Bus Route Distance			
		Route No. 3215 (Pilot Study Route)	Route No. 3202	Route No. 3222	Route No. 3235
Traffic Counts (Annual Average Daily Traffic)	< 5,000	10%	8%	7%	9%
	5,000 - 25,000	18%	20%	42%	27%
	25,001 - 100,000	34%	38%	28%	44%
	> 100,000	38%	33%	22%	20%
Roadway Type	Freeway	38%	35%	22%	26%
	Surface Streets	62%	65%	78%	74%

[1] For a definition of roadway type by traffic count see Table 4.1.5.2.

In addition, Table 4.1.5.2 shows the amount of time spent on each roadway type for Bus Route 3215. Approximately 25% of the time during this bus commute was spent on each of the four major roadway types: residential, minor arterial, major arterial, freeway. This table clearly demonstrates Route No. 3215 provided a sufficient distribution of roadway types and traffic conditions to explore the effects of these factors on children's exposure during bus commutes.

Moreover, based on the comparison in Table 4.1.5.1, we concluded this route contained roadway types and traffic conditions that were comparable to bus commutes in other parts of Los Angeles.

Our rationale for focusing primarily on a single route during the main study was to allow us to more unambiguously investigate the importance of other variables related to pollutant exposures during bus commutes, without the inevitable complications and confounders introduced by using more than one route. Specifically, we wanted to elucidate the effects of key variables such as fuel type and emission control status of the bus, hence it was important to hold constant the nature of the route. Otherwise, potential differences in traffic density and other route-related variables, even if not large in most cases, may have unnecessarily complicated and even confounded comparisons between fuel type and vehicle emission status. It is axiomatic in any scientific investigation that one would like to control all but one variable at a time in order to elucidate the effects of the remaining changed variable (e.g. fuel type). It was challenging enough to establish the effects and magnitudes of changing bus emissions and fuel types without the complication of using many different routes.

Table 4.1.5.2 Distribution of roadway types of BSMS bus route No. 3215.

		Percent of Time During Bus Run
Annual Average Daily Traffic	Roadway Type	Route No. 3215 (Pilot Study Route)
< 5,000	Residential	20%
5,000 - 25,000	Minor Arterial	20%
25,001 - 100,000	Major Arterial	30%
> 100,000	Freeways	25%
	Freeway	25%
	Surface Streets	75%

The broader issue concerning route choice and number of routes was one of how "representative" were the resulting data. This is an issue inherent in any vehicle-related study of this kind. In general, there are never sufficient resources (e.g. individual experimental runs) to conduct an investigation that is truly representative of the enormous variability in, for example, vehicle type and emissions, or roadway type and traffic characteristics. This is especially true for a region like southern California with more than 10 million vehicles and a geographic area of more than 10,000 square kilometers. Under these constraints the study design necessarily involved severe tradeoffs and conflicting or competing objectives. The present school bus study was no exception, given that only a fixed number of runs were possible within the funding, staff and equipment resources available. For every new run added, some other experiment had to be eliminated. Thus, the number of routes investigated could not have been expanded greatly without sacrificing some other important aspect of the study. Moreover, we felt there were

significant scientific advantages in leveraging the pilot study data by continuing to use the same bus route in the main study.

For all of the reasons given above, Route No. 3215 (the same route used in the pilot study) was selected as most appropriate for the primary focus of the main study. A map of this route and its bus stops is presented in Figure 4.1.5.1. The route was driven with the fully instrumented bus from the fifth stop to BSMS in the morning, and the reverse route in the afternoon. This somewhat shortened route relative to the pilot study was 18 miles long and provided approximately one hour of commute time. The percentage of time spent on the freeways increased to approximately 40% of the total run time with the truncated route. This route involved a wide variety of traffic conditions and roadway types, ranging from single lane residential streets with little or no traffic, to heavily congested, multi-lane surface streets which had high traffic densities during rush hour. Additionally, portions of this route traveled on the two most heavily congested freeways in the United States (I-405 and I-10) during rush hour traffic. The vehicle mix on these freeways, observed during the pilot and main studies, included a high percentage of medium and heavy duty diesel vehicles, including trucks and other buses. Particularly during the morning commutes, the percentage of diesel vehicles on the freeways was high compared with surface streets.

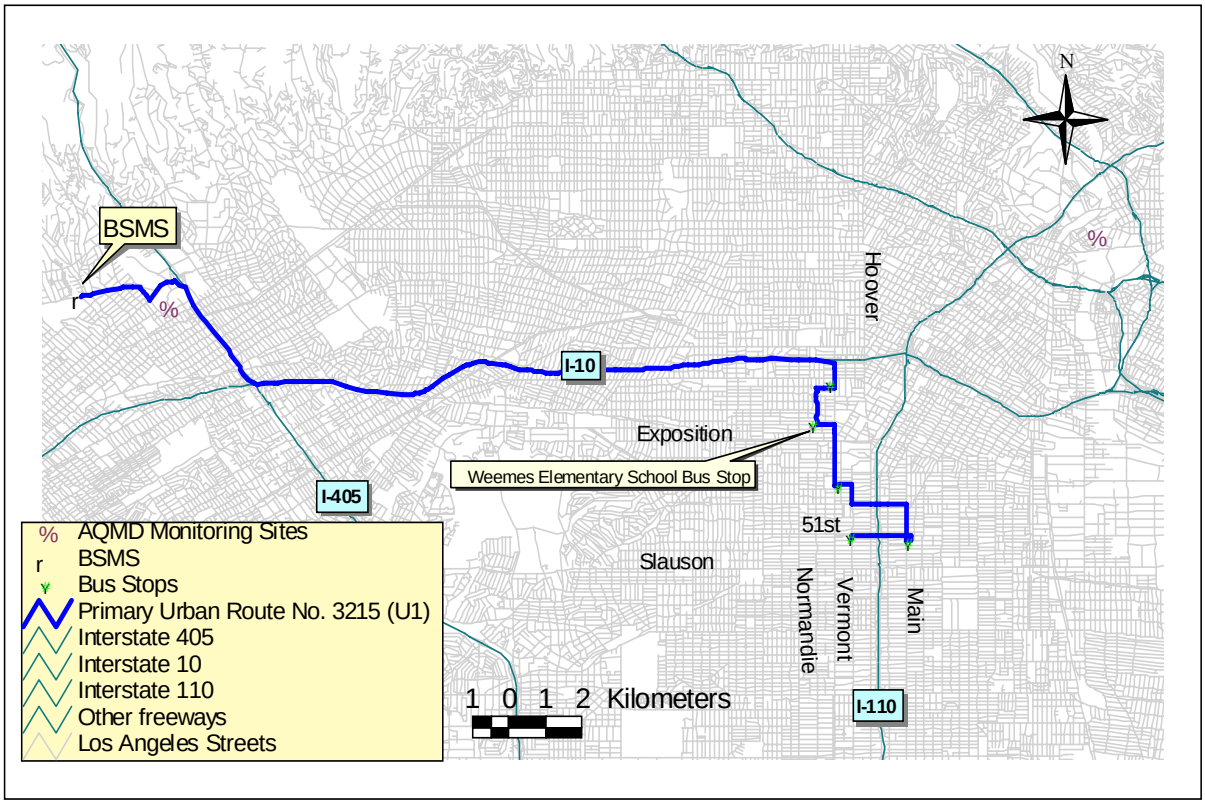


Figure 4.1.5.1 Primary urban bus route No. 3215.

The variety of neighborhoods along the primary route, from the west side to south central Los Angeles, also provided a varied mix of vehicle types, ranging from predominantly passenger vehicles and light trucks close to Brentwood, to older cars and more transit and school buses in

south central Los Angeles. In addition, this route traveled through inner city neighborhoods, making stops in locations we judged to have a racial/ethnic composition of approximately 80% Hispanic, 15% African American, and 5% Asian or other, thus having had relevance to concerns about environmental justice.

Measurements were made on the primary route during twenty bus runs in April, May and June 2002, consisting of ten morning and ten afternoon commutes. Each morning run started at 6:35 at the fifth pick-up location, and ended at approximately 7:40 at BSMS. During the afternoon runs, the bus left BSMS at about 15:05 and reached the fifth drop-off location at approximately 16:10. At each bus stop, the bus pulled up to the curb, opened the doors and waited for one minute before driving away, to simulate the conditions of children loading or unloading from the bus (Photograph 9). All the windows on the bus were closed during the morning runs, while during the afternoon runs, approximately half the windows were partially opened, to simulate conditions observed on in-use school buses.

4.1.5.2 Urban Route Two

While we determined the route used for the pilot study was appropriate for the great majority of the main study bus runs, we also believed it would be interesting to investigate a contrast between the primary route and another urban route. Based on our evaluation of the BSMS bus routes, we selected a second urban route for this purpose. We concluded that because the primary urban route extensively covered an appropriate range of urban driving scenarios, including travel on the two most heavily congested freeways in the U.S. and through heavily congested inner-city neighborhoods, an interesting contrast would be an urban route which covered a wider geographic area, traveled exclusively on surface streets, and included more industrialized areas of the city.

Photograph 9. Another school bus passing our idling bus at a bus stop during a morning run.

Although the majority of the bus routes from the BSMS travel on freeways for approximately 20 to 40 percent of their total distance, as discussed earlier, we identified a route which only travels on surface streets for approximately one hour of its commute time, while also covering a different geographic area than the route selected as the primary urban route. The route selected was the BSMS Route No.3235 (Figure 4.1.5.2). In the morning, this route started in Carson, just north of Long Beach, and traveled through industrial areas with a high percentage of heavy duty diesel traffic. The route continued north through neighborhoods of Rosewood, Gardena, Hawthorne, Inglewood and Hyde Park, and included middle to lower middle income neighborhoods with well maintained medium density housing, to higher density economically-disadvantaged neighborhoods. This portion of the route included 10 bus stops and, as noted, traveled exclusively on surface streets. After the final bus stop, the BSMS bus continued on to the Brentwood School. However, during the main study, we truncated the route at the final stop because this provided us with over one hour of commute time (comparable to the primary route) through a variety of neighborhoods and traffic densities, and included only travel on surface streets. We observed the racial/ethnic composition of the neighborhoods covered by this route to be different from the primary route, consisting of approximately 80 – 90% African American, with the remaining portion primarily Hispanic. The combination of the diverse neighborhoods covered, absence of freeway travel, and a potentially different mix of congestion scenarios on this route compared with Route No. 3215 allowed for interesting contrasts between bus routes for the main study.

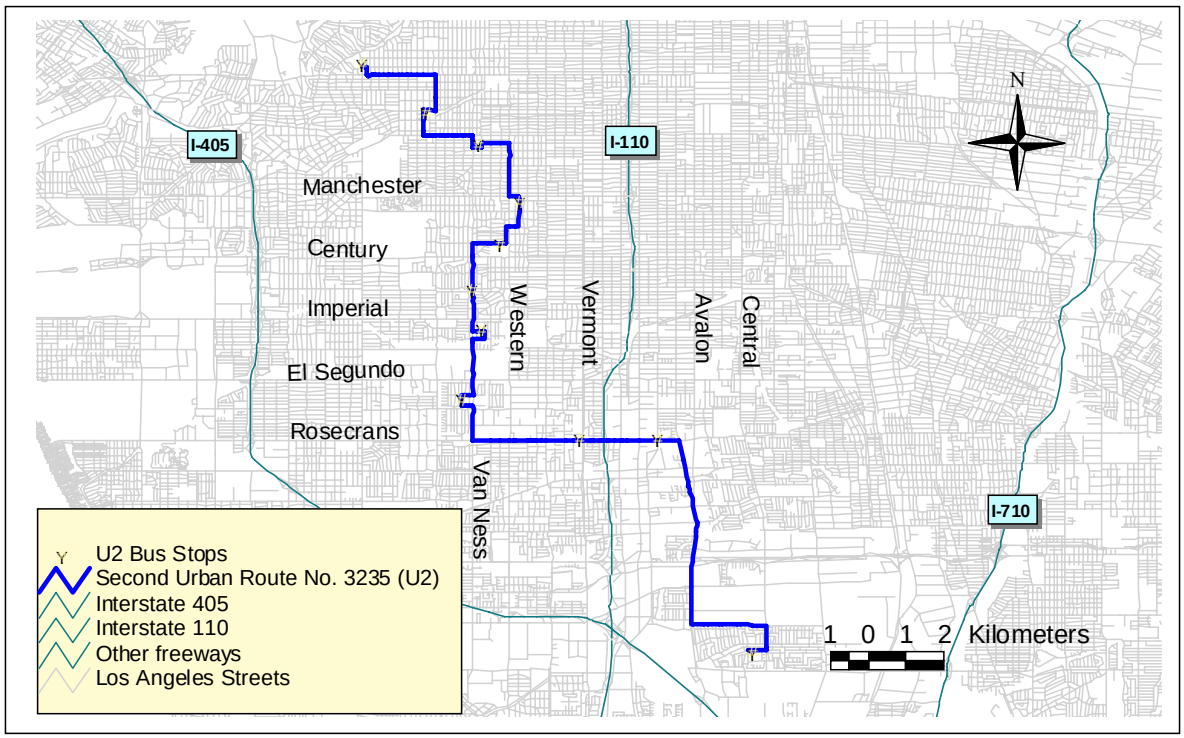


Figure 4.1.5.2 Second urban bus route No. 3235.

Four exposure test runs were done on the second urban route, two in the morning and two in the afternoon on May 29 and 30, 2002. The morning runs started at approximately 6:15 at the first stop in Carson, and finished at approximately 7:30 at the final stop. In the afternoon, we drove the route in reverse, leaving from the morning's final stop at approximately 15:30 and arriving at the Carson bus stop at approximately 17:00. All conditions on the bus during the runs on the second urban route were identical to those on the primary route, including window position, the use of run logs to characterize the route, and pollutants measured.

4.1.5.3 Rural/Suburban Route

In addition to the primary and secondary urban routes, which covered a wide range of urban driving conditions, we felt it was important to evaluate the concentrations inside school buses driving in low density rural and suburban neighborhoods. We expected these driving conditions would provide us with a number of scenarios not encountered in urban areas, including driving for longer periods of time at a constant speed and reduced idling time, and with constant conditions (e.g. less rapidly changing composition of surrounding vehicles). In addition, we expected lower roadway concentrations of traffic-related pollutants on the rural/suburban route due to greatly reduced traffic congestion would provide additional information for our comparison between different bus engine technologies and fuel types, and the influence on concentrations inside the bus.

The criteria for the selection of the rural and suburban routes were areas of little or no heavy duty truck traffic, no freeways within one mile of the route and lower housing and population densities compared to the primary urban route. The bus routes from the BSMS were not adequate to meet our needs for low-density driving conditions because they cover areas of Los Angeles County which are primarily high density, urban neighborhoods. Even the more suburban routes in the San Fernando Valley to and from the BSMS traveled adjacent to major traffic corridors (such as Interstate 405 and Interstate 5) and included heavily congested arterials (e.g. Ventura Blvd) during portions of their commute.

However, because the buses were outfitted with our sampling equipment at CE-CERT in Riverside before being brought to UCLA for the urban bus runs, we took advantage of the low density rural and suburban areas close to Riverside to obtain the appropriate rural/suburban route. The route selected (Figure 4.1.5.3) provided us with about one hour of commute time and traveled first through predominantly rural areas, and continued on to low density suburban areas. The selected route started on Mission Boulevard in Riverside and traveled through rural parts of Riverside, Jurupa, Mira Loma and Chino, with approximately two-thirds of the route passing by dairy farms and through undeveloped areas (see Photograph 10). This section of the route included predominantly light traffic (split evenly between trucks and passenger cars) on roads with two lanes in each direction. The route became more suburban as it turned north through Chino Hills and Diamond Bar, with areas of residential housing and small businesses (e.g., gas stations and strip malls).

Traffic conditions were heavier in the suburban portion of the route, but included very few heavy duty trucks. The route ended at the 60 Freeway in Diamond Bar.

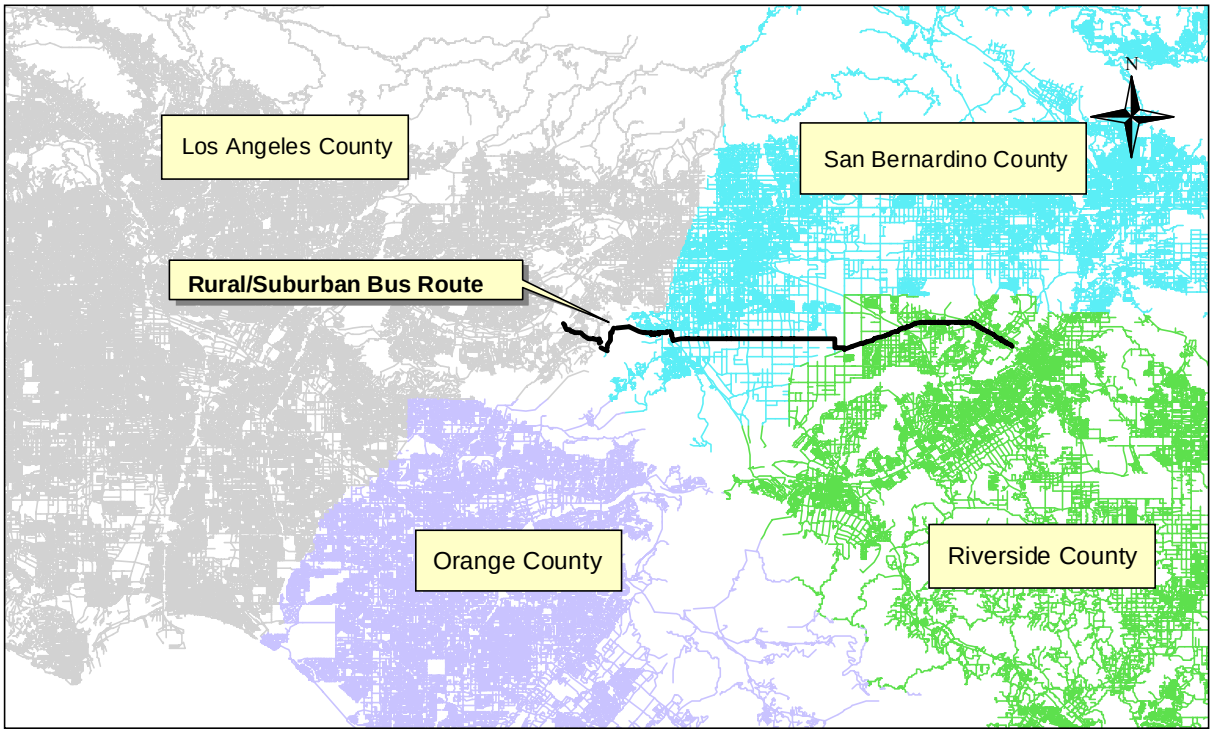


Figure 4.1.5.3 Rural/suburban bus route.

(a) (b)

Photograph 10. Rural (a) and suburban (b) segments of the rural/suburban route.

Due to logistical considerations, the exposure test runs on the rural/suburban route were conducted in the afternoon. One run on the rural/suburban route was conducted for each bus evaluated in the main study, generally one day prior to the test runs on the primary urban route for the same bus. A total of seven rural/suburban exposure test runs were completed, one during each week of the main study beginning the week of April 22, and ending the week of June 11 (not including the week of May 29). The start times for the rural/suburban test runs varied from approximately 11:30 to 15:00, although most started after 12:30. For each test on the rural/suburban route, approximately half the windows on the bus were partially opened. The video camera was operated on three of the seven rural test runs and notes about traffic events during the rural/suburban runs were maintained in a notebook by the field technician.

4.1.6 Characterization and Justification for the Selected Bus Stop

Concentrations observed in the pilot study during bus commutes were generally significantly higher than those at the Vermont Elementary School bus stop (the first stop along the pilot study bus route) or at the loading/unloading zone at the BSMS. Based on this finding, in combination with our observations children typically spent only short periods of time waiting at bus stops or in loading/unloading zones, we recommended the main study focus primarily on bus commutes. We determined, however, that a limited number of additional measurements at a second bus stop during the main study would allow us to better define potential exposures associated with this microenvironment. These additional runs were recommended in part because a fairly high-emitting bus was involved in the pilot study. We felt a lower-emitting bus, of the kinds used in the main study, might exhibit lower average exposure ratios, for the commute versus the bus stop, leading to a greater relative contribution from bus stops to total exposure.

Since this study was designed to include range-finding for high-end exposure conditions, the second bus stop along the primary route was selected for factors such as high adjacent traffic, long bus idle time, and where children were waiting for long periods of time to be picked up by the bus. By observation we determined which stops on the inner city portion of the primary route involved these high exposure scenarios. By following several buses over different routes on several different days we determined the range of relevant variables, such as the time spent by the bus at a given stop and whether the bus idled or turned off its engine. Logistical considerations such as site security and positioning of the monitoring equipment were also factors included in selecting the bus stop location.

Based on these criteria, the second bus stop selected along the primary route was the Weemes Elementary School, located on 36th Place, three blocks south of Vermont Avenue near downtown Los Angeles. Similar to the first bus stop measured during the pilot study, this stop was characterized by increasing traffic congestion during peak periods, and served as a bus stop for several bus routes. Diesel buses arrived to pick up or drop off children frequently during the period just before this school started and after it let out. The average time spent by a bus at stops was 60 seconds, with the engine of the bus remaining on after the bus pulled up to the curb and children loaded/unloaded. In addition, this stop had an appropriate place to set up the monitoring equipment, and was located in a relatively secure area.

Measurements at the bus stop were made during one morning and one afternoon on May 23, 2002. Each measurement period started at least 30 minutes before the expected arrival time of the BSMS bus, and continued for at least 30 minutes after that time. The fully instrumented bus was parked (engine off) next to the sidewalk on the street in front of the school. Other buses and passenger vehicles pulled in front of or behind the instrumented bus to drop off/pick-up children at the school.

4.1.7 Characterization of Window Position Test

The objective of the window position tests was to establish the effect of window position on concentrations inside the bus cabin. Two sets of real time instruments were used, one set sampled inside the bus and the other sampled immediately outside the bus. During window position tests, we alternated between open and closed windows for designated intervals. However, during all other bus commutes, the position of the windows was fixed, with the windows closed during all morning commutes, and the windows open during all afternoon commutes.

We selected the I-405 (north/south direction) between Wilshire and Century Boulevards as the route for the window position tests. Although there could have been ambient air pollutant concentration gradients along this north/south route, we expected them to be smaller than those observed for east/west routes. Specifically, during the pilot study we observed background concentrations tended to increase as the bus approached the inner city (heading east) and decreased approaching the Westside and Brentwood (heading west), likely due to changes in traffic density and composition, as well as meteorological effects. The I-405 runs approximately north/south, with more uniform traffic densities and meteorological conditions, therefore, we anticipated relatively minor ambient air pollutant concentration gradients and measurements taken outside the bus during the test allowed us to characterize any concentration gradients along the route.

Typically, we began a window position test on I-405 at Wilshire Boulevard traveling south to Century Boulevard, which took approximately 20 minutes, followed by the more congested return back up the I-405 north for about 40 minutes, for a total sampling time of one hour. During the test we opened the windows for eight minutes, then closed the windows for eight minutes, alternating between the two positions every eight minutes for the duration of the test. This amount of time was short enough to allow several tests under similar traffic conditions (i.e., north and south directions) but long enough to allow stabilization of conditions inside the bus. During open window times, on the driver's side of the bus, every other window was opened about ten inches, while on the other side of the bus, where the instruments were located, every other window was opened only two inches.

The window position tests were conducted immediately following the morning bus commutes on May 8, and June 5 and 6, 2002, between 8:15 and 9:15. Different average speeds along the route allowed us to test the effect of window position for different ventilation rates inside the bus cabin.

4.2 Data Analysis Methods

The data collected during this study can be categorized as continuous, non-stationary and deterministic. We used time-series analysis techniques as well as conventional statistical procedures to analyze the data set. This section presents a detailed explanation of all procedures used to analyze the main study data.

4.2.1 Goals of Time Series Analysis

Time series analysis can be used to accomplish two different goals: identification of the nature of the phenomenon represented by the sequence, and the forecasting or prediction of future values of the time series variable. In this study, we focused on the first objective.

The presence of stochastic events that modified the general pattern occurred constantly in our time series (e.g., encounters with other HD diesel vehicles; random stops). In addition, the data did not exhibit deterministic cyclic patterns. Finally, the concentrations measured by the instruments were due to the contribution of "background pollution" concentrations as well as primary emissions from vehicles.

Cross correlation (establishment of relationships between different time series and their implications) was used to establish precision between paired instruments measuring the same pollutant simultaneously and to identify correlations between different pollutants (Section 5.1.3).

4.2.2 Data Analysis Techniques

To provide a measure of the importance of a given microenvironment in terms of exposure, we calculated exposure factors for school bus commutes and bus stops and loading/unloading zone (from the pilot study). An exposure factor was defined as the product of the average concentration for a specific pollutant multiplied by the time spent by children in that microenvironment.

To determine the effects of fixed variables on concentrations inside the bus during each run, parametric and nonparametric statistical tools and graphical methods were used to identify variables governing exposure during bus commutes (Section 5.3). The data from different runs were grouped into categories by variable types. The objectives were to understand the contrasts defined by the study's experimental design as well as the relative importance of the variables (e.g. type of bus, type of route, windows position, time of day, position within the cabin, type of roadway). Basic descriptive statistics, such as the median and interquartile ranges, and the 95% confidence intervals were calculated for the grouped data. Graphical representations were used to describe the contrast between data sets, including cumulative frequency distributions and boxplots.

Tests for autocorrelation were applied, to assure the validity of subsequent statistical procedures. Since the concentration at a particular time is dependent on the concentration a period of time before, the consideration of autocorrelation was mandatory in the data analysis. The time-lag (k) within which there is correlation was established for several data sets. Correlograms (a plot of the autocorrelation coefficient (r_k) in terms of the lag (k)) provided a visual inspection of a range of correlation coefficients at relevant time lags so that significant values were identified. An r_k value of $+-2/(N)^{1/2}$ denotes a significant difference from zero and

signifies the presence of autocorrelation (Norusis, 2002). Note also that as k becomes larger, r_k becomes smaller.

Finally, we included more detailed analysis for a limited number of bus runs, which involved identification of events influencing real-time concentrations inside the bus during a run. This effort included the most comprehensive analysis and was performed for a selected set of runs. This included a description of events (including a picture gallery); an explanation/correlation of peak pollutant concentrations inside the bus with events (e.g. idling or driving behind another diesel bus) using video-camera records and activity logs; analysis of meteorological data and their effects on the selected runs; and compilation of selected video clips to demonstrate major events during runs.

Our goal for this analysis was to be as methodical and consistent as possible. We developed a specific protocol for identification of peak concentrations, as well as their correlation with external events recorded on the video-camera and/or the activity log. This formality was necessary as the assessment and interpretation of data can be affected by the reporting practices (Henry et al., 2000).

4.2.3 Video Record Analysis

An 8mm high-resolution video camera was mounted at the front of the test bus to record traffic conditions and other exposure-related events occurring in front of and adjacent to the test bus during most of the measurement periods. All video camera records (created for 28 out of 36 runs) were digitized into MPEG format, which is considered state-of-the-art for long-term preservation and future migration to new formats. Selected videotape records were carefully examined to relate peaks in pollutant concentrations with traffic conditions, type of vehicle being followed, and other variables that might affect the concentrations inside the cabin of the bus.

4.2.3.1 Definitions

In principle, a peak occurs when the concentration time series changes behavior during a short period, exhibiting prominent high values compared with the baseline. This is not the same as an outlier, which is defined as measurements that are extremely large or small relative to the remaining data and, therefore, are suspected of misrepresenting the population from which they were collected (U.S. EPA, 2000). See Figure 4.2.3.1 for a schematic representation of a peak, an outlier, and the baseline.

4.2.3.2 Data Generated from Videotapes

Two basic approaches can be taken when linking time series concentrations with events recorded by the video camera. The first method includes a formal definition of relevant concentration behavior in a time series with which one can try to define what constitutes a peak using a formal and systematic procedure and attempt to explain such a peak by association with an event. For example, the use of time-series intervention analysis (McDowall et al., 1980) mathematically determines if an event has an impact on a time series. This method uses hypothesis-testing in conjunction with ARIMA models (Box and Jenkins, 1976; Box and Tiao, 1975).

The second approach, which we adopted in our analysis, focused on identifying all conditions throughout a run, instead of focusing only on periods of peak concentrations. This was necessary since a single peak may be generated by several events, and one event may cause a signal that lasts for more than a few seconds. These concepts are illustrated in Figure 4.2.3.2. This time series corresponds to the PAH concentrations measured in the rear of the bus during

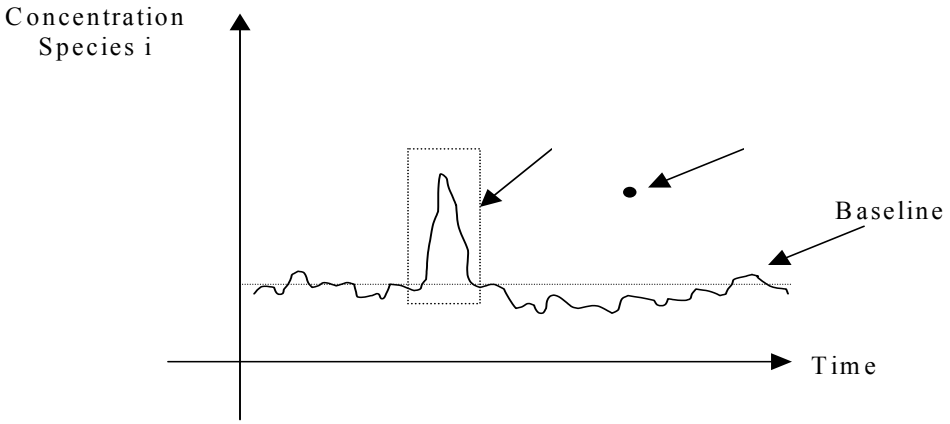


Figure 4.2.3.1 Example of a peak, an outlier and the baseline.

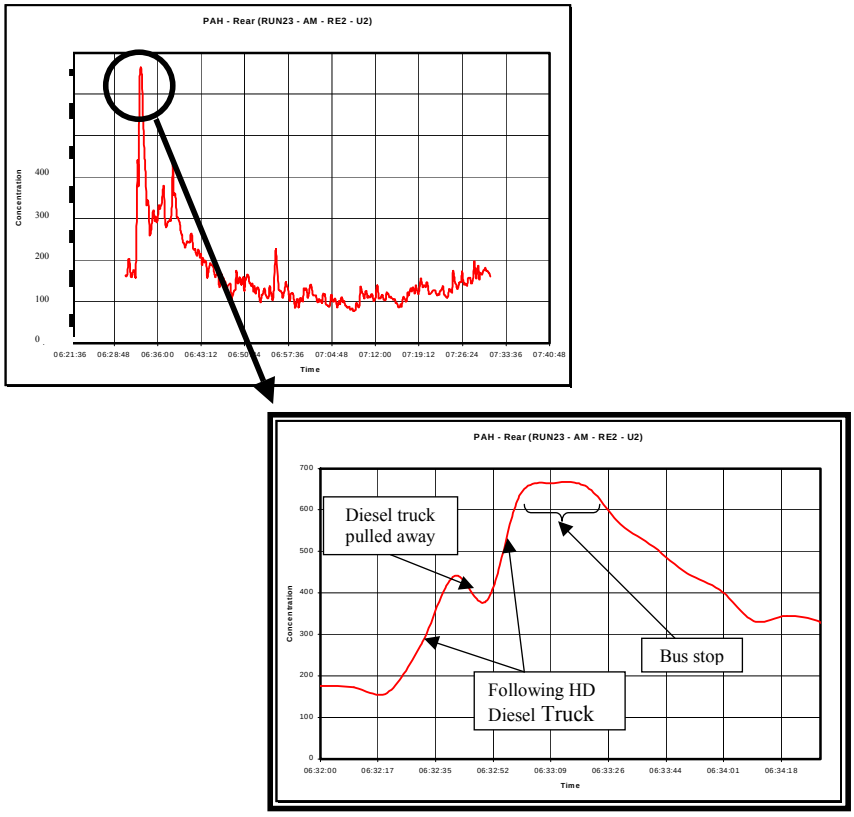


Figure 4.2.3.2 PAH time series for Run 23. Lower panel shows circled peak expanded and resolved into two peaks.

Run 23 where a major peak can be observed at the beginning of the run. After analyzing the events occurring during this peak, we were able to divide the peak into three distinct regions. The first region showed a monotonous increasing concentration truncated at about 6:32:40, followed by a decrease for several tens of seconds, and was associated with the presence of a heavy duty diesel truck. We started following this vehicle at 6:32:00 and continued following it until 6:32:40 (increasing trend) where a red light allowed the truck to pull away from us (truncated signal). The second region included another increasing concentration that reached a maximum value at about 6:33:00. During this period, we were again following a heavy duty diesel truck. The third region showed a flat behavior that lasted for about thirty seconds. At 6:33:00 we stopped at a school bus stop where we stayed for about one minute (flat line). This example illustrates how multiple sources can contribute to a single peak concentrations. Therefore, to properly correlate and explain peak concentrations with events we attempted to describe all relevant events and characteristics during a run using the protocol described below.

4.2.3.3 Videotape Analysis Protocol

The goal of our videotape analysis was to gain a better understanding of the factors that affect pollutant concentrations inside the bus, such as the effect of surrounding vehicles in combination with the contribution from the bus's own exhaust for different fuel types and engine after-treatment technology. Based on the experience obtained during the pilot study and the results of the initial analysis of the videotapes from the main study, a set of events or characteristics with the potential to influence in-vehicle concentrations was defined. These events or characteristics were the basis for the final videotape analysis during the main study, and included the following:

1. Presence of any diesel-powered vehicle in front of or adjacent to the test bus. Only those vehicles within approximately three car lengths (based on a qualitative assessment of distance) were considered.
2. Exhaust location of the vehicle being followed (high, low).
3. Presence of visible emissions or smell of exhaust from the vehicle being followed, or from the test bus.
4. Idling at red lights, stop signs, in traffic or at bus stops.
5. Level of traffic congestion (heavy, moderate, light).
6. Roadway type (freeway, major/minor arterial, small residential streets).
7. The direction the test bus was facing (e.g. north, northeast, etc.).

As discussed previously, we used the ten-second concentration data (time resolution established for the data acquisition system in the field) for each pollutant inside the test bus during each run for the following analyses. Using primarily the videotaped record of an exposure run, which showed the view of the road in front of the test bus, we systematically identified the events and characteristics (described above) that occurred in the area surrounding the test bus during each ten second period for a selected number of runs. We also used the activity logs created during the runs to supplement the video record. The use of these logs was necessary since important events were not always easily visible, in part due to the quality of the digitized videotape records (e.g., smoke coming out of an exhaust pipe), but were noted by the

field technician during the run and recorded in the run logs. In addition, there were events that could not be identified by the videotape records (e.g., diesel exhaust odors).

Every ten-second time interval during a given run was defined by the events/characteristics that occurred during the interval. An event/characteristic (based on the criteria noted above) was assigned to a ten-second time interval if the event/characteristic occurred for at least five seconds during that interval (50% of the gap). If it continued for more than ten seconds, the event was assigned to all intervals during which it occurred. For example, if a diesel truck was followed for 30 seconds, this event was assigned to three time intervals. It was also possible for more than one event to occur during a given ten-second interval. If the events were different, multiple assignments were made. For example, if the test bus was idling behind another school bus at a traffic light for one interval, both following a diesel vehicle and idling were assigned to that time interval. However, if two different diesel vehicles were in front of the test bus within three car lengths, only the closest one was assigned to the interval.

An assignment of the type of vehicle being followed was made for every ten-second interval during a run. Vehicles were classified into one of five vehicle types: heavy-duty diesel truck (3-5 axles), medium duty diesel truck (2 axles), diesel school bus, diesel transit bus and passenger car/no target (Fruin, 2003). The location of the exhaust pipe on all diesel vehicles and the presence of visible emissions or odors was noted when possible.

Because of the subjective nature of the videotape analysis, every effort was made to assure consistent, repeatable and accurate assignments of events were made. The distance to the vehicle in front was the most subjective portion of the analysis. Vehicles more than approximately three car lengths (about sixty feet) away from the test bus were not included in the event assignments. In order to validate this criterion, each run was analyzed by two different individuals, and any discrepancies between the results were carefully scrutinized before a final assignment was made to assure consistency. In addition, assignments of idling were validated using the bus's speed (also measured in real-time during every run). Finally, to assure unbiased assignment of events/characteristics based solely on the criteria described above, pollutant concentrations were not considered until after the videotapes had been reviewed and all event assignments made.

4.2.3.4 Linking Events/Characteristics and Concentrations

The final step was to identify significant concentration differences between time periods associated with each type of event or characteristic. For example, we were interested in describing any difference found between concentrations in the bus when idling, when traveling behind another diesel vehicle, between vehicles followed which had high or low exhaust, or when not following any diesel vehicles. We combined all the time periods with a given event/characteristic assignment, such as when following a diesel school bus, or behind a heavy duty diesel vehicle, and made comparisons of pollutant concentrations associated with those events. In addition, we produced graphs similar to Figure 4.2.3.2, representing the time series for a given pollutant and a set of explanatory textboxes indicating the type of events associated with the real-time concentration behavior.

5.0 RESULTS AND DISCUSSION

Our discussion of the results from this study begins with a brief assessment of our measurement methods. This involved both an assessment of the (a) accuracy of our instruments, as determined by external audits or best estimates for those instruments for which known standards were not available; and (b) an assessment of precision, based on measurements from paired instruments.

Although these measurement method evaluations were important, the primary approach to data analysis used in this study, and upon which the majority of our important findings were based, relied on the relative concentration differences measured during different exposure runs using the same set of instruments. Specifically, all the measurements made during the main study that were used to evaluate the importance of test bus type, roadway type, window position, surrounding traffic and events, and the bus stop vs. commute microenvironments, were made with the same set of instruments. These comparisons relied on the assumption the same instrument operated in the same manner during the course of the eight weeks of the main study.

In addition, a limited number of experiments were done in which measurements were taken using paired instruments, including the comparison of the front vs. the back of the bus, and the inside vs. the outside of the bus. The results of our precision estimates were important for these comparisons, and only those instrument pairs with adequate precision (as described in Section 5.1.3) were used for these comparisons.

It is important to recognize in interpreting results presented in this chapter that, as shown in Table 5.1, with one exception (HE3), all front/rear contrasts were conducted only for the conventional diesel buses (Runs 1 to 20 and 23 to 26), and that all inside/outside contrasts were made only for the trap-outfitted and CNG buses (Runs 27 to 36).

As mentioned before, throughout this report, PM2.5 data are from the Harvard Impactor integrated sampler unless stated otherwise (i.e., data from the DustTrak instrument)

5.1 Assessment of Measurement Methods

5.1.1 Overview

In this study an important objective was to compare measurements made with pairs of instruments for each pollutant for contrasts such as inside/outside and front/rear of the test bus.

To evaluate such comparisons, the statistical variability of the measurements under collocated sampling conditions was needed to determine the probability at a stated significance level, that the paired measurements were significantly different. Thus, determination of collocation comparisons was the focus rather than establishing detection limits per se. A comprehensive discussion and analysis of collocated sampling data is given in Section 5.1.3.

5.1.2 Instrument Accuracy

Instrument accuracy is normally defined by comparisons to known standards, preferably traceable to the National Institute of Standards and Technology (NIST). Accuracy is defined as absence of bias from the known standard, while for the purpose of this study was defined based

Table 5.1 Summary of main study exposure runs.

Run Number	Date	Time of Day	Type of Bus	Contrast	Type of Route
1	4/22/2002	Afternoon	HE1	Front/rear	Rural/suburban
2	4/23/2002	Morning	HE1	Front/rear	Urban 1
3	4/23/2002	Afternoon	HE1	Front/rear	Urban 1
4	4/30/2002	Afternoon	HE2	Front/rear	Rural/suburban
5	5/1/2002	Morning	HE2	Front/rear	Urban 1
6	5/1/2002	Afternoon	HE2	Front/rear	Urban 1
7	5/7/2002	Afternoon	HE3	Inside/outside	Rural/suburban
8	5/8/2002	Morning	HE3	Inside/outside	Urban 1
9	5/8/2002	Morning	HE3	Inside/outside	Window position test
10	5/8/2002	Afternoon	HE3	Inside/outside	Urban 1
11	5/13/2002	Afternoon	RE1	Front/rear	Rural/suburban
12	5/14/2002	Morning	RE1	Front/rear	Urban 1
13	5/14/2002	Afternoon	RE1	Front/rear	Urban 1
14	5/16/2002	Morning	RE1	Front/rear	Urban 1
15	5/16/2002	Afternoon	RE1	Front/rear	Urban 1
16	5/20/2002	Afternoon	RE2	Front/rear	Rural/suburban
17	5/21/2002	Morning	RE2	Front/rear	Urban 1
18	5/21/2002	Afternoon	RE2	Front/rear	Urban 1
19	5/22/2002	Morning	RE2	Front/rear	Urban 1
20	5/22/2002	Afternoon	RE2	Front/rear	Urban 1
21	5/23/2002	Morning	N/A	N/A	Bus Stop
22	5/23/2002	Afternoon	N/A	N/A	Bus Stop
23	5/29/2002	Morning	RE2	Front/rear	Urban 2
24	5/29/2002	Afternoon	RE2	Front/rear	Urban 2
25	5/30/2002	Morning	RE2	Front/rear	Urban 2
26	5/30/2002	Afternoon	RE2	Front/rear	Urban 2
27	6/4/2002	Afternoon	TO1	Inside/outside	Rural/suburban
28	6/5/2002	Morning	TO1	Inside/outside	Urban 1
29	6/5/2002	Morning	TO1	Inside/outside	Window position test
30	6/5/2002	Afternoon	TO1	Inside/outside	Urban 1
31	6/6/2002	Morning	TO1	Inside/outside	Urban 1
32	6/6/2002	Morning	TO1	Inside/outside	Window position test
33	6/6/2002	Afternoon	TO1	Inside/outside	Urban 1
34	6/11/2002	Afternoon	CNG	Inside/outside	Rural/suburban
35	6/12/2002	Morning	CNG	Inside/outside	Urban 1
36	6/12/2002	Afternoon	CNG	Inside/outside	Urban 1

on the comparisons of measurements made with identical pairs of instruments (see Section 5.1.3). Poor precision can limit the assessment of accuracy, although a measurement can be very accurate even with poor instrument precision if enough measurements are made. For measurements of the kind conducted in the present bus study, accuracy is normally assessed

during an independent audit of the instrument performance where known standards were available. For many of the instruments used in this study such standards were not available or did not exist. The following section describes our best estimation of the accuracy of the measurements and the limitations involved.

5.1.2.1 Audited Instruments

A system and performance audit was performed May 10, 2002 following the procedures described in the project Quality Integrated Work Plan (QIWP) (Gemmill, 2002a). The audit was performed on most of the instruments which measured gaseous pollutants. The system audit included an inspection and evaluation of the overall monitoring system setup in a bus being prepared for use in the study. The accuracy of the CO, NO/NO_x, and NO_2 analyzers were assessed by this performance audit. The performance audit consisted of challenging the analyzers with a test atmosphere from an independent calibration source. The test gas concentrations entered each instrument through the same sampling system used during sample runs. After the instruments' responses had equilibrated, their outputs were read from the data logger, correction factors previously determined by the system operator were applied, and these corrected outputs were compared to the known test gas inputs. The following is a summary of the audit findings:

1. Continuous Analyzers: The audit results for the Thermal Electron Instruments (TEI) NO/NO_x and CO analyzers indicated the project accuracy data quality objectives (see QWIP, Gemmill, 2002a) were met for those instruments. However, the objectives for the UCR NO_2 analyzers were not met, probably due to the difficult moving platform environment in which they were operated. An audit was not performed on the SF_6 analyzer because it was not operational at the time of the audit.

2. System: The general system condition, operations, and maintenance were evaluated. A completed system evaluation report is presented in the Quality Assurance Audit Report which indicated that no significant unsatisfactory audit results were encountered (Gemmill, 2002b). The procedures followed in the execution of this project were consistent with those described in the QIWP (Gemmill, 2002a). Overall, the monitoring equipment was securely installed in the bus and was in satisfactory operational condition. All instrument calibrations were done in a proper manner and all documentation was complete, concise, and up to date.

3. Calibration Results: A review of the calibration results for the main study indicated the data quality objectives were generally met for the TEI Model 42 NO/NO_x analyzer, but were not met for the CE-CERT NO_2 and Dasibi 3003 CO analyzers (see QWIP—Gemmill, 2002b). Most of these data were considered valid.

Complete details of the audit, including procedures and findings are provided in a separate audit report (Gemmill, 2002b).

5.1.2.2 Unaudited Instruments

5.1.2.2.1 Particulate Matter Mass

5.1.2.2.1.1 Integrated PM_{10} and $PM_{2.5}$

The largest sources of error for these instruments were the measurement and control of the flow rates and filter handling. The flow rates were controlled with needle valves and measured with a rotameter. The accuracy of the flow maintenance was defined by the characterization of volume, which depends on both temperature and pressure. The flow was set to a specific volumetric rate since the inlet was designed to operate under those conditions. While temperature and pressure changed during the course of sampling, the specified volumetric flow setting for these devices may deviate was believed to stay within approximately 5-10% of true volumetric flow.

Minor sources of error included the size-cut characteristics of the PM inlet and weighing accuracy. Since evaluation of the size-cut characteristics were beyond the scope of this project, we relied on previous comparisons. The inlet design we employed has been used in many other exposure related studies; and provided results comparable with other research. It should be noted that any inertial size separation is not exact, and that designation of a size cut off is an operational definition. Specifically, the size cut of an inlet is defined as the size at which 50% of the mass of the specified aerodynamic size penetrates the inlet. The sharpness of the size cut (particles greater or less than the specified aerodynamic size will always penetrate the inlet) depends on the design of the inlet.

We believe the error in filter mass weighing was minor due to our weighing procedure. We used a class M calibration standard before and after every weighing session. The calibration standard was weighed to the nearest 0.1 μg, and little deviation was observed (typically +/- 0.1ug). It should be noted that in practice it is not possible to weigh filters to closer than 1 μg as the weight will fluctuate due to factors such as humidity much more than a metallic standard. All filters were weighed until three weighings agreed to within 3 μg. Since typical loadings were 50-100 μg, the error in the weighing process should not have exceeded a few percent. The errors due to filter handling are random and therefore related to the precision of the measurement.

5.1.2.2.1.2 Continuous PM_{10} and $PM_{2.5}$

The DustTrak instruments used for this measurement are based on light scattering of particles. The amount of light scattered is dependent not only on concentration, but also is highly dependent on the particle size distribution and composition. In order to relate the light scattering to a concentration it is necessary to choose a particle size distribution since these distributions are constantly changing in ambient air. The response of these instruments was calibrated by the manufacturer based on the particle size distribution of Arizona road dust, which was chosen because it is an NIST standard reference material. Thus the inherent accuracy can be related to an NIST standard, but the accuracy at the time measurements are made must be established by comparing the measurements with mass measurements from collected PM. The accuracy determination for the measurements made in this study was therefore limited to that of the integrated filter collection methods described above. In addition the accuracy of the instrument varies directly with the flow rate. A rotameter and needle valve were also used to adjust this flow and the estimated uncertainty introduced was five to ten percent as discussed

above. See the next section for details of a comparison of the DustTrak against a reference method.

5.1.2.2.1.3 Comparison of Measurement Methods for $PM_{2.5}$ and PM_{10}

The present study compared two instruments that measured the mass concentration of $PM_{2.5}$ and PM_{10}: the Thermo Systems Inc. Model 8520 DustTrak Aerosol Monitor (DustTrak) and the Harvard Impactor (HI). The DustTrak measured the PM concentration continuously in real-time while the Harvard Impactor used standard filter-based sampling methods. Continuous monitors for airborne particulate matter measurements have an advantage over traditional filter-based sampling methods that do not provide real-time data and cannot characterize short-term high concentrations (Babich et al., 2000; Chung et al., 2001). However, continuous monitors currently in use may suffer from problems of accuracy and appear to be more useful on a relative rather than absolute basis (Ramachandran et al., 2000; Chung et al., 2001; Moosmuller et al., 2001; Yanosky and Maclntosh 2001; Yanosky et al., 2002).

Several studies have been conducted comparing different $PM_{2.5}$ or PM_{10} samplers (Babich et al., 2000; Ramachandran et al., 2000; Chung et al., 2001; Yanosky and Maclntosh 2001; Yanosky et al., 2002). Babich et al. (2000) and Yanosky and Maclntosh (2001) compared $PM_{2.5}$ mass concentrations from the HI against U.S. Environmental Protection Agency-designated Federal Reference Method (FRM) $PM_{2.5}$ samplers. Yanosky et al. (2002) investigated the performance of the DustTrak measuring $PM_{2.5}$ concentrations of indoor air in comparison with the FRM. The gravimetric method, (i.e. FRM) was used as the standard for evaluation of the real-time continuous monitors because it has proven to be stable and accurate. In Ramachandran et al. (2000) DustTrak measurements of $PM_{2.5}$ concentrations indoors and outdoors were compared with gravimetric measurements (MSP, Inc. $PM_{2.5}$ inlet for indoors; Andersen RAAS2.5-300 sampler for outdoors). Chung et al. (2001) compared measurements of PM_{10} by the DustTrak with those by a size-selective inlet sampler (SSI).

All these studies employed simple linear regression analyses in order to calculate differences between sampler types. When the concentrations from a sampler, *y*, are regressed on those from a reference sampler, x, the intercept should be zero and the slope on *x* should be unity, i.e. $y = 0 + 1*x$, if the measurements between the two samplers coincide. If the intercept estimate is significantly different from zero statistically, there is a systematic bias between samplers. A statistically significant difference in slope suggests a proportional bias between samplers (Yanosky and Maclntosh 2001). The regression results of the above-mentioned studies are summarized in Table 5.1.2.1.

From Table 5.1.2.1, the first two studies showed excellent agreement between the HI and FRM methods, indicating the Harvard impactor could serve as a reference method for the DustTrak instruments employed in the present study. Three other studies showed the DustTrak overestimated $PM_{2.5}$ or PM_{10} mass concentration by factors of two to three (the slope column in Table 5.1.2.1) compared with filter-based methods, although no systematic biases were observed in these three studies. These findings are consistent with the fact that the DustTrak is a nephelometer, which measures mass concentration of an aerosol stream by converting the amount of light scattered from many particles at once. Thus, the DustTrak output depends on particle size distribution and refractive index (Moosmuller et al. 2001) and the response of the

instrument for an aerosol stream consisting mainly of finer particles can produce errors in concentration of a factor of two or more (Hinds 1999; Ramachandran et al., 2000; Chung et al., 2001).

Table 5.1.2.1 Summary of previous studies comparing samplers employed for measurements of $PM_{2.5}$ and PM_{10} mass concentrations.

Reference	PM size	Samplers	n	Intercept[1]	Slope	R^2
Babich et al. (2000)	$PM_{2.5}$	HI/FRM	81	1.36	0.97[2]	0.99
Yanosky and Maclntosh (2001)	$PM_{2.5}$	HI/FRM	21	-0.06	1.03[2]	0.96
Yanosky et al. (2002)	$PM_{2.5}$	DustTrak/FRM	17	-1.73	2.57	0.86
Ramachandran et al. (2000)	$PM_{2.5}$	DustTrak/Andersen[3]	N/A	N/A	2.99	0.92
		DustTrak/MSP[4]	N/A	N/A	1.94	0.76
Chung et al. (2001)	PM_{10}	DustTrak/SSI	60	-13.60	3.00	0.92

1. The intercept was not significantly different from zero ($\alpha = 0.05$).
2. The slope was not significantly different from zero ($\alpha = 0.05$).
3. Gravimetric sampler for outdoor measurement
4. Gravimetric sampler for indoor measurement
5. N/A: statistics are not available.

5.1.2.2.1.3.1 Descriptive Statistics

Descriptive statistics for $PM_{2.5}$ and PM_{10} measurements by the DustTrak and the HI are given in Tables 5.1.2.2 and 5.1.2.3, respectively. Real-time ten second interval measurements by the DustTrak were averaged to properly match the integrated sampling times of the HI measurements. Difference between the means and medians of the DustTrak data were modest for both $PM_{2.5}$ and PM_{10}, indicating the distributions were not skewed. However, there were significant disparities between the DustTrak and HI samplers. For instance, in the morning runs (from 23 to 35) and afternoon runs (24, 27, 30, and 33), integrated $PM_{2.5}$ concentrations from the HI fell outside the range (i.e. maximum – minimum) of the time-resolved DustTrak measurements.

The ratios of the DustTrak-measured $PM_{2.5}$ mean concentration divided by the HI-measured $PM_{2.5}$ concentration varied substantially. Seven runs (23, 24, 25, 27, 30, 31, and 33) involving three types of routes (RS, U1, and U2) and two types of buses (RE2 and TO1) had DustTrak/HI $PM_{2.5}$ ratios greater than 2. No obvious patterns were seen across bus type, including for the CNG bus or route. Figures 5.1.2.1 and 5.1.2.2 show the differences between the DustTrak and HI samplers across morning and afternoon runs, respectively.

Table 5.1.2.2 Descriptive statistics for $PM_{2.5}$ ($\mu g/m^3$) measured by HI and DustTrak.

Run	Date	Time	Week	Bus	Route	DustTrak Minimum	DustTrak Maximum	DustTrak Median	DustTrak Mean	HI	Ratio[1]
4	04/30/02	PM	2	HE2	RS	9.0	34.0	14.0	14.9	40.6	0.4
5	05/01/02	AM	2	HE2	U1	35.0	148.0	53.0	58.8	40.6	1.4
6	05/01/02	PM	2	HE2	U1	8.5	115.5	15.0	20.3	12.5	1.6
7	05/07/02	PM	3	HE3	RS	51.0	161.0	74.0	76.5	44.4	1.7
10	05/08/02	PM	3	HE3	U1	29.0	70.5	37.0	39.0	29.0	1.3
11	05/13/02	PM	4	RE1	RS	29.5	161.5	47.0	51.7	47.1	1.1
12	05/14/02	AM	4	RE1	U1	33.0	61.0	43.5	44.0	37.8	1.2
14	05/16/02	AM	4	RE1	U1	37.5	89.5	61.0	61.8	42.0	1.5
15	05/16/02	PM	4	RE1	U1	26.0	157.0	41.0	47.5	31.5	1.5
16	05/20/02	PM	4	RE1	RS	4.0	43.0	11.0	13.4	18.1	0.7
17	05/21/02	AM	5	RE2	U1	41.0	147.0	75.3	78.0	50.9	1.5
18	05/21/02	PM	5	RE2	U1	3.5	167.0	11.0	15.4	21.2	0.7
19	05/22/02	AM	5	RE2	U1	25	146.5	78.5	78.8	48.5	1.6
20	05/22/02	PM	5	RE2	U1	12.0	204.5	24.0	32.5	22.4	1.5
23	05/29/02	AM	6	RE2	U2	59.5	148.0	112.8	111.0	49.2	2.3
24	05/29/02	PM	6	RE2	U2	58.5	227.0	109.0	109.0	41.6	2.6
25	05/30/02	AM	6	RE2	U2	74.0	180.5	129.5	128.7	48.9	2.6
26	05/30/02	PM	6	RE2	U2	35.0	219.5	72.5	74.2	41.3	1.8
27	06/04/02	PM	7	TO1	RS	64.5	211.5	118.5	117.3	40.3	2.9
28	06/05/02	AM	7	TO1	U1	79.0	128.5	100.0	99.8	57.0	1.8
30	06/05/02	PM	7	TO1	U1	105.0	163.5	119.0	123.0	61.8	2.0
31	06/06/02	AM	7	TO1	U1	97.0	158.0	127.5	128.0	62.1	2.1
33	06/06/02	PM	7	TO1	U1	78.0	224.0	103.0	106.0	49.6	2.1
34	06/11/02	PM	8	CNG	RS	57.5	211.0	91.0	89.6	57.1	1.6
35	06/12/02	AM	8	CNG	U1	39.5	68.5	48.5	49.0	36.1	1.4
36	06/12/02	PM	8	CNG	U1	13.5	117.5	21.0	27.8	26.7	1.0

1. Ratio = (DustTrak $PM_{2.5}$ mean concentration) / (HI $PM_{2.5}$)

For PM_{10} measurements, the overall differences between the DustTrak and HI samplers were less pronounced than for $PM_{2.5}$. Only two runs (30 and 33) showed HI integrated concentrations outside the range of the DustTrak time-resolved data (Table5.1.2.3) and the DustTrak/HI PM_{10} ratios ranged from 0.4 to 1.7, more closely distributed around unity than for $PM_{2.5}$. Fourteen of twenty-six runs (i.e. 50% of all bus commute runs) had ratios less than 1, while only three runs for $PM_{2.5}$ had ratios less than 1.

Another problem with certain DustTrak data in this study was that mean $PM_{2.5}$ mass concentrations were higher than mean PM_{10} mass concentrations for several morning and afternoon runs (16, 19, 20, 23, 24, and 25).

Table 5.1.2.3 Descriptive statistics for PM_{10} ($\mu g/m^3$) measured by HI and DustTrak.

Run	Date	Time	Week	Bus	Route	DustTrak Minimum	DustTrak Maximum	DustTrak Median	DustTrak Mean	HI	Ratio[1]
4	04/30/02	PM	2	HE2	RS	13.0	75.5	23.5	24.4	37.0	0.7
5	05/01/02	AM	2	HE2	U1	33.5	123.5	50.5	54.8	63.7	0.9
6	05/01/02	PM	2	HE2	U1	9.5	102.0	17.0	21.5	28.2	0.8
7	05/07/02	PM	3	HE3	RS	53.5	145.5	76.5	77.5	65.6	1.2
10	05/08/02	PM	3	HE3	U1	35.0	74.0	44.5	45.8	52.0	0.9
11	05/13/02	PM	4	RE1	RS	32.0	870.5	57.0	69.3	105.5	0.7
12	05/14/02	AM	4	RE1	U1	37.0	71.5	50.0	50.9	78.3	0.6
14	05/16/02	AM	4	RE1	U1	38.5	101.0	61.5	62.1	77.3	0.8
15	05/16/02	PM	4	RE1	U1	29.5	142.0	45.5	50.2	50.9	1.0
16	05/20/02	PM	4	RE1	RS	2.0	43.0	10.0	11.7	27.3	0.4
17	05/21/02	AM	5	RE2	U1	51.0	157.5	76.8	81.4	86.3	0.9
18	05/21/02	PM	5	RE2	U1	6.0	159.5	16.0	20.4	24.9	0.8
19	05/22/02	AM	5	RE2	U1	24.5	146.5	76.0	78.1	86.9	0.9
20	05/22/02	PM	5	RE2	U1	11.0	173.0	23.8	32.3	47.0	0.7
23	05/29/02	AM	6	RE2	U2	50.0	154.5	104.0	103.2	80.3	1.3
24	05/29/02	PM	6	RE2	U2	57.5	258.0	102.0	104.9	71.5	1.5
25	05/30/02	AM	6	RE2	U2	67.5	171.5	125.0	122.6	72.8	1.7
26	05/30/02	PM	6	RE2	U2	40.5	238.5	73.0	76.4	66.7	1.1
27	06/04/02	PM	7	TO1	RS	77.0	242.0	140.0	139.9	98.3	1.4
28	06/05/02	AM	7	TO1	U1	80.5	139.0	100.5	101.5	104.8	1.0
30	06/05/02	PM	7	TO1	U1	102.5	174.0	123.0	126.4	97.3	1.3
31	06/06/02	AM	7	TO1	U1	98.0	166.0	129.5	130.5	99.4	1.3
33	06/06/02	PM	7	TO1	U1	80.5	230.0	106.0	109.5	73.7	1.5
34	06/11/02	PM	8	CNG	RS	61.5	450.0	100.5	101.3	101.8	1.0
35	06/12/02	AM	8	CNG	U1	41.0	73.0	53.0	53.3	68.8	0.8
36	06/12/02	PM	8	CNG	U1	24.5	121.5	31.5	37.1	50.3	0.7

1. Ratio = (DustTrak PM_{10} mean concentration) / (HI PM_{10})

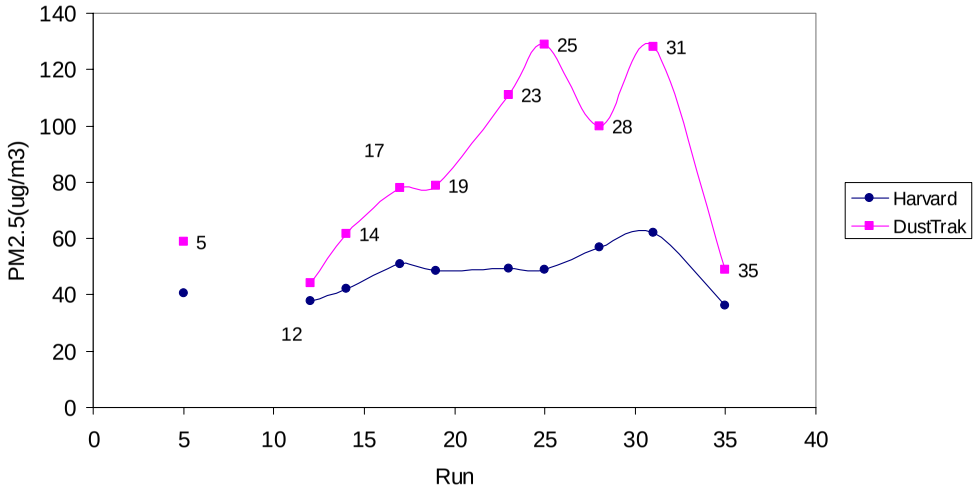


Figure 5.1.2.1 Comparison between DustTrak and HI data for $PM_{2.5}$ (morning only).

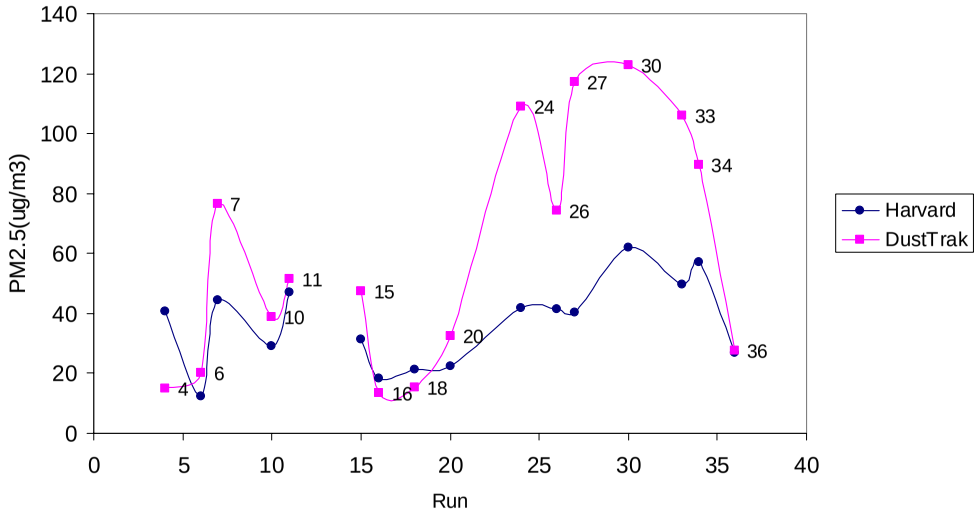


Figure 5.1.2.2 Comparison between DustTrak and HI data for $PM_{2.5}$ (afternoon only).

5.1.2.2.1.3.2 Assessment of Difference Between Samplers

To assess the difference between samplers quantitatively, simple linear regressions were performed for both $PM_{2.5}$ and PM_{10} data. As noted earlier, in previous studies (Ramachandran et al., 2000; Chung et al., 2001; Yanosky et al., 2002), the DustTrak overestimated the concentration of airborne particulate matter by factors of two to three (Table 5.1.2.1) compared to filter-based sampling techniques and a similar result was observed in the present study for $PM_{2.5}$ but not for PM_{10}.

The intercept terms were not found to be statistically significant (i.e., no systematic bias) at the significance level of 0.05 for both $PM_{2.5}$ and PM_{10}. However, the estimated slopes were significantly different from zero at the significance level of 0.05, indicating a proportional bias. The slope of 2.34 for $PM_{2.5}$ fell within the range observed in previous studies (Ramachandran et al., 2000; Yanosky et al., 2002), while the slope of 1.25 for PM_{10} was lower than the value of 3 found by Chung et al. (2001), who employed the SSI gravimetric method, different from the HI used in the present study.

The correlations observed in the present study were modest (R^2 of 0.66 for $PM_{2.5}$ and 0.67 for PM_{10}), compared with previous studies (R^2 from 0.76 to 0.92) reported in Table 5.1.2.1. However, in the present study data were collected on moving platforms, school buses, while the previous studies were conducted conventionally on roofs or inside of buildings. A degradation in correlation for instruments operated on moving platforms might be expected, although it is interesting that the patterns of differences in measurements between the DustTrak and the HI samplers were similar in the present study to the previous studies for $PM_{2.5}$.

Based on the results presented above, and considering the HIs are the Federal reference method, we decided to separate the use of the data obtained by these and the DustTrak

instruments. The HI integrated samples data were used to provide absolute concentrations which could be compared with other studies. The data collected with the DustTrak instruments were used only on a relative basis or when real-time series were required for the analysis.

5.1.2.2.2 Real-Time Fine Particle Counts (0.3 to 10.0 μm)

Optical particle counters count pulses of scattered light with high efficiency. The intensity of the pulse is directly (but not linearly) related to particle size. The output pulses are sorted into bins that are therefore related to particle size. The intensity of the pulse is dependent on the physical characteristics of the particle scattering the light. Refractive index and shape are two characteristics that have the greatest affect on particle size. The size bins from the output of optical particle counters were calibrated by the manufacturer by sampling aerosol consisting of spherical latex particles of known size and refractive index. While this calibration can be highly accurate (latex particle sizes are known to better than 1%), ambient particles have a wide variety of shapes and refractive indices. Optical particle counters therefore provide a size output that is operationally defined, assuming the particles scatter light in the same manner as latex beads. Measurement accuracy can only be determined for latex beads and the accuracy in size determination of ambient aerosol cannot be measured.

5.1.2.2.3 Real-Time Ultrafine Particle Counts (0.03 to 0.8 μm)

A CE-CERT Scanning Electrical Mobility Spectrometer (SEMS) was used to measure particles in the range of 0.03 to 0.8 um electrical mobility diameter. These analyzers charge particles and then separate them by size based on their electrical mobility. The particles are then detected with a condensation nuclei counter (CNC). Only a small fraction of the particles become charged, with the charging efficiency dependent on particle size, shape, and composition. The CNC can count the particles essentially quantitatively. For the SEMS, the size range is determined by sampling aerosol particles of known size, shape and composition, therefore, it is not possible to establish a meaningful accuracy when sampling ambient aerosol since the size, shape and composition can vary widely. This imposes significant restrictions and complexities in the analysis of the data generated by the instrument. Although we collected samples using the SEMS for the majority of the bus runs, analyses of these data were beyond the scope of the present project.

5.1.2.2.4 Real-Time Black Carbon

These analyzers (Aethalometers) determine the concentration of black carbon by determining the light absorption after collection on a filter. Black carbon is generally the only significant component of ambient particulate that absorbs light at the near IR wavelength used. The calibration cannot be determined directly, but must be established empirically by comparison with PM collected on filters and analyzed by thermal decomposition. This analysis method also has inherent errors since the amount of elemental carbon can change during analysis by charring and pyrolysis, the degree of which is dependent on the chemical composition of the PM. The measurement of elemental carbon in ambient air is therefore an operational definition, and accuracy is difficult to establish and depends on the aerosol composition. Thermal methods sampling ambient air generally agree to within about 10 percent and this may be used to estimate accuracy. The real time analyzers generally agree to within about ten percent of that determined by filter collection followed by thermal analysis. The overall accuracy therefore is expected to be in the range of twenty percent.

5.1.2.2.5 Real-Time Particle Phase PAH

This analyzer does not directly measure PAHs, but rather their photoionization products. The manufacturer's calibration is based on an empirical relationship, with PAH measured in ambient air by filter collection followed by extraction and gas chromatographic analysis. Since the response depends on the composition of the PM, it is not possible to establish the accuracy of the method. Based on the manufacturers data, we expect ambient air accuracy to generally be fifteen percent.

5.1.2.2.6 Gaseous Analyses

Tedlar bag samples were analyzed for 1,3-butadiene by gas chromatography with flame ionization detection. The accuracy is dependent on sample injection accuracy, loss of compound in the bag, and calibration standards used as the chromatographic standard. Sample injection is done with a loop of fixed size, which changes volume only as a result of atmospheric pressure. We have shown that losses of 1, 3-butadiene are within the precision of the measurement (several percent) for periods of up to four days. This was done by collecting ambient samples enriched in 1,3-butadiene by sampling near an idling diesel engine and then periodically analyzing them. The GC accuracy was established by calibrating using a gas cylinder certified by the manufacturer to +/- 2% based on NIST standards. The overall accuracy is estimated to be within five percent.

Tenax samples were collected using needle valves for flow control and rotameters for flow measurement. As with the filter collection the accuracy of flow control is estimated to be within five to ten percent. Gas chromatography with flame ionization was used to measure aromatic compounds desorbed from the Tenax. The calibration of the measurement was based on a compressed gas standard certified by the manufacturer to within two percent. The overall measurement accuracy was estimated to be within ten percent.

Carbonyl compounds were measured by collecting them as a 2,4-dinitrophenyl hydrazine adduct and analyzing the adduct by high performance liquid chromatography (HPLC). The flow rate was controlled by needle valves and measured with a rotameter. As with the filter collection the accuracy of flow control is estimated to be within five to ten percent. Gaseous calibration standards are not available for carbonyls and the calibration is established by weighing the adduct. Weighing and dilution accuracy is estimated from methods traceable to the NIST and based on these procedures the accuracy of the standards is estimated to be within ten percent. Overall accuracy of the method is estimated to be twenty percent based on this analysis and field comparisons with spectroscopic methods.

5.1.2.2.7 Portable Carbon Monoxide Analyzers

These analyzers were not audited, but their accuracy may be estimated by comparison with the infrared analyzer that was audited. The accuracy may be affected when sampling ambient air due to interferences, since electrochemical cells do not have high specificity. The interference due to hydrogen also needs to be addressed since this is a known interferent and the lead-acid batteries carried in the bus may have been a source of hydrogen.

5.1.3 Evaluation of Precision Between Paired Instruments

Two contrasts tested in this study were differences in concentrations at the front and the rear of the bus cabin, and between inside and outside the bus. To help validate these contrasts, we assessed the precision of the pairs of instruments which were used to simultaneously measure at the front and rear of the bus cabin, or inside and outside the bus during a commute. If there was not substantial agreement between the readings of the pair of the same instruments when they were sampling the same volume of air under common sampling conditions, we would be limited in the inferences we could draw from the measurements made when sampling different air parcels (e.g., inside versus outside). Thus the results of our precision experiments were used to determine which pairs of instruments had an acceptable level of precision for evaluating simultaneous measurements at the front and rear of the bus cabin, or inside and outside the bus. It is important to note, for all comparisons which relied solely on a single set of instruments instead of paired instruments (e.g. comparisons between bus types, windows open or closed, different microenvironments, comparisons to ambient air), the precision results reported here do not apply.

During the pilot study, low pollutant concentrations during the precision checks for a number of the instruments made it difficult to interpret differences observed in pollutant concentrations between the front and back of the cabin, and between inside and outside the bus. One of the improvements of the main study experimental design was that we performed instrument precision measurements in locations where the higher concentrations of many of the analyzed pollutants were expected to provide more robust precision assessment. In general, all instruments improved their precision when exposed to higher concentrations. Therefore, the precision was potentially better during the bus commutes since the values observed during these tests were much higher than during the precision experiments.

Precision between paired Aethalometers, however, did not improve during the main study precision tests. The relatively clean background conditions in West Los Angeles, even a few tens of meters from the I-405, often resulted in measured black carbon concentrations that were not sufficiently above the detection limit. During the collocation experiments, the measured concentrations for this pollutant were most of the time below 4 ug/m^3 and for these values the signal-to-noise ratio produced by the Aethalometers did not yield appropriate precision results. Even so, the post-study controlled experiments (see below) showed the precision between the pairs of Aethalometers improved substantially at higher concentrations. Therefore we believe that at the peak black carbon concentrations encountered during the main study runs, the precision between the paired Aethalometers was substantially better than that indicated by the precision experiments.

During the main study, seventeen instrument precision experiments were conducted when the bus was parked with the engine off and the windows fully opened. For these experiments, the instruments were sampling inside concentrations and the sample inlets from each pair of instruments were separated by the length of the bus. That is, one set of instruments had its sample inlets in the rear of the cabin while the other set had them in the front. This arrangement was necessary since the logistics of relocating the sampling inlets to be adjacent to each other was too complex to be performed once the bus was instrumented for the runs.

Although the sampling inlets were not exactly in the same location, the fact that the bus was at rest and fully ventilated was expected to lead to uniform pollutant concentrations from ambient air entrained throughout the cabin. Thus, under these conditions, no significant differences were expected in measurements by the pairs of instruments.

Six instrument precision experiments were conducted with the test bus parked close to the intersection of two congested streets (Sepulveda Blvd and Wilshire Blvd) and near the I-405 freeway in West Los Angeles. The remaining instrument precision experiments were conducted either inside a parking lot at UCLA, in front of the Brentwood Science Magnet School, or in front of several of the schools which served as bus stops on the routes used for the bus runs.

We also conducted a post-study experiment to assess the precision between the paired instruments at much higher concentrations in a more controlled laboratory environment. Table 5.1.3.1 summarizes the main characteristics of the instrument precision experiments and Figures 5.1.3.1 to 5.1.3.5 show a series of boxplots illustrating, as discussed below, the variation in differences between the pairs of instruments over the course of the study. With the exception of the Aethalometers, these graphs were plotted using the same numerical scale (although in several cases with different units) to emphasize the variation of the differences not only within runs but also between instruments. Figures 5.1.3.1 to 5.1.3.5 depict the results only in terms of absolute precision. Relative precision analyses are presented in Section 5.1.3.1.1.

Table 5.1.3.1 General characteristics of the paired instrument precision experiments.

Collocation Run Number	Date	Time	Start Time	Stop Time	Location	Week Number
101	04/23/02	PM	10:00	12:00	UCLA	1
201	05/01/02	PM	17:32	18:25	UCLA	2
301	05/07/02	PM	15:00	15:33	Sepulveda Blvd	3
401	05/13/02	PM	14:51	15:21	Sepulveda Blvd	4
402	05/14/02	AM	6:22	6:36	First Bus Stop	4
403	05/14/02	PM	14:50	14:59	Brentwood	4
404	05/16/02	AM	8:32	9:34	UCLA	4
405	05/16/02	PM	11:00	14:10	UCLA	4
501	05/20/02	PM	15:00	15:30	Sepulveda Blvd	4
502	05/20/02	PM	15:56	16:57	Sepulveda Blvd	5
503	05/22/02	AM	6:10	6:29	First Bus Stop	5
504	05/23/02	PM	15:04	15:19	Brentwood	5
601	05/29/02	PM	14:55	15:25	First Bus Stop	6
602	05/30/02	PM	15:08	15:27	First Bus Stop	6
701	06/04/02	PM	15:23	16:02	Sepulveda Blvd	7
702	06/05/02	AM	6:15	6:29	First Bus Stop	7
801	06/11/02	PM	15:04	16:43	Sepulveda Blvd	8
802	06/17/02	PM	12:55	16:21	Laboratory	

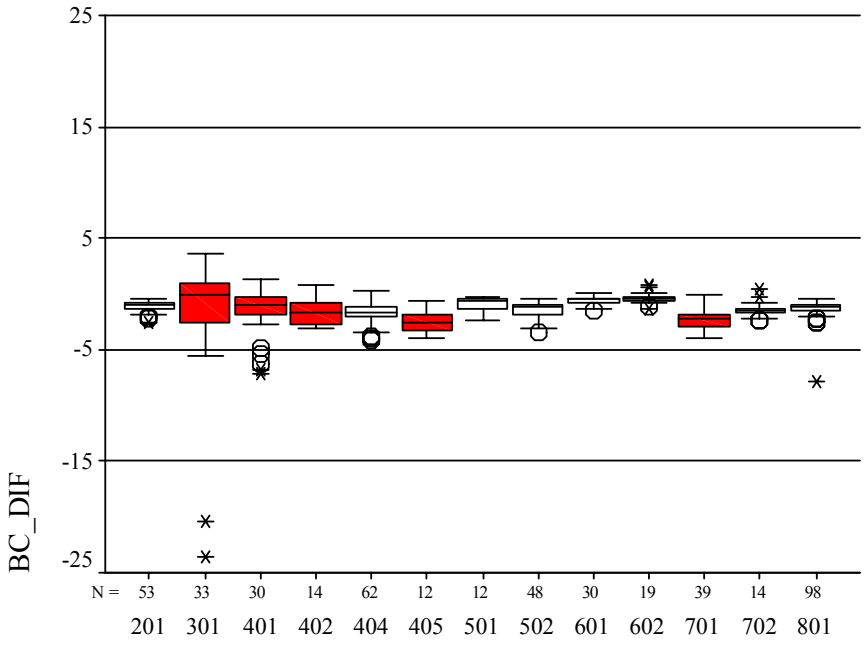


Figure 5.1.3.1 Variation in absolute differences (µg/m^3) between paired Aethalometers over the course of the main study.

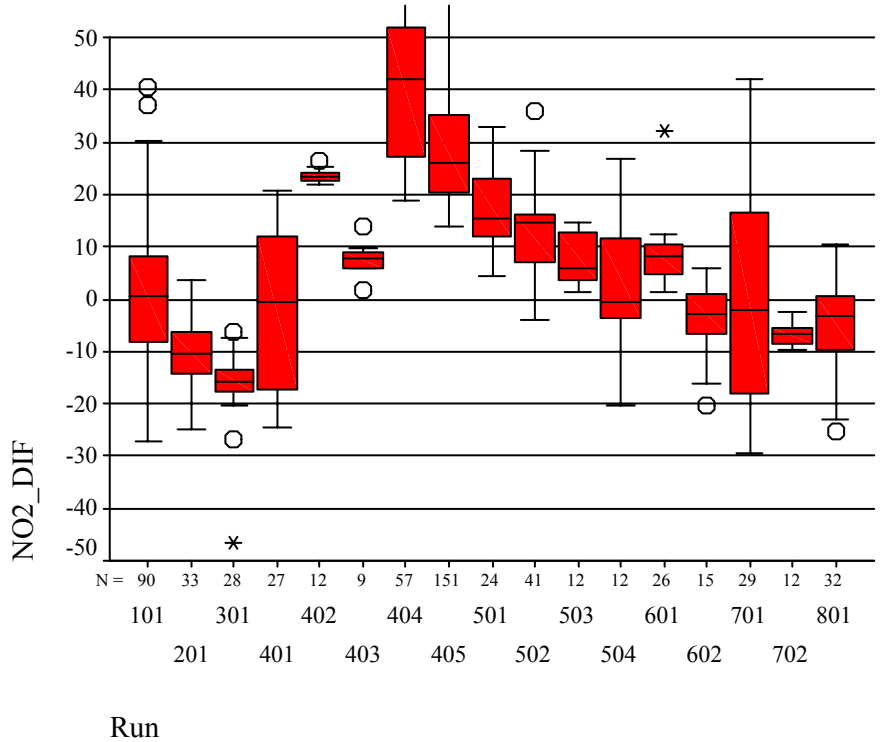


Figure 5.1.3.2 Variation in absolute differences (ppb) between paired NO_2 instruments over the course of the main study.

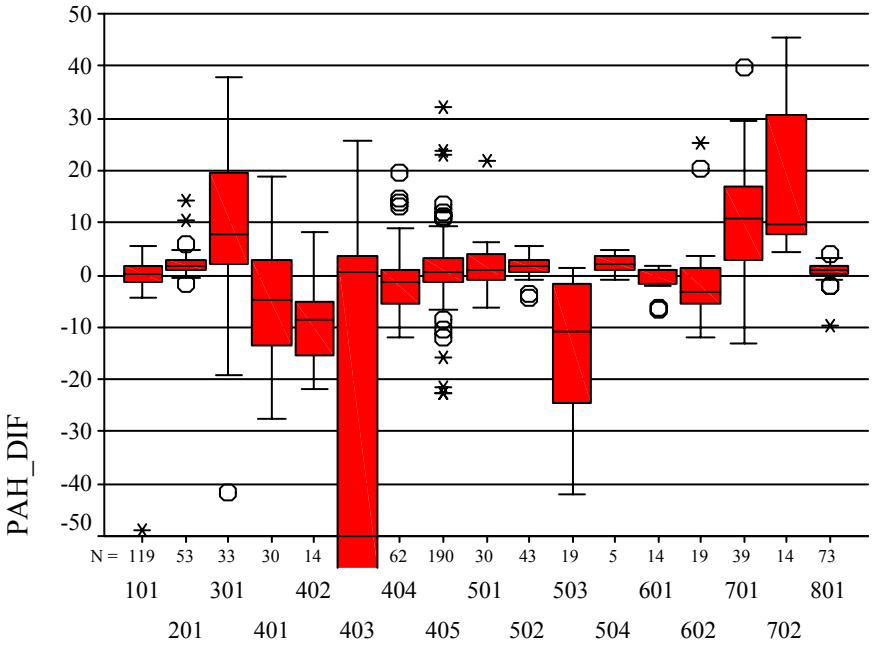


Figure 5.1.3.3 Variation in absolute differences (ng/m^3) between paired PAH instruments over the course of the main study.

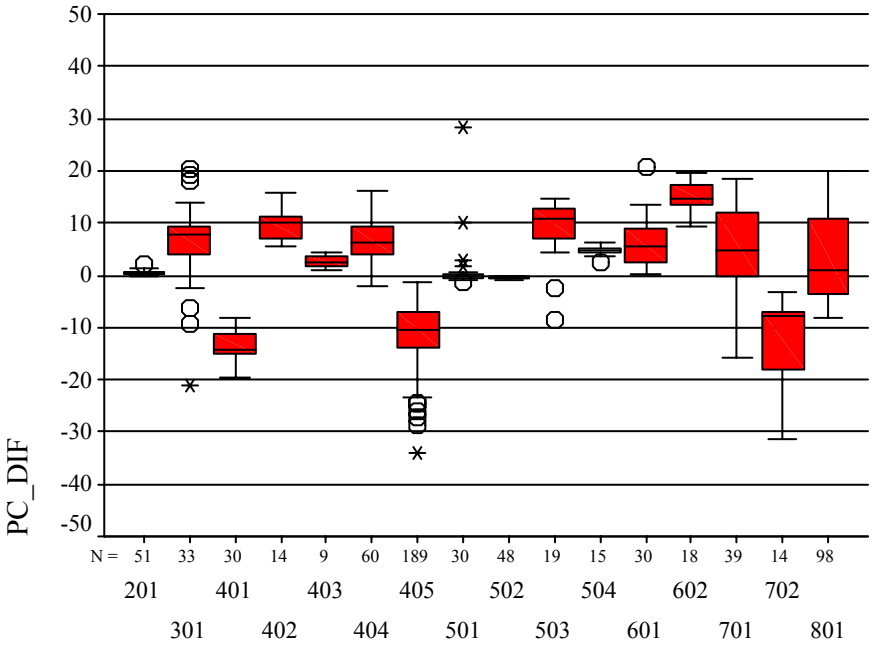


Figure 5.1.3.4 Variation in absolute differences (#/cm^3) between paired fine particle count instruments over the course of the main study.

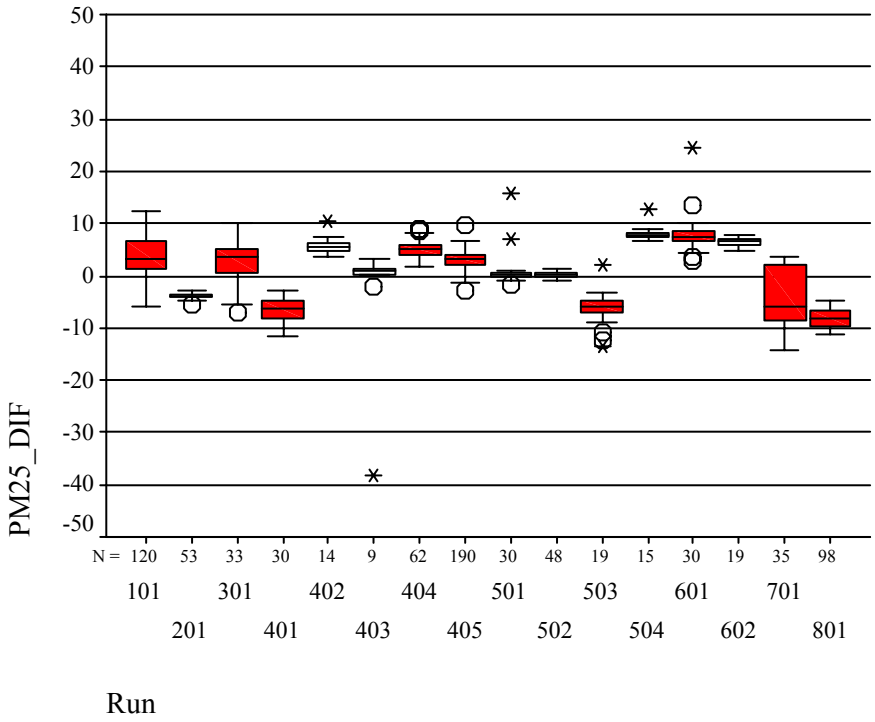


Figure 5.1.3.5 Variation in absolute differences (μg/m^3) between the DustTrak $PM_{2.5}$ instruments over the course of the main study.

The subsequent assessment of the instrument precision data quality was performed via two different methods: precision estimation and correlation between pairs of instruments.

5.1.3.1 Precision Estimation Between Paired Instruments

The simplest way to establish the quality of the paired instrument precision data was to compute the precision of the paired instruments, which we did by using absolute differences, percent differences and coefficients of variation.

To perform these calculations, the original ten-second data were smoothed using a one-minute moving average. This procedure was used only for this analysis. The remaining calculations presented in this report used the validated 10-second data. We tested different periods for the moving average, ranging from 30-second to 10-minute averages. After analyzing the effects of such averaging periods on the results of the precision calculations, we selected the 1-minute averaging time since it provided enough time resolution while minimizing noise in the concentration-time series. Although this procedure improved the results of the precision experiments, it did not affect the outcome of our analyses. The paired instrument precision values were only used on a relative basis to rank the performance of the paired instruments and to determine their suitability for evaluating front/rear and inside/outside contrasts (see Section 5.1.3.3).

5.1.3.1.1 Absolute and Percent Differences

We used equation (5.1.3.1) to calculate the average absolute precision (P) between paired instruments:

$$P = \frac{\sum c_1 - c_2}{N} \pm \left(1.96 \bullet \sigma_\mu\right) \qquad (5.1.3.1)$$

where c_1 is the concentration of a pollutant obtained with one of the instruments at a given time, c_2 is the concentration of the same pollutant obtained with the second instrument at the same time, N is the sample size, and σ_μ is the standard deviation of the distribution of differences (standard error).

To give context to the results, we computed the median of all bus run data for each pollutant. We then calculated the ratio between the average absolute precision between paired instruments during precision experiments, and the median value of the bus runs. We refer to this ratio as the percentage precision. Table 5.1.3.2 summarizes these results for the pollutants that were considered for the real-time analyses.

We observed from Table 5.1.3.2, that in terms of percentage precision between paired instruments, the pair of instruments reading fine particle counts in the size range of 0.3–0.5 μm (PC) exhibited the best behavior (5% precision), closely followed by the PAH and DustTrak $PM_{2.5}$ analyzers (8 and 9%, respectively). The NO_2 instruments precision was 13% while the Aethalometer (measuring black carbon) exhibited a 53% precision.

During weeks four and five, the NO_2 analyzers experienced technical problems that made the interpretation of the data difficult. Hence, for the estimation of precision of NO_2 measurements, we did not use the data from the NO_2 precision experiments conducted during those weeks.

The poor precision for Aethalometers (53% precision) limited front-to-back and inside-to-outside comparisons of black carbon bus commute data. Therefore, for black carbon, we focused on comparisons that did not require the use of paired instruments, including our assessment of variables such as bus type, roadway type, and window position, which were done using the same instrument on all buses.

The results of the laboratory post-study precision experiment confirmed findings from the bus-based precision measurements discussed above. The precisions obtained during the post-study tests were similar to those obtained during the weekly instrument precision experiments for black carbon, NO_2, PAH, and PC instruments. For the $PM_{2.5}$ analyzers (DustTrak), the precision improved considerably under laboratory conditions (see Table 5.1.3.2). The relative precision obtained during the laboratory test was approximately 65% for black carbon, 20% for NO_2, 10% for PAH, 7.5% for PC, and about 1% for $PM_{2.5.}$

Several of the instrument precision experiments had sample sizes too small to allow meaningful calculations and therefore not all of the seventeen runs mentioned in Table 5.1.3.1 are presented in Table 5.1.3.2.

Table 5.1.3.2 Paired instrument precision (average absolute and percent differences) (*)

	Units	101	201	301	401	404	405	501	502	601	701	801	802 (1)	Range of controlled experiment (2)	Percent difference controlled experiment (%)	Average absolute precision (3)	Median (4)	Percent difference (%)	Typical peak (5)
Black carbon	ug/m^3	N/A	-1.15 ± 0.1	-1.7 ± 0.1	-1.6 ± 0.3	-1.7 ± 0.1	-2.5 ± 0.3	-0.9 ± 0.2	-1.5 ± 0.1	-0.6 ± 0.1	-2.3 ± 0.1	-1.3 ± 0.1	-17.5 ± 5.6	0.2 - 52	24	± 1.6	3	53%	20 - 40
NO_2	ppb	1.4 ± 1.0	-10.1 ± 2.4	-15.7 ± 0.8	-2 ± 1.3	40 ± 1.2	28.3 ± 0.6	16.5 ± 1.1	12.7 ± 1.0	8.4 ± 0.9	1.8 ± 3.2	-4.9 ± 1.1	-3.1 ± 0.7	7 - 29.3	17	± 7	55	13% (6)	150
PAH	ng/m^3	0.18 ± 0.2	1.78 ± 0.5	9.6 ± 2.4	-4.1 ± 1.6	-1.2 ± 0.6	0.6 ± 0.3	1.4 ± 0.6	1.4 ± 0.4	-0.8 ± 0.6	9.6 ± 1.3	0.7 ± 0.1	-98.9 ± 14.3	0.5 - 2000	10	± 3.0	40	8%	500 - 1000
PC	$\#/cm^3$	N/A	0.52 ± 0.2	6.2 ± 1.2	-13.4 ± 0.4	6.7 ± 0.5	-11.2 ± 0.3	0.85 ± 0.6	-0.5 ± 0.1	5.4 ± 0.7	5.3 ± 1.0	2.9 ± 0.6	-52.1 ± 19.9	6 - 1400	7	± 5.2	95	5%	150 - 200
$PM_{2.5}$	ug/m^3	3.5 ± 0.3	-3.9 ± 0.2	2.7 ± 0.6	-6.6 ± 0.3	5 ± 0.1	3.1 ± 0.1	0.5 ± 0.3	0.3 ± 0.1	5.8 ± 0.9	-5.8 ± 1.9	-9.8 ± 0.4	13.8 ± 140	29 - 31600	1	± 4.3	50	9%	100 - 200

(*) These values correspond to the mean and the confidence interval of the absolute difference distributions among the instrument precision experime
The absolute difference was computed subtracting the concentrations in thee front from the concentrations in the rear of the cat
The original 10-seconds data were smoothed using a 1-minute moving averag

[1] Instrument precision experiment performed under controlled conditions in the laboratory (post-study test).
[2] Range of values observed during the instrument precision experiments performed in the laboratory (Run 802).
[3] Average precision of experiments performed inside the bus. Does not include run 802.
[4] Median of the entire data set, including all the runs.
[5] Range of peak concentrations observed during exposure runs.
[6] The NO_2 instruments experienced technical problems during weeks four and five, therefore, the instrument precision results for these weeks were not considered for this calculation.

5.1.3.1.2 Coefficient of Variation

U.S. EPA's guidelines (U.S. EPA, 2000) recommend the use of the coefficient of variation as a measure of precision between instruments sampling the same volume of air under the same sampling conditions. The first step of this method is to calculate the percent difference (D_i) between each measurement using equation (5.1.3.2):

$$D_i = \frac{Y_i - X_i}{\frac{(Y_i + X_i)}{2}} \bullet 100 \qquad (5.1.3.2)$$

where Y_i is the concentration measured by one of the instruments at a given time and X_i is the concentration measured by the other instrument at the same time. The percent difference is then used to calculate the coefficient of variation (CV_i) for each data point using equation (5.1.3.3):

$$CV_i = \frac{|D_i|}{\sqrt{2}} \qquad (5.1.3.3)$$

Finally, we computed the overall coefficient of variation (CV) as the root mean square of the distribution of individual coefficients using equation (5.1.3.4):

$$CV = \sqrt{\frac{\sum CV_i}{N}} \qquad (5.1.3.4)$$

These results are summarized in Table 5.1.3.3, which shows that in terms of the coefficient of variation, the pair of instruments measuring fine particle counts in the size range of 0.3 – 0.5 μm (PC) exhibited the best behavior (7% precision), the $PM_{2.5}$ analyzers (DustTrak) ranked second (14% precision), and the PAH and NO_2 instruments ranked third and fourth (23 and 33%, respectively). Again the Aethalometers ranked last (52%).

5.1.3.2 Correlation Between Pairs of Instruments

Precision is one means of measuring agreement. Another is tests of correlation. We used the Pearson's correlation coefficient to describe the strength of the linear association between the readings of the pairs of instruments.

Table 5.1.3.4 summarizes the correlation coefficients found for the instrument precision experiments. During these tests both instruments in each pair were supposed to behave similarly, therefore, the values of the correlation coefficients were expected to be close to one. Additional logarithmic transformations of the data distributions (not shown) did not appreciably improve the values presented below.

The highest average correlation coefficient was obtained for the PC data (0.70), closely followed by PAH (0.65) and then by NO_2 (0.57) and $PM_{2.5}$ (0.54). Once again, the poorest results were obtained for the BC data (0.18).

For the correlation coefficients analysis, the laboratory post-study experiments showed different results than the weekly instrument precision experiments. $PM_{2.5}$ exhibited the best correlation during the laboratory tests (0.90), while the correlation coefficient associated with

Table 5.1.3.3 Estimation of precision between paired instruments using the coefficient of variation (%) [(*)].

	101	201	301	401	404	405	501	502	601	701	801	Average Variation
Black Carbon	N/A	67 ± 12	34 ± 18	33 ± 22	55 ± 10	61 ± 22	39 ± 26	74 ± 16	29 ± 16	54 ± 11	68 ± 11	52%
NO_2	N/A	26 ± 9	33 ± 27	15 ± 4	50 ± 3	58 ± 3	20 ± 11	23 ± 12	31 ± 30	48 ± 17	26 ± 14	33%
PAH	N/A	23 ± 7	23 ± 11	14 ± 7	25 ± 10	17 ± 16	22 ± 16	35 ± 15	31 ± 18	20 ± 8	19 ± 12	23%
PC	N/A	3 ± 2	7 ± 2	23 ± 4	8 ± 2	12 ± 5	4 ± 2	4 ± 1	2 ± 1	5 ± 3	6 ± 2	7%
$PM_{2.5}$	N/A	30 ± 3	8 ± 3	17 ± 4	13 ± 2	6 ± 3	10 ± 8	9 ± 4	4 ± 2	12 ± 4	29 ± 4	14%

(*) The values in the table represent the coefficient of variation and the 95% confidence interval
The original 10-seconds data were smoothed using a 1-minute moving average

Table 5.1.3.4. Correlation coefficients for paired instrument precision experiments

	101	201	301	401	404	405	501	502	601	701	801	802 (1)	Average
Black Carbon	N/A	0.13	0.34	0.03	0.10	0.04	0.10	0.04	0.24	0.65	0.14	0.62	0.18
PAH	0.45	0.76	0.69	0.76	0.73	0.71	0.83	0.13	0.46	0.79	0.85	0.88	0.65
NO_2	0.88	0.83	0.14	0.08	0.91	0.89	0.60	0.89	0.56	0.12	0.42	N/A[3]	0.57
PC	N/A	0.84	0.44	0.34	0.59	0.93	0.88	0.88	0.76	0.39	0.98	0.62	0.70
$PM_{2.5}$[2]	0.83	0.35	0.12	0.54	0.51	0.87	0.67	0.34	0.41	0.34	0.98	0.88	0.54

[1] Instrument precision experiments performed under controlled conditions in the laboratory

[2] Data obtained using the DustTraks

[3] Not applicable (see text)

PAH was 0.89, and for both BC and PC, 0.62. We were not able to establish a meaningful correlation coefficient for the NO_2 data due to the previously mentioned problem with the NO_2 instrument part way through the study.

The results for the higher concentration post-study experiments show that the limitations experienced during the in-field instrument precision experiments, in particular with the Aethalometer, are associated with the relatively low concentrations encountered during the bus-based precision experiments.

In addition to the correlation coefficient, which was used as a measure of variability between pairs of instruments, we considered the slope obtained from the linear regression. This quantity is a measure of bias between instruments and can be used to estimate correction factors if necessary. Table 5.1.3.5 summarizes these findings.

To better understand the implications of these results, we created Table 5.1.3.6, which shows the bias between instruments, expressed as the percent difference between the expected slope (one) and the slope observed from the linear regression (Table 5.1.3.5). From Table 5.1.3.6 we find, again, the PAH and the PC analyzers exhibited the best behavior along the course of the study. The average bias for those instruments was about 30%. $PM_{2.5}$ and NO_2 had biases of 46% and 61%, respectively, while black carbon measurements showed a bias of 85%.

5.1.3.3 Overall Ranking

Based on the results presented above, we generated an overall ranking for the instrument precision data. As described in Table 5.1.3.7, the best instrument pairs in terms of precision were the OPC (PC) and the PAH analyzer, which ranked first and second respectively. The DustTraks ($PM_{2.5}$) ranked third, the NO_2 analyzers fourth, and the Aethalometer (BC) fifth.

Although the DustTraks exhibited a relative high precision, these instruments also showed low accuracy (see Section 5.1.2.2). These results were somewhat expected since this type of instrument is known to be less accurate (Hinds, 1999).

Table 5.1.3.7 Overall ranking of the paired instrument precision data quality.

		CRITERIA			
Instrument	Pollutant	Precision (%)	Coefficient of Variation	Correlation	Overall Ranking
OPC Climet	Fine Particle Counts (0.3 to 0.5 um)	1	1	1	1
Ecochem PAS 2000	Particle-bound PAH	2	3	2	2
DustTrak	$PM_{2.5}$ Mass	2	2	4	3
UCR analyzer	NO_2	4	4	3	4
Aethalometer	Black Carbon	5	5	5	5

Table 5.1.3.5 Regression results for the paired instrument precision data

	101	201	301	401	404	405	501	502	601	701	801
Black Carbon	N/A	0.32	3.38	0.19	0.30	0.07	0.45	0.13	0.28	1.50	0.63
NO_2	0.74	0.91	0.08	-0.40	0.48	0.39	0.34	0.80	0.42	0.06	0.47
PAH	0.73	0.48	0.72	1.13	0.68	0.66	0.65	0.36	0.81	0.92	0.96
PC	N/A	0.85	0.66	0.51	0.52	1.01	0.18	1.00	0.76	0.66	0.79
$PM_{2.5}$	0.74	0.38	0.40	1.00	0.44	0.75	0.21	0.27	0.42	0.50	0.80

(*) The values in the table represent the slope calculated for the linear regression model.

The values in color represent cases where the p-values indicated non-significant results.

Table 5.1.3.6 Bias between the instruments during instrument precision experiments (*).

	101	201	301	401	404	405	501	502	601	701	801	Average Difference
Black Carbon	N/A	68%	238%	81%	70%	93%	55%	87%	72%	50%	37%	85%
NO_2	26%	9%	92%	140%	52%	61%	66%	20%	58%	94%	53%	61%
PAH	27%	52%	28%	13%	32%	34%	35%	64%	19%	8%	4%	29%
PC	N/A	15%	34%	49%	48%	1%	82%	0%	24%	34%	21%	31%
$PM_{2.5}$	26%	62%	60%	0%	56%	25%	79%	73%	58%	50%	20%	46%

(*) The values in the table represent the difference between the expected (one) and the observed slope from the linear regression.

As mentioned above, the poor results obtained for instrument precision for black carbon measurements are likely to be related to relatively low concentrations encountered during those tests. As stated earlier however, these precision estimates only apply to the precision between pairs of instruments. Paired instruments were only used to evaluate concentration differences between the front and rear of the bus cabin, and between the inside and outside of the bus during the same run. Paired instruments were not used to evaluate the differences in concentrations between buses. For each pollutant, the same instrument was used to measure concentrations at the rear of the bus cabin for all buses tested, and this measurement was used for the comparison between bus types (and all other comparisons reported except front/rear and inside/outside). Thus, the precision results reported here allowed us to select the best pairs of instruments for studying the front/rear and inside/outside contrasts, but do not represent an overall measure of precision of a given instrument.

5.1.3.4 Use of Correlation Coefficients

Table 5.1.3.7 shows the overall ranking of the instrument precision data quality, according to the criteria discussed above. PC and PAH measurements, during instrument precision experiments, ranked first and second respectively. These results suggest the PC and PAH analyzers provided the most reliable data for observing contrasts between pairs of instruments. For this reason, we focused on these two pollutants when performing the comparisons between pollutant concentrations measured at different locations during the bus runs (e.g., outside versus inside).

The results of the instrument precision experiments were used to understand the limitations of the data and to determine which instruments were suitable for certain analyses developed in the study. We did not apply any corrections to make one instrument equivalent to the other.

However, we also included the other pollutants as appropriate. For the bus runs, we found reasonably strong correlations between PAH and black carbon and between PC and $PM_{2.5}$. Tables 5.1.3.8 and 5.1.3.9 summarize these findings and again, suggest the relatively poor precision of the pair of Aethalometers was probably due to low concentrations encountered during the instrument precision experiments. Tables 5.1.3.8 and 5.1.3.9 show that black carbon and PAH tended to exhibit similar behaviors during the bus runs, while $PM_{2.5}$ and PC also exhibited similar trends and responses to the variables determining pollutant concentrations. Hence, for certain comparison we assumed black carbon and $PM_{2.5}$ concentrations could be explained based on the results obtained for PAH and PC, respectively.

Because the NO_2 data gathered during instrument precision experiments generally did not exhibit good precision we did not include NO_2 in the analyses of front/rear and inside/outside contrasts.

5.1.4 Evaluation of Sampling Line Losses

We have previously evaluated losses of PM_{10} through the sampling lines used in this study (1/4 OD polyethylene impregnated with metal to reduce static charge buildup). To do this, we used three collocated DustTraks equipped with PM_{10} inlets. They sampled ambient air for two hours, alternating ten minutes without tubing and ten minutes with tubing (1.7 m in length).

Table 5.1.3.8. Correlation between PAH and other pollutants during exposure runs

	HE1	HE2	HE3			RE1				RE2								TO1				CNG			
Run	2	5	6	8	10	12	13	14	15	17	18	19	20	23	24	25	26	28	30	31	33	35	36	Average (1)	All (2)
NO_2	0.01	0.01	0.27	0.02	0.25	0.02	0.31	0.01	0.40	0.28	0.16	0.01	0.08	0.10	0.05	0.21	0.29	0.41	0.13	0.57	0.18	0.45	0.09	0.19	0.03
BC	0.57	0.41	0.65	0.60	0.81	0.33	0.56	0.90	0.91	0.71	0.40	0.45	0.69	0.68	0.75	0.91	0.84	0.57	0.66	0.76	0.80	0.56	0.87	0.67	0.51
$PM_{2.5}$	0.01	0.56	0.53	0.48	0.69	0.81	0.35	0.45	0.85	0.03	0.10	0.01	0.57	0.01	0.05	0.76	0.02	0.18	0.48	0.24	0.64	0.12	0.84	0.38	0.03
PC	0.30	0.63	0.51	0.36	0.28	0.77	0.01	0.10	0.25	0.01	0.15	0.06	0.44	0.01	0.01	0.73	0.03	0.01	0.23	0.01	0.54	0.01	0.60	0.26	0.01

Table 5.1.3.9. Correlation between PC and other pollutants during exposure runs

	HE1	HE2	HE3			RE1				RE2								TO1				CNG			
Run	2	5	6	8	10	12	13	14	15	17	18	19	20	23	24	25	26	28	30	31	33	35	36	Average (1)	All (2)
NO_2	0.02	0.04	0.11	0.04	0.10	0.01	0.01	0.34	0.41	0.23	0.02	0.28	0.03	0.01	0.03	0.54	0.01	0.01	0.10	0.01	0.23	0.11	0.10	0.12	0.01
BC	0.36	0.45	0.51	0.56	0.24	0.20	0.01	0.18	0.21	0.03	0.19	0.33	0.39	0.02	0.04	0.73	0.04	0.01	0.24	0.02	0.52	0.01	0.61	0.26	0.01
$PM_{2.5}$	0.10	0.81	0.60	0.84	0.35	0.71	0.07	0.66	0.41	0.63	0.62	0.76	0.57	0.12	0.60	0.82	0.62	0.01	0.39	0.38	0.66	0.68	0.69	0.53	0.81

[1] Arithmetic average of runs presented in the table

[2] Correlation coefficient estimated using the entire data set

Data were collected as thirty-second averages. The data are summarized in Table 5.1.4.1. Based on the means, the tubing caused a loss of PM ranging from 21% to 29% depending on which DustTrak was evaluated.

Table 5.1.4.1 Mean DustTrak PM_{10} response on ambient air with and without Bev-A-Line tubing.

Mean DustTrak 1 (mg/m^3) w/o tubing	Mean DustTrak 1 (mg/m^3) w/ tubing	Difference DT 1 (w/o-w)	% Difference DT 1 (w/o-w)	SD of Difference
0.269	0.213	0.056	20.8%	0.117
Mean DustTrak 2 (mg/m^3) w/o tubing	Mean DustTrak 2 (mg/m^3) w/ tubing	Difference DT 2 (w/o-w)	% Difference DT 2 (w/o-w)	SD of Difference
0.359	0.272	0.087	24.2%	0.132
Mean DustTrak 3 (mg/m^3) w/o tubing	Mean DustTrak 3 (mg/m^3) w/ tubing	Difference DT 3 (w/o-w)	% Difference DT 3 (w/o-w)	SD of Difference
0.253	0.179	0.074	29.3%	0.110

In addition, the data sets for each DustTrak were compared with and without the tubing using the Wilcoxan (Mendenhall, 1971) non-parametric ranking test (in this test the data do not have to be normally distributed). For all three DustTraks the data sets were shown to be statistically different. The loss of PM_{10} due to the tubing, therefore, is significant and should be considered in future PM_{10} data analysis.

However, we did not evaluate sampling line losses for any pollutants analyzed in this report, or make corrections to our data to account for sampling line losses because sampling line losses will be highly dependent on the particle size distribution and therefore will vary depending on the characteristics of the PM being sampled, and because we believe any losses were small. Specifically, the estimates of sampling line losses given here for PM_{10} represent a worst-case, upper-limit. The PM sampling lines we used inside the buses ranged from a few centimeters (integrated $PM_{2.5}$ mass both front and rear) up to 1and 1.5 meters in length at the front and rear of the bus, respectively, for real-time $PM_{2.5}$ mass and fine particle counts in the range of 0.3 – 0.5 μm in diameter, the PM size ranges our analyses focused on (see Section 4.1.2.2.2 for a discussion of sampling line lengths). Based on results from other studies, we expect lower sampling line losses for particles in these size ranges compared with PM_{10}. In addition, we also expect the shorter sampling lines used inside the buses compared with the study cited above would result in lower sampling line losses. Moreover, it should be emphasized that most of our analyses were based on relative comparisons between buses and did not use absolute concentrations or particle counts. In summary, we expect particle losses were small, particularly for measurements made inside the buses, since we focused our main study analysis on particles < 2.5 μm in diameter.

5.1.5 Autocorrelation

Autocorrelation, also known as serial correlation, is the statistic that measures the correlation of a variable with itself over time. It can be calculated for lags (k) of any size, where the effective sample size (n) of paired values will decrease by one of each lag.

Assessing the autocorrelation is important since data sets with this attribute are more complex to analyze. Several statistical procedures, such as significance tests, assume the data are free of autocorrelation (i.e., the series of observations are independent of one another). Among other effects, autocorrelation can severely reduce the effective sample size and degrees of freedom in a time series.

In any time series containing non-random patterns of behavior, it is likely that any particular item in the series is somewhat related to other items in the same series. For example, ambient air data are, in general, likely to exhibit autocorrelation. Autocorrelation can be measured by estimating the sample autocorrelation coefficients (r_k), where values of $\pm 2/\sqrt{n}$ denote autocorrelations significantly different from zero (Norusis, 2002). We used equation (5.1.5.1) to compute the simple autocorrelation:

$$\frac{\sum[(x_i - \bar{x})(y_i - \bar{y})]}{\sum(y_i - \bar{y})^2} \qquad (5.1.5.1)$$

where y is the value of the object series, x is a lagged value of it, and $\bar{y}$ and $\bar{x}$ are the respective means; there are n-k terms in the numerator summation, and n terms in the denominator summation. Where simple autocorrelations are computed, no account is taken of the correlation between other pairs of entries.

An alternative method for assessing autocorrelation is via the partial autocorrelation coefficient (for order k), which measures the strength of correlation among pairs of entries in the time series while accounting for (i.e., removing the effects of) all autocorrelations below order k. For example, the partial autocorrelation coefficient for order k=5 is computed in such a manner that the effects of the k=1, 2, 3, and 4 partial autocorrelations have been excluded. The results of this type of analysis are often portrayed graphically in the form of a correlogram, which is a plot of the autocorrelation coefficient (simple or partial) against the lag.

We generated simple autocorrelation correlograms and partial correlation correlograms for all of the exposure runs. Figures 5.1.5.1 to 5.1.5.3 illustrate examples of partial correlograms obtained for three pollutants. The partial autocorrelation correlograms exhibit non-zero coefficients for only one or two lags, beyond which they are not significantly different from zero. Therefore we assumed insignificant autocorrelation among these variables. Similar results were obtained for most of the runs and, in general, our data did not exhibit significant autocorrelation. Hence, we did not perform any type of transformation or statistical treatment to account for autocorrelation during our data analysis.

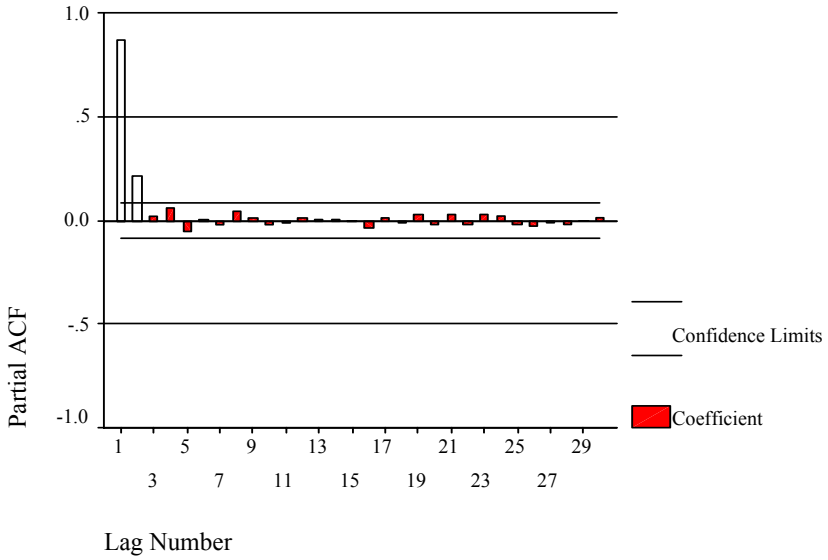


Figure 5.1.5.1 Partial autocorrelation correlogram for black carbon during Run 6.

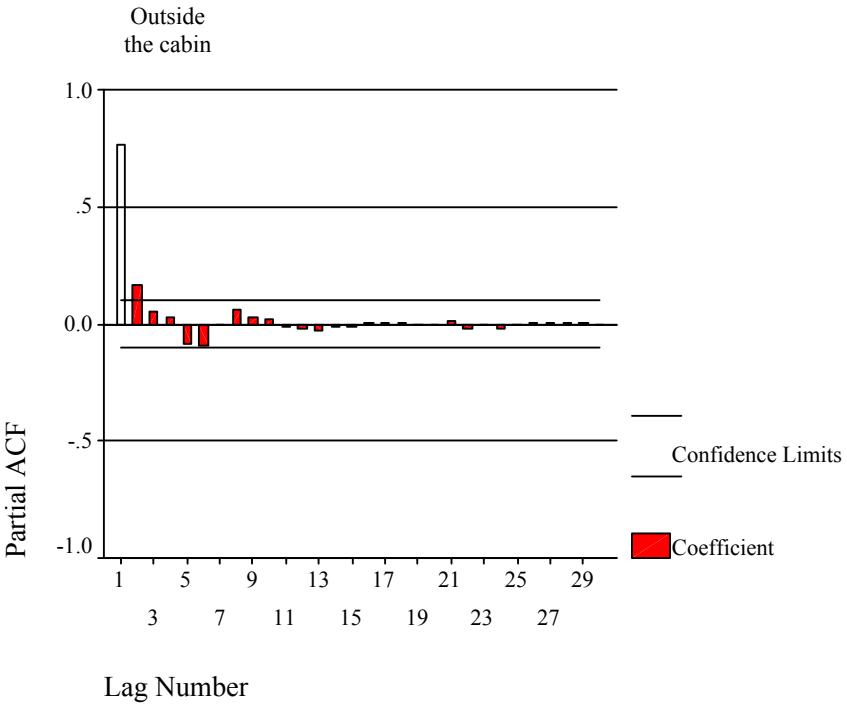


Figure 5.1.5.2 Partial autocorrelation correlogram for PAH during Run 6.

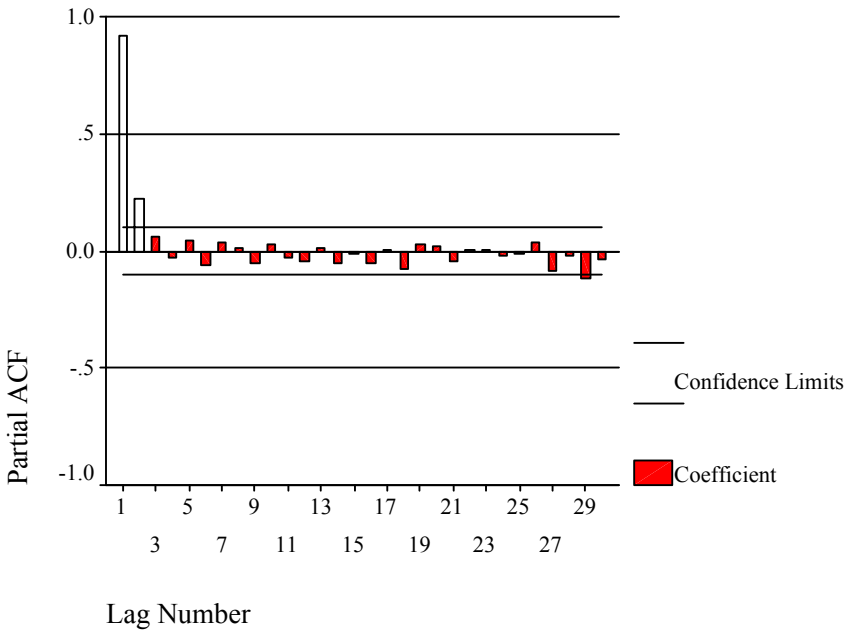


Figure 5.1.5.3 Partial autocorrelation correlogram for $PM_{2.5}$ during Run 6 (data collected with the DustTrak instruments).

One possible explanation for the relative lack of autocorrelation in our data is related to the frequent presence of high-emitting vehicles in the surroundings of the bus. Encounters with heavy duty vehicles (particularly other school buses) occurred randomly many times during the runs and, as described in Section 5.3.5, the presence of these high emitting vehicles was one of the most important factors determining pollutant concentrations inside the cabin; hence, the time series were, to some extent, dominated by these high emissions.

It was clear from the fieldwork that encounters with high emitting vehicles were largely random occurrences except when we were caravanning with other BSMS buses at the beginning of the afternoon runs. Although events such as red lights, stop signs, and bus stops had the potential to be not entirely random, the autocorrelation analyses suggest their effect was not as important as the random encounters with high emitters.

The analysis described above addresses only the autocorrelation within runs. It does not deal with the autocorrelation between runs if we combined them for a particular analysis. For example, it could be interesting to combine data for all morning runs and compare them to the combined data for all afternoon runs but this procedure is complicated. Combining all morning runs, for instance, would create an interrupted time series, which requires the use of pooled time-series cross-section models (TSCS) in order to assess autocorrelation. These methods are complex and were deemed to be beyond the scope of this report.

In summary, we did not use time series analysis for data sets comprised of more than one exposure run. However, combining data from different runs was required for other types of analyses such as comparisons between bus types and bus routes. Assuming the data distributions

were similar from runs conducted with the same type of bus under similar traffic and environmental conditions, and considering the lack of autocorrelation within runs as discussed above, we were able to combine data from different runs to calculate means and confidence intervals and then used these results to determine significant differences between the analyzed variables (e.g., bus types).

5.1.6 Normality

Most statistical procedures assume the distributions of the variables under consideration come from a normal population, or the sample sizes are large enough so the distributions of sample means are approximately normal. Our data sets were, in general, large enough to satisfy this assumption (greater than 50); however, we ran formal tests to determine if the pollutant concentration time series fit normal distributions.

We used the Kolmogorov-Smirnov and Shapiro-Wilk Criteria and found most of the data sets (dominated by the bus runs) failed the normality tests (the only exception being the black carbon data collected during Run 19). Thus, the data obtained during the bus runs were generally not normally distributed. The logarithmic transformation is a widely used procedure that could, in principle, apply to our case since many studies have shown ambient air concentrations tend to follow log-normal distributions. We used the base 10 transformation and repeated the tests. However, even after applying the logarithmic transformation, the data related to the bus runs were not normally distributed.

Figure 5.1.6.1 shows the histogram of the data collected with the $PM_{2.5}$ analyzer (DustTrak) located at the rear of the bus during Run 14 (RE1-AM). The distribution of these data is a representative example of the majority of the bus run data for most pollutants. Figure 5.1.6.1 exhibits a somewhat tri-modal distribution where each mode could be related to different conditions experienced during the bus runs. One possible interpretation of these results is that the first mode (low concentration) was related to low-traffic areas in West Los Angeles, the second mode was related to higher traffic-density streets in the inner city, and the third mode (high concentration) was related to concentration peaks resulting from self pollution or encounters with high emitting sources in both low-traffic and high-traffic areas (see Section 5.3.5). Proving this hypothesis would require a series of thorough and detailed analyses that were beyond the scope of the present project. However, this would be an interesting topic for further study.

As mentioned above our data sets were in all cases sufficiently large (greater than 50 data points) that the violation of the normality assumption was not a factor for the type of analyses we performed. According to R. Berk (Personal Communication, 2002), for this dataset, basic analyses were the most useful. We therefore focused our analyses on the use of confidence intervals and central tendency measures. For these analyses, the means are statistically significantly different if the confidence intervals do not overlap (although the reverse is not necessarily true). As explained in Section 5.1.5, in general, our data set did not exhibit significant serial correlation, which again is consistent with the use of confidence intervals (which assume independent samples).

Finally, it is noteworthy that if future analyses of our data involving more advanced statistical tools were to be developed, it would be necessary to reconsider the issues of normality and serial correlation.

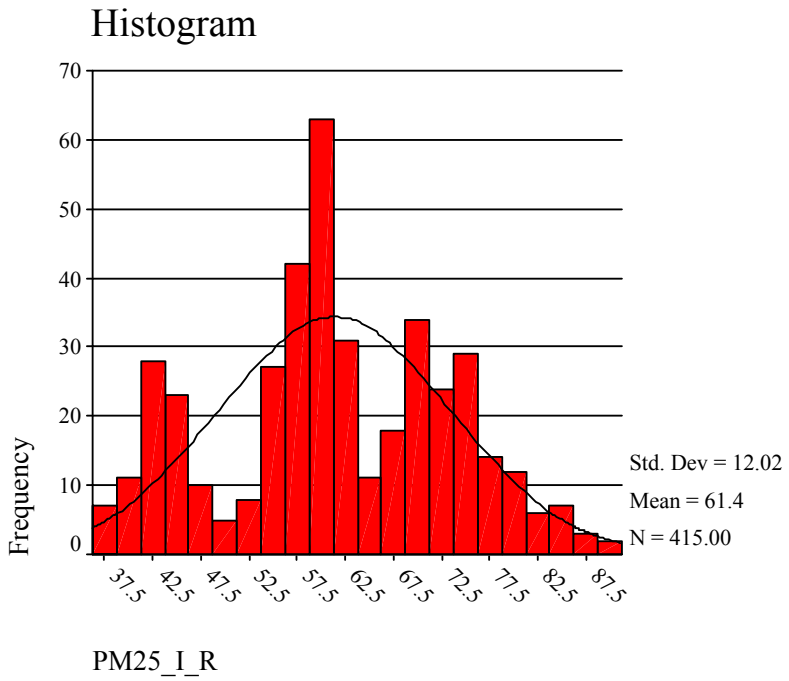
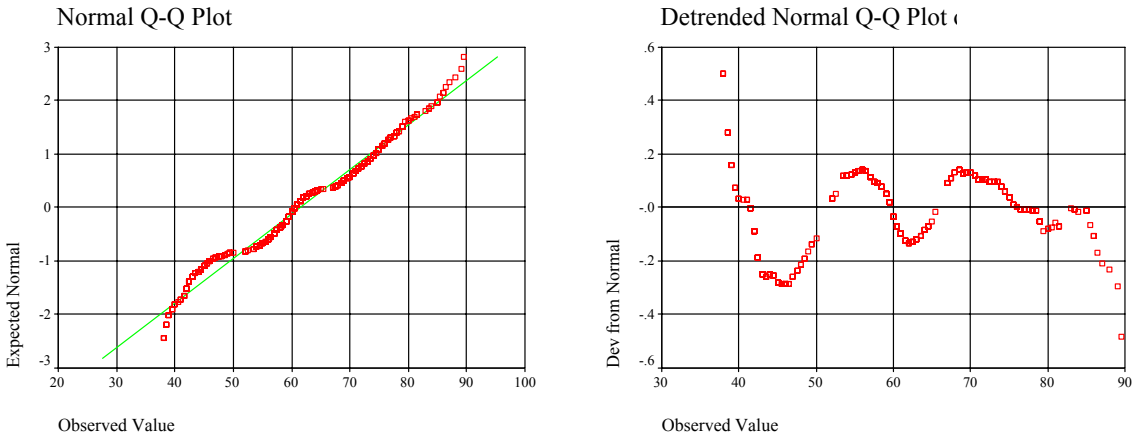


Figure 5.1.6.1 Histogram of $PM_{2.5}$ during Run 14.

Figures 5.1.6.2 show the normal Q-Q plot and the de-trended normal Q-Q plot, respectively, for $PM_{2.5}$ during Run 14. These representations can be used to further describe the not-normal characteristics of the data sets collected during the bus runs. Figure 5.1.6.2b, in particular, re-emphasizes the tri-modal characteristics of the observed data distributions.



(a) (b)

Figure 5.1.6.2 Normal Q-Q plots of $PM_{2.5}$ during Run 14 (data collected with the DustTrak instruments).

5.2. Relative Contribution of School Bus-Related Microenvironments to Children's Exposure

As described in previous sections, three microenvironments were investigated concerning children's pollutant exposure due to school bus-related activities: urban bus routes, bus stops, and a loading/unloading zone at BSMS. Specifically, two urban routes to the BSMS were investigated; and two of the stops along urban route one (U1) were studied (Vermont Elementary School and Weemes Elementary), where diesel buses arrived to pick up or drop off students traveling to or from other schools, and a large number of parents dropped off or picked up children using their personal vehicles. The loading/unloading zone was investigated during the pilot study by conducting measurements with an instrumented van parked at the sidewalk of the BSMS. During the main study, the same instruments were used for all bus stop and bus commute measurements.

Calculations of pollutant-specific exposure factors, defined as the product of the average concentration for a specific pollutant times the time spent by children in the particular microenvironment, were made for the three microenvironments, based on pilot study data. These initial calculations suggested the relative importance of bus commutes versus bus stops versus the loading/unloading zone were quite different. Specifically, from the pilot study data, we found the exposure contributions of the bus stop and loading/unloading zone were about 3% and 1%, respectively, of the bus commute exposures. We concluded the loading/unloading zone microenvironment is generally not as important as the other two (in part because of the small amount of time children spent at the loading/unloading zone).

In order to confirm these findings, we conducted measurements at an additional bus stop during the main study, in part because a dirty bus was involved in the pilot study. Cleaner buses, of the kind used in several runs of the main study were expected to exhibit lower commute exposures, potentially leading to a greater exposure contribution from bus stops.

However, except for black carbon, average commute concentrations were higher during the main study (see Table 5.2.1). These results could be explained by several factors, including differences in meteorological conditions between the two sampling periods. The pilot study was conducted during winter time, when wind speeds and ventilation of the basin are generally higher than during the spring/summer season, when the main study was conducted. During the main study, meteorological conditions were consistent throughout the eight weeks of the sampling period in May and June.

Another potential explanation could be the shortening of the route used during the main compared with the pilot study. The pilot study route started at 6:00 A.M. on small residential streets with almost no traffic for a considerable part of the commute. This may have reduced the average concentrations during the morning commutes with windows closed despite involving a "dirty" bus. Finally, it is also possible concentrations averaged over the entire main study period were dominated by the high concentrations resulting from the several high emitter buses employed.

Table 5.2.1 summarizes the mean concentrations observed in the three microenvironments, over appropriate periods of time, for both the main and pilot studies. These

values are commonly referred to in the literature as average exposure (Monn, C., 2001). The values reported for the loading/unloading zone runs during the pilot study and for the collocation experiments during the main study may be considered as urban background concentrations for the times and locations at which the experiments were conducted.

Table 5.2.1 Mean concentrations in three microenvironments in pilot and main studies.

	Average Concentrations			
	Collocation[1]	Loading/ unloading Zone[2]	Bus Stops	Bus Commutes[3,4]
BC (ug/m^3)	2	2	4	3 - 19 (8)
PAH (ng/m^3)	23	15	47	64 - 400 (134)
NO_2 (ppb)	64	35	54	34 - 110 (73)
PC ($\#/cm^3$)	72	8	62	70 - 212 (130)
$PM_{2.5}$ (ug/m^3)	23[5]	N/A	26	21 - 62 (43)

[1] These values were measured during collocation experiments and likely typify urban background concentrations during the main study

[2] These measurements were conducted during the pilot study and likely typify urban background concentrations during the pilot study

[3] The ranges are associated with the different bus types (see Table 5.3.3.1)

[4] The values within parentheses are the averages for all runs

[5] 2001 annual arithmetic mean from AQMD station at Central Los Angeles

Table 5.2.1 includes the results of the pilot study but does not include the rural/suburban route data from the main study. The relevance of this latter microenvironment is described below (see Section 5.3.5.2.4).

The results obtained during the main study were, in general, similar to those obtained during the pilot study except for PC, for which the average concentration measured at the main study bus stop (Weemes ES) was about two times the average concentration measured at the pilot study bus stop (Vermont ES).

Similarly, the average PC concentration inside the bus during main study urban route commute was about three times the average concentration measured inside the bus during the pilot study commutes. The majority of these differences could be explained by a change in the particle counts analyzer; during the main study we used a more sophisticated device than in the pilot study.

The average BC concentration inside the cabin for the pilot study route (14 $\mu g/m^3$) was about two times that measured for the main study urban routes (8 $\mu g/m^3$). The BC results, both for the pilot and main studies were consistent with the findings of a recent Natural Resources Defense Council (NRDC) study (Solomon et. al., 2001). The NRDC study also found higher concentrations in the rear of the bus and when the windows were closed, similar to our results as described in Sections 5.3.6.2 and 5.3.2.

Table 5.2.1 shows the loading/unloading microenvironment in front of BSMS was quite clean compared to the urban route commutes or bus stops, with the average concentration observed at the BSMS within, and slightly below, the range of urban background values reported in the literature.

We calculated the ratios of the average concentrations in the bus runs and stops relative to the loading/unloading zone microenvironment and these results are presented in Table 5.2.2. Average concentrations at the bus stop microenvironment were between two and eight times higher than at the loading/unloading zone microenvironment (dependent upon the pollutant) while average concentrations inside the bus on the urban routes were between 2 and 16 times higher than at the loading/unloading zone. Average concentrations inside the bus on the urban routes were on average 1.3 to 2.8 times higher than at the bus stops.

Table 5.2.2 Ratios of mean concentrations inside the bus and at bus stops relative to the loading/unloading zone.

	Ratios of Average Concentrations (relative to L/U)		
	Loading/ unloading	Bus stops	Urban routes (main study)
BC	1	2	4
PAH	1	3	9
NO_2	1	2	2
PC	1	8	16
$PM_{2.5}$	1	N/A	N/A

Table 5.2.3 shows our estimations of the time spent in each microenvironment. The commute time, or time spent inside the bus was the mean duration of the urban commute routes used during the main study (in the case of U1 representing the midway point between the first and last child on the full route used in the Pilot Study--see Section 4.1.5.1). The loading/unloading and bus stop times were estimated from observations in the field. These values are consistent with the results reported in the South Coast School Bus Idling Study (Solomon et. al., 2002). In our study, time spent in commuting on the bus was between 15 and 38 times greater than for the other two microenvironments.

Table 5.2.3 Estimated time spent in each microenvironment.

Microenvironment	Time spent (min.)
Loading/unloading (each time)	3
Bus stops (each one)	2 - 5
Urban route (each way)	76

Table 5.2.4 shows the exposure factors resulting from the product of the average concentration and the time spent in each microenvironment. Across all pollutants the exposure during bus commutes on urban routes was much higher than for the other two microenvironments. These results confirmed the findings of the pilot study that in general the loading/unloading microenvironment is the least important in terms of school bus related exposure, both because children generally spend a short time on the sidewalk (three minutes or less) before entering the school in the morning or boarding the buses in the afternoon, and because bus drivers were required by school district policy to turn off their engines as soon as they arrived in the morning, before the children left the buses. Similarly, in the afternoon, drivers are not supposed to turn on their engines before all children are aboard the buses and the entire fleet is prepared to depart. During both the pilot study and the main study we observed a near 100% of compliance with these regulations in the morning with somewhat less than full compliance in the afternoon, due to buses leaving while some children were still waiting to board. However, we have anecdotal evidence from a BSMS teacher that this policy is not always rigorously enforced (Fogel, 2002). In addition, according to a recent NRDC study (Solomon et. al., 2002), lack of explicit idling policies is common in Los Angeles area district schools.

During our monitoring periods at both the Vermont Avenue School and the Weemes Elementary School bus stops, at various times as many as half a dozen buses would pull up in front of the school and wait with the engine idling until children boarded. In several cases, buses were early and waited with their engines idling for several minutes, while students waiting for other buses stood nearby. Buses would then accelerate away from the curb, often releasing an exhaust cloud of black smoke. Similarly, parents dropping off or picking up children at those schools serving as bus stops, in a number of cases were driving what appeared to be high emitter vehicles (as documented by us in our field notebooks).

Table 5.2.4 Mean exposure factor calculations.

	Exposure Factors		
	Loading/ unloading	Bus stops[1]	Urban commutes
BC (ug/m^3-min)	5	8 - 20	600
PAH (ng/m^3-min)	45	95 - 230	10000
NO2 (ppb-min)	105	110 - 270	5500
PC (#/cm^3-min)	25	125 - 310	10000
PM2.5 (ug/m^3-min)	N/A	50 - 130	3500

[1] The ranges are associated with the different values for time spent in each microenvironment used in the calculations (see Table 5.2.3)

The calculated ratios of exposure factors for the three microenvironments and these results are summarized in Table 5.2.5.

Table 5.2.5 Exposure ratios for the three microenvironments.

	Exposure Ratios		
	Bus stops / L/U [2]	Urban commutes / L/U [2]	Urban commutes / Bus stop[1]
BC	1.3 - 3.5	100	30 - 75
PAH	2.1 - 5.2	220	45 - 110
NO2	1.0 - 2.6	50	20 - 50
PC	5.2 - 13	400	30 - 80
PM2.5	N/A	N/A	25 - 70

[1] The ranges are associated with the different values for time spent in each microenvironment used in the calculations (see Table 5.2.3)

[2] L/U = loading/unloading zone

In terms of exposure, the bus stops microenvironment was between one and thirteen times more important than the loading/unloading microenvironment, while the urban bus commutes were between 50 and 400 times more important than the loading/unloading microenvironment. The urban bus commutes led to between 20 and 100 times higher exposures than at the bus stops, depending upon the pollutant.

5.3 Important Variables Governing Exposure During Bus Commutes

The design of this study allowed us to investigate the importance of multiple variables governing children's exposure to vehicle-related pollutants during school bus commutes. The variables analyzed in this study included contribution of the bus's own exhaust, window position (open versus closed), bus type (based on engine type and age, fuel, and after-treatment technology), route (rural/suburban versus urban and between urban routes), roadway type (freeway versus non-freeway), the effect of following other diesel vehicles, the effect of idling, position within the bus (front versus rear) and the difference between inside and immediately outside the bus. Only the front/rear and inside/outside comparisons involved using paired instruments. All other variables were evaluated based on measurements taken at the rear of the bus cabin with the same instruments (e.g. the same Aethalometer was used for all black carbon comparisons between buses, etc.).

The exposure runs on HE1, performed during the first week of the main study, were different than those conducted in subsequent weeks for a number of reasons, and thus were generally not included in our analyses. First, the instruments on the bus were powered by a

propane-fueled generator which was towed behind the bus, as was done in the pilot study. This created several problems. During the rural/suburban run, the generator became disconnected from the bus, likely due to the vigorous motion of the bus at higher speeds. During the morning run on the primary urban route, the muffler fell off the generator while the bus was traveling on the freeway, which resulted in the bus sitting on the side of the freeway for several minutes with the windows open and the instruments still measuring pollutants in the bus. In addition, during these runs, the window position on the bus was not fixed. We alternated between windows opened and windows closed for both runs on the primary urban route. By contrast, the window position was fixed on all subsequent buses, with windows in the morning always closed, and windows in the afternoon always partially opened, reflecting the practices we observed on in-use school buses. Finally, during the afternoon run, the BSMS let out one hour earlier than usual on this day. In an attempt to mimic regular BSMS bus commutes, we decided to run the route at this earlier time because we felt it was important to capture the effect of caravanning (which occurs when leaving the school with the nineteen BSMS buses as shown in Photograph 11). However, without the muffler, the noise from the generator was so great the BSMS officials would not allow our bus to wait in front of the school and we had to leave immediately, without waiting for the other buses. Because the run started an hour earlier than usual, traffic conditions on the route were considerably lighter than encountered at the regular time. Because of these difficulties and confounding factors, we felt the exposure runs on HE1 should not be included in the majority of our comparisons with the other buses. However, these runs were still valuable because they provided data on bus commutes during periods of lighter traffic and allowed us to perfect our sampling strategy for the subsequent bus runs.

For the analyses described in the remainder of Section 5.3, only data from the second through eighth weeks of the study were used (e.g. HE2, HE3, RE1, RE2, TO1, CNG).

Photograph 11. Caravanning after departing from BSMS during an afternoon run (line of 5 buses).

5.3.1 Determination of Contribution of Bus's Own Exhaust Inside the Cabin

Because a potentially important source of children's exposure to pollutants from vehicle exhaust is the introduction of a bus's own exhaust into the passenger compartment, our study included a method to determine the percentage of air in the bus cabin originating from the bus's

own exhaust. A tracer gas, SF_6, was metered into the bus's exhaust system from a high concentration cylinder using a mass flow controller whose flow rate was logged by the data acquisition system and processed with the rest of the measurement data. The concentration of SF_6 and the flow rate were selected to detect if at least 0.1 percent of the air in the passenger compartment of the bus was from the vehicle's exhaust. We emphasize these experiments were completely independent of, and not affected by the SF_6 experiments conducted to determine the ventilation rates in the passenger compartment (see Section 4.1.3).

The SF_6 injection probe extended ~ 15 cm into the bus's exhaust pipe in order to provide reasonable mixing of the SF_6 without attempting to snake the injection probe around the bends in the exhaust system. At the injection point, the maximum exhaust temperature of the bus was expected to be about 200°C, well below the temperature at which SF_6 decomposes.

The SF_6 concentration in the bus was measured on a continuous basis using a CTA-1000 continuous tracer gas analyzer whose inlet was connected to a series of solenoids that switched the sample inlet between the front and rear of the bus cabin. To account for baseline drift of the CTA-1000 additional solenoids switched the instrument to SF_6-free air drawn through a line located outside the bus next to the right rear-view mirror. Although this third sample line generally provided a reference zero value, it also yielded data indicating that under the right wind conditions (i.e., wind from the rear) when the bus was stopped and idling, significant amounts of the bus's own exhaust reached the location next to the passenger door at the front of the bus.

Control of the switching and logging of the solenoid position was performed by the data acquisition system and switching times from 2-4 minutes were used during the initial bus runs. After reviewing the system's response, a switching time of four minutes per position (twelve minutes per cycle) was determined to provide the optimum data and this switching time was used for all subsequent buses. When the solenoids switched from one position to the next, they generated a pressure pulse in the sample line that temporarily threw the CTA-1000 off scale. The instrument's recovery time from the solenoid switching was determined to be close to three minutes. As a consequence, the first three minutes of data after each solenoid position change were voided and only the final minute of data was employed in the following analysis.

5.3.1.1 Analysis of SF_6 Data

Figure 5.3.1.1 depicts the conditions at the point of injection (A):

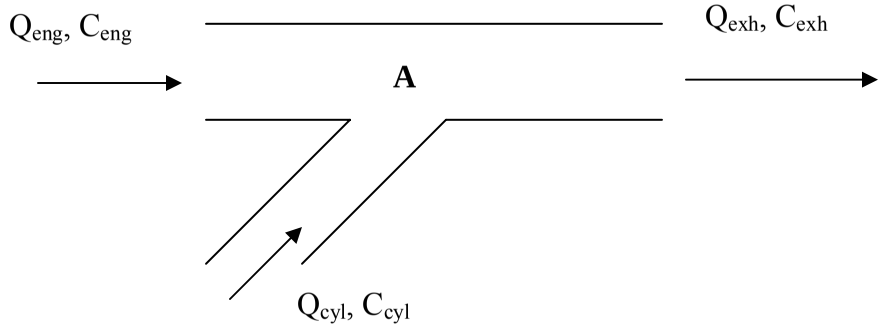


Figure 5.3.1.1 Schematic of the SF_6 injection point.

where:

Q_{eng} : Flow rate of the engine gases
C_{eng} : Concentration of SF_6 in the engine gases (zero)
Q_{cyl} : Flow rate of the high-concentration SF_6 cylinder
C_{cyl} : Concentration of SF_6 in the cylinder
Q_{exh} : Flow rate of the exhaust (end of tailpipe)
C_{exh} : Concentration of SF_6 in the exhaust

The mass flow rates were expressed as:

$$M_{cyl} = Q_{cyl} \bullet C_{cyl} \qquad (5.3.1.1)$$

where:

M_{cyl} : Mass flow rate of SF_6 from the cylinder

NOTE: similar equations may be written for the mass flow rates from both, the engine and the exhaust.

From the mass conservation principle (at the point of injection) we know

$$M_{exh} = M_{eng} + M_{cyl} \qquad (5.3.1.2)$$

considering 5.3.1.2 and solving for C_{exh} yields

$$C_{exh} = \frac{Q_{cyl}}{Q_{exh}} \bullet C_{cyl} \qquad (5.3.1.3)$$

To estimate the concentration of SF_6 in the exhaust, we need to know the flow rate of exhaust which is given by equation (5.3.1.4), a commonly used expression in mechanical engineering (Western Filter Co, Inc., 2002)

$$Q_{exh} = \frac{E_{rpm} \bullet D \bullet V_{eff}}{C} \qquad (5.3.1.4)$$

where:

Q_{exh} : Flow rate of exhaust gases
E_{rpm} : Engine's revolutions per minute
D : Engine's displacement
V_{eff} : Volumetric efficiency
C : Engine's cycle factor

For a turbocharged diesel engine, the average volumetric efficiency (measure of the charge that enters the cylinder compared to ideal conditions) is about 1.5. Two-cycle engines displace their engine size every revolution while 4-cycle engines displace their engine size every two revolution; therefore, the engine's cycle factor (C), a measure of the number of spark-plug fires per revolution, is one and two for 2-cycle and a 4-cycle engines, respectively. Five of the seven buses used in the study had 4-cycle engines and the buses used in the study had engine displacements ranging from six to 10 liters. Diesel engines typically operate around 1500 rpm for a variety of speed/gear combinations (this number was selected after reviewing several diesel engine specifications and performance curves).

We used Run 31 to illustrate our calculations. The bus used for this run had an 8.3 liter, 4-cycle, turbocharged engine. Using equation (5.3.1.4) and the values mentioned above we estimated the average flow rate of exhaust gases during Run 31 to be approximately 9000 liters per minute.

$$Q_{exh} = \frac{1500\,\frac{rev}{min} \bullet 8.3L \bullet 1.5}{2\,\frac{1}{rev}} \cong 9000\,\frac{L}{min}$$

The release rate of SF_6 from the high-concentration gas cylinder varied between runs and even within runs. The release rate during Run 31 was about 2.3 liters per minute. The concentration of SF_6 in the canister was 5000 ppm for the first five weeks of the study and 10000 ppm for the last three weeks. During Run 31, we used a 10000 ppm cylinder.

Using equation (5.3.1.3) we estimated the concentration of SF_6 in the exhaust during Run 31 to be approximately 2.5 ppm (the average concentration of SF_6 measured inside the cabin during Run 31 was about 1000 ppt or 1 ppb).

$$C_{exh\text{-}exhaust} = \frac{2.3\,\frac{L}{min}}{9000\,\frac{L}{min}} \bullet 10000ppm \cong 2.5ppm$$

The ratio (R), which represents the percentage of pollutant concentration inside the cabin originating from the exhaust, or the concentration of SF_6 in the cabin divided by the concentration of SF_6 in the exhaust was

$$R = \frac{C_{cabin}}{C_{Exhaust}} = \frac{1000ppt}{2.5ppm} \bullet \frac{1ppm}{1x10^6\,ppt} \cong 0.04\% \qquad (5.3.1.5)$$

Table 5.3.1.1 summarizes the results obtained for all of the bus runs (except from the first week) using the procedure described above for SF_6 measurements at the rear of the cabin. The results presented in this table as well as in Figure 5.3.1.2 show the percentage of air inside the cabin originating from the exhaust were a function of bus type as well as window position. Values were substantially higher during morning runs, when all the windows of the bus were closed compared with afternoon runs, when the windows were partially opened. Thus, the SF_6 data suggest exposure to pollutants originating from the bus's own emissions were higher when

Table 5.3.1.1. SF_6 measurement results

Bus type	Displacement (L)	Cycles	Exhaust flow rate (lpm) [1]	SF_6 in the cylinder (ppm)	Run number	Median release rate (lpm) [2]	Average SF_6 concentration at the tailpipe (ppm) [3]	Average SF_6 concentration (ppt) [4]	Cabin air from the exhaust (%) [5]
HE2	6	2	13500	5000	5	1.9	0.704	922	0.13
	6	2	13500	5000	6	4.4	1.630	620	0.04
HE3	6	2	13500	5000	8	1.1	0.407	1184	0.29
	6	2	13500	5000	10	1.1	0.407	439	0.11
RE1	8.3	4	9338	5000	12	1.0	0.535	N/A	N/A
	8.3	4	9338	5000	13	1.0	0.535	N/A	N/A
	8.3	4	9338	5000	14	1.1	0.589	N/A	N/A
	8.3	4	9338	5000	15	5.5	2.945	966	0.03
RE2	6.6	4	7425	10020	17	3.8	5.128	1026	0.02
	6.6	4	7425	10020	18	3.9	5.263	304	0.01
	6.6	4	7425	10020	19	3.9	5.263	815	0.02
	6.6	4	7425	10020	20	3.9	5.263	868	0.02
	6.6	4	7425	10020	23	3.9	5.263	997	0.02
	6.6	4	7425	10020	24	3.9	5.263	741	0.01
	6.6	4	7425	10020	25	2.7	3.644	1134	0.03
	6.6	4	7425	10020	26	2.9	3.914	N/A	N/A
TO1	8.3	4	9338	10020	28	2.0	2.146	827	0.04
	8.3	4	9338	10020	30	4.3	4.614	848	0.02
	8.3	4	9338	10020	31	2.3	2.447	801	0.03
	8.3	4	9338	10020	33	2.0	2.146	519	0.02
CNG	8.1	4	9113	10020	35	2.9	3.189	1164	0.04
	8.1	4	9113	10020	36	2.9	3.189	807	0.03

[1] Calculated using equation 5.3.1.4

[2] Calculated as the median of the 10-seconds release rate data for each run.

[3] Calculated using equation 5.3.1.3

[4] Calculated as the mean of the concentrations measured at the rear of the cabin for each run

[5] Calculated using equation 5.3.1.5

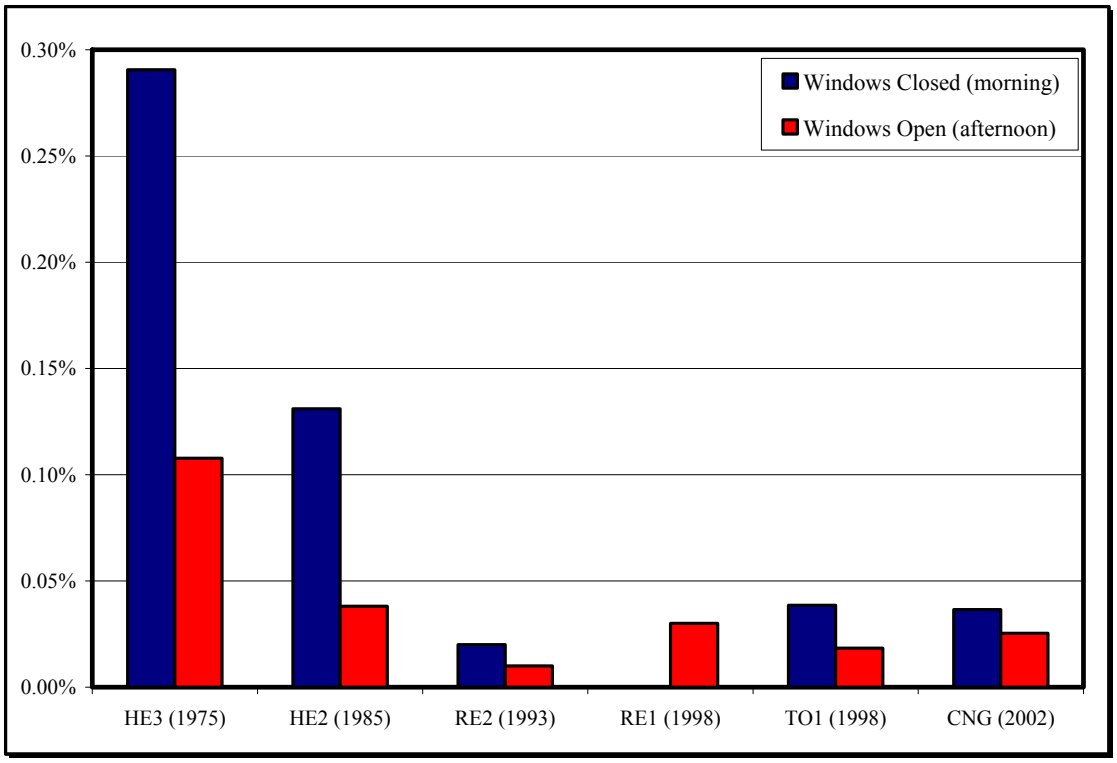


Figure 5.3.1.2 Percentage of air inside the cabin originating from the bus's own exhaust (%).

the windows were closed. It is also important to consider that, as described in Section 4.1.3, the ventilation rate of the bus cabin was determined, primarily, by the position of the windows, since each bus was checked for exhaust system leaks.

Figure 5.3.1.2 also shows the importance of bus type. The older buses HE2 (1985) and HE3 (1975) showed a larger percentage of their own exhaust entering into the cabin compared with newer buses RE2 (1993), TO1 (1998), CNG (2002). These results are consistent with the older buses not being as well isolated (airtight) from outside air compared with the newer buses, due to design or construction. However, other factors have to be considered as well.

Figure 5.3.1.3 presents the results of the snap-and-idle tests we conducted (see Section 4.1.1). The HE2 bus exhibited the highest emission factors among the buses used in the study, followed by HE3, RE2 and HE1. The TO1 and the RE1 buses were the cleanest according to those tests (it was not possible to perform the test for the CNG bus). Thus, even without considering the pollutant concentration data, we would expect exposures associated with HE2 to be among the highest for the buses tested.

The snap-and-idle test is an experiment convenient and easy to perform, however, these results can only be used as qualitative measures. We included these results in our analyses since they were the only source of information about the tested buses emissions.

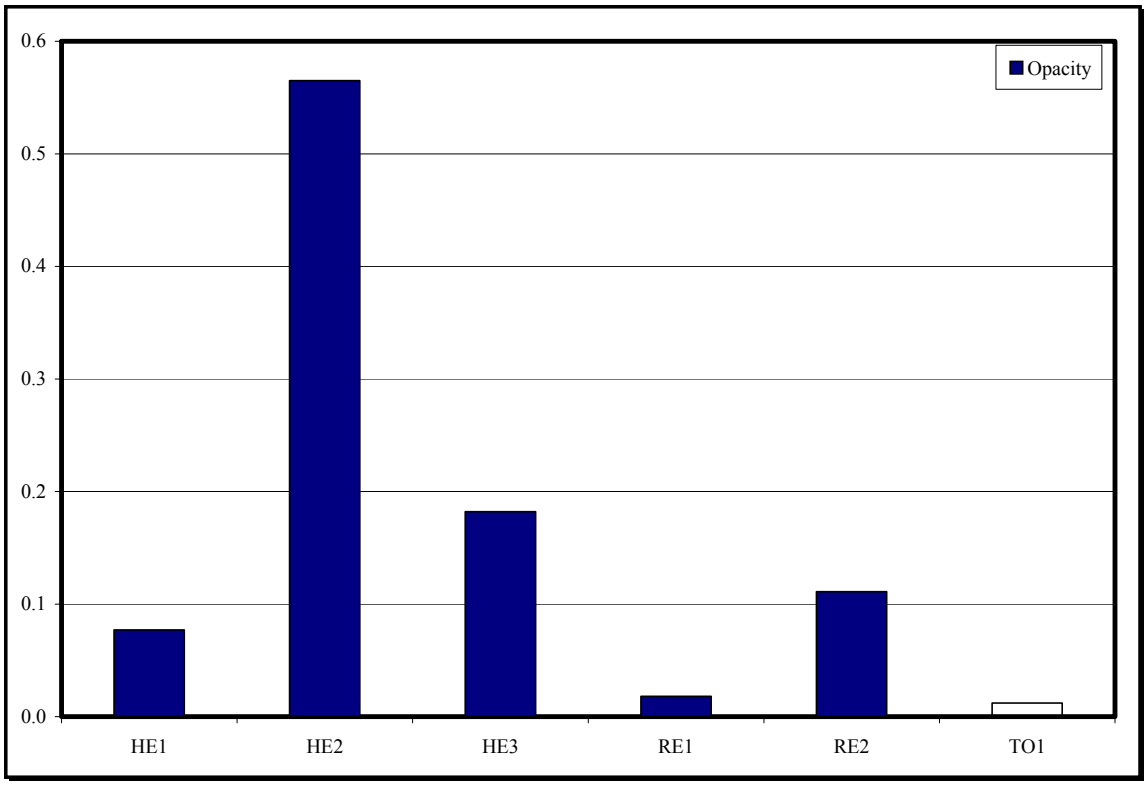


Figure 5.3.1.3 Snap-and-idle test results expressed in terms of opacity (fraction).

5.3.1.1.1 Measurements of SF_6 Concentrations

As mentioned above, three sampling positions were employed: inside/front, inside/rear and outside/front. Figure 5.3.1.4 shows a schematic of the location of these sampling probes.

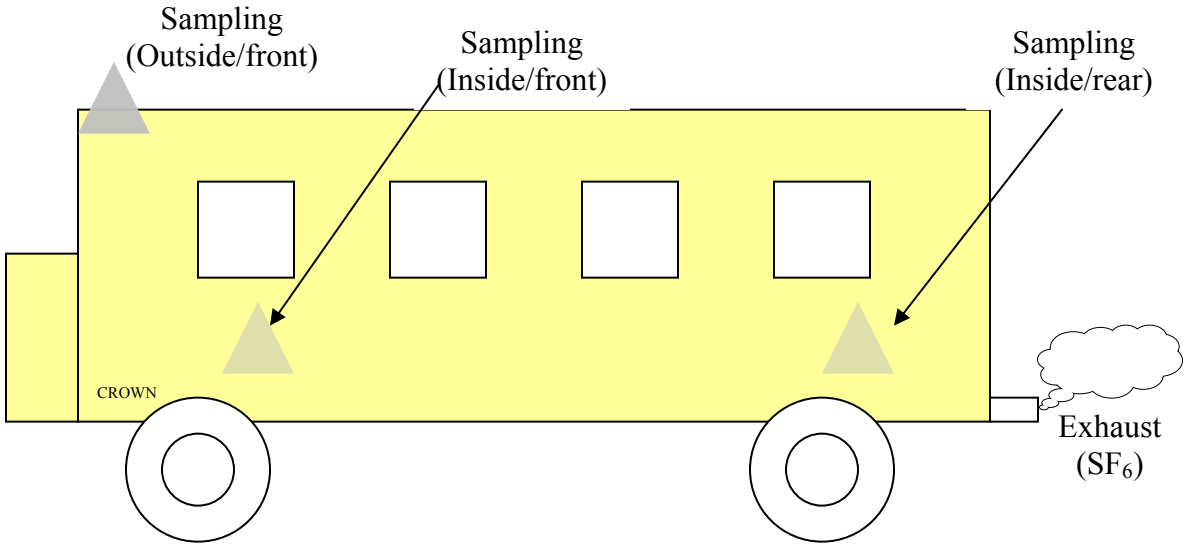


Figure 5.3.1.4 Schematic of school bus indicating the location of the SF_6 sampling probes.

Figure 5.3.1.5 shows the means and 95% confidence intervals for the average SF_6 concentrations measured at the three locations (front, rear, and outside) during the morning runs, indicating that, in general, there was a substantial difference between the measurements at the front and the rear of the cabin. Specifically, the SF_6 data indicated exposure to pollutants from the bus's own emissions were higher at the rear of the cabin than at the front. The influence of position inside the cabin is further analyzed in Section 5.3.6.2.

Figure 5.3.1.5 also shows there were detectable SF_6 concentrations outside at the front of the bus at times during all runs, although, these concentrations were much smaller than the concentrations inside the bus. Since we injected the tracer gas in the tailpipe and there cannot be any other source of it, these results show the bus's exhaust was able to reach the sampling line located at the front (outside) of the bus, for example, when buses were idling (e.g., at a red light or a bus stop) and the wind was blowing from the rear to the front of the bus. This also shows there are two different means for the exhaust gases to enter the cabin of the buses. First, leaks from the engine's compartment into the cabin and second the exhaust plume traveling from the rear of the bus and entering through the windows (if these are open). Because the SF_6 concentrations measured at the front (outside) of the bus were substantially lower than inside the bus (either front or back), the second mechanism appeared to be less important than the first.

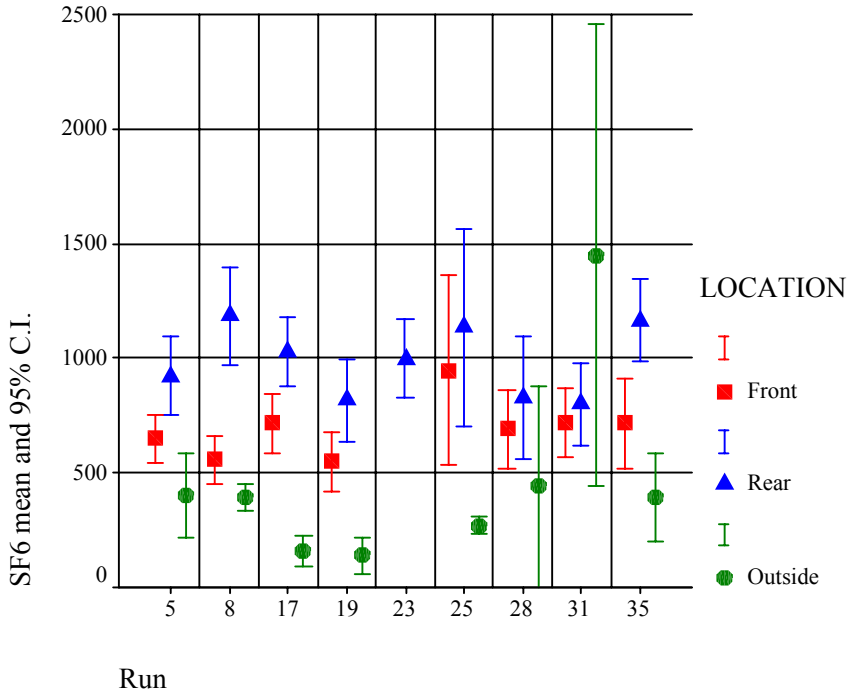


Figure 5.3.1.5 Means and 95% confidence intervals of SF_6 concentrations (ppt) measured at front, rear, and outside locations during morning runs.

Figure 5.3.1.6 shows the 95% confidence intervals for the SF_6 concentrations measured at the three locations during the afternoon runs, and these were, in general, somewhat lower than during morning runs. Figure 5.3.1.6 also shows the differences between the SF_6 concentrations measured at the three locations are smaller during the afternoon runs compared to differences measured during morning runs. Thus, when the windows of the buses were partially open, for the majority of the runs there were no significant differences between measurements at the front of the cabin, at the rear of the cabin, and outside the bus.

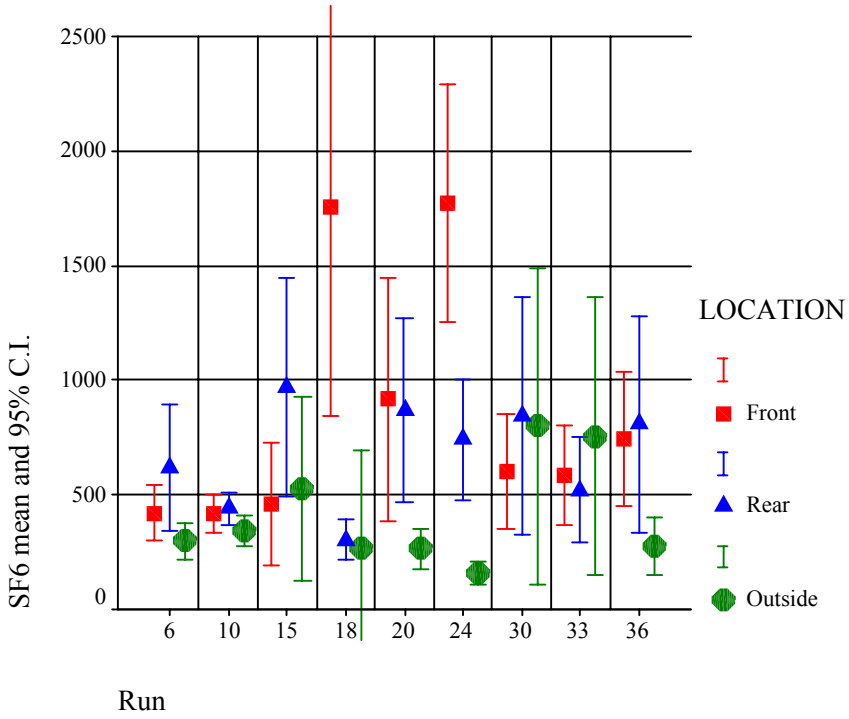


Figure 5.3.1.6 Means and 95% confidence intervals of SF_6 concentrations (ppt) measured at front, rear, and outside locations during afternoon runs.

5.3.1.2 Correlation between SF_6 versus Black Carbon and $PM_{2.5}$.

Section 5.1.3.2 describes how the correlations between black carbon and PAH were higher than those between $PM_{2.5}$ and PC (between 0.3 and 0.5 μm). $PM_{2.5}$ and PC exhibited different behavior but followed similar patterns between them, consistent with multiple other sources and a generally high elevated background of $PM_{2.5}$ (Zhu et al., 2002, a,b).

We selected black carbon and $PM_{2.5}$ (one pollutant from each of the two groups) to analyze the implications of the SF_6 data in terms of the differences between emissions among the different buses. As mentioned above, based on the results of the correlation coefficients between the analyzed pollutants (see Tables 5.1.3.8 and 5.1.3.9), we assumed black carbon and $PM_{2.5}$ concentrations may be used to explain the behavior of PAH and PC, respectively. It is important to note that although we show in Section 5.1.3 that the precision between paired Aethalometers (used to measure black carbon) was low, these results only apply to analyses which used paired instruments. The correlations between SF6 and black carbon were estimated using data from a

single Aethalometer (the rear instrument), therefore the precision estimates for paired instruments described in Section 5.1.3 do not apply to the following analysis.

To obtain the results presented below we normalized the data using the maximum concentrations. For example, during Run 6 the maximum concentration of black carbon was about 40 μg/m^3 and we divided every data point of Run 6 by this value. This procedure was applied for all runs for black carbon and $PM_{2.5}$. Since it is easier to interpret the differences between absolute quantities, rather than the differing concentrations of the different species, we eliminated the units of mass per volume.

Figure 5.3.1.7 summarizes the results obtained for the correlation coefficients between SF_6 and black carbon (normalized) and between SF_6 and $PM_{2.5}$ (normalized). To avoid confounding factors we only analyzed the morning runs in which, as mentioned elsewhere, all the windows were closed. The calculated Pearson's regression coefficients ranged from 0.1 to 0.6 for $PM_{2.5}$ whereas for black carbon they were about 0.5 for the two runs with available data. The consistent, relatively high correlations between SF_6 and black carbon suggested a stronger relationship between black carbon and bus emissions than between $PM_{2.5}$ and bus emissions.

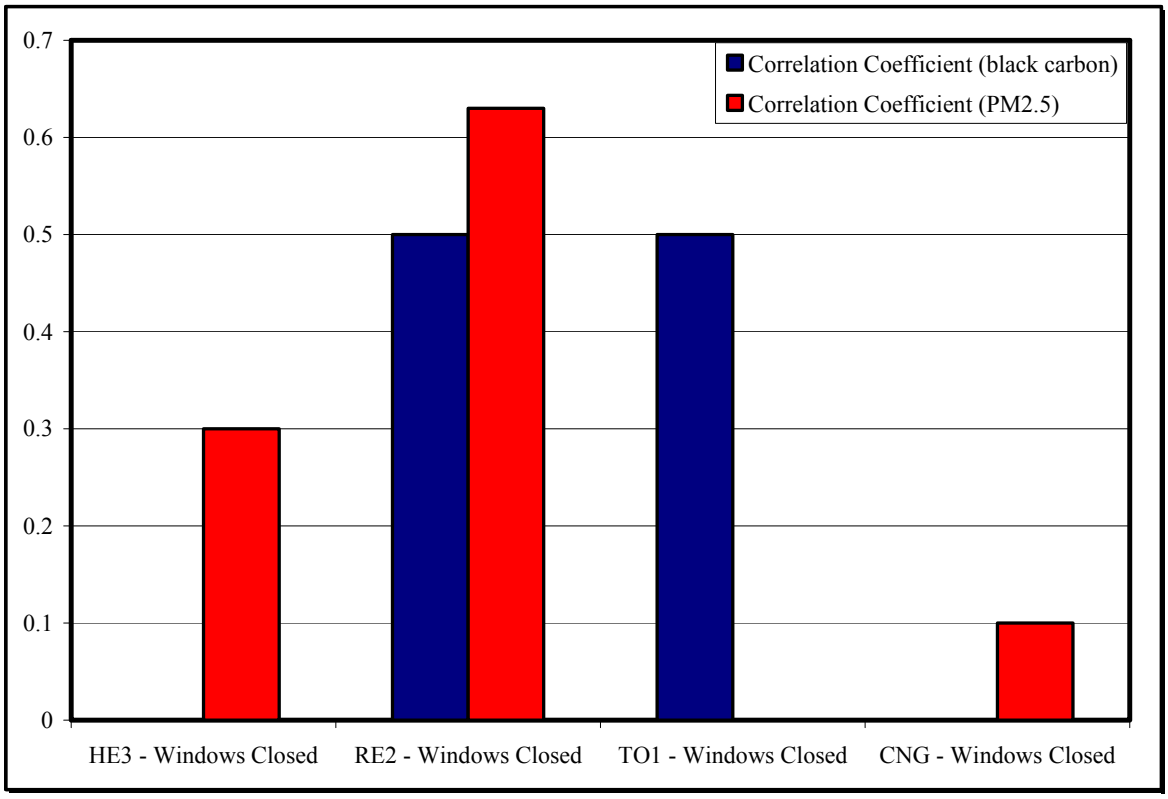


Figure 5.3.1.7 Correlation coefficients between SF_6 versus black carbon and between SF_6 versus $PM_{2.5}$ (data collected with the DustTrak instruments)

5.3.1.3 Estimation of Bus Cabin Black Carbon Concentrations Using the SF_6 Data

As just shown, among the pollutants measured during the study, black carbon correlated well with the bus's own exhaust. Since all of the SF_6 found in the cabin was from the bus's own exhaust, we could use the ratio of the concentration of SF_6 in the cabin to the concentration of SF_6 in the exhaust to obtain a rough estimate of the amount of black carbon inside the cabin originating from the bus's own exhaust.

Our first-order-approximation model started with an estimate by Miguel et al. (1998) that black carbon emissions from heavy-duty diesel vehicles are 1440 ± 160 milligrams per kilogram of fuel burned. We used the upper end of this range (1600 mg/Kg) as the emission rate for the bus used during Run 25 (RE2-AM) and assumed a typical fuel consumption rate of 10 miles/gallon. For this bus commute, the distance covered was about 20 miles and the total time of the run was about 80 minutes. From these values, we estimated the amount of fuel burned per unit of time to be about 0.025 gallons per minute.

$$\frac{20\,\text{miles}}{80\,\text{minutes}} \bullet \frac{1\text{gallon}}{10\,\text{miles}} = 0.025\,\frac{\text{gallons}}{\text{minute}}$$

We also know the density of diesel fuel:

$$\rho_{\text{Diesel}} \cong 830\,\frac{\text{Kg}}{\text{m}^3} \cong 3.15\,\frac{\text{Kg}}{\text{gallon}}$$

Thus, the fuel burned per time can be expressed as

$$0.025\,\frac{\text{gallon}}{\text{min}} \bullet 3.15\,\frac{\text{Kg}}{\text{gallon}} \cong 0.08\,\frac{\text{Kg - fuel}}{\text{min}}$$

We used this value and the emission rate reported by Miguel et al. (1998) to estimate the mass flow rate of black carbon (M_{BC})

$$\boxed{M_{BC} = 1600\,\frac{\text{mg}-\text{BC}}{\text{Kg}-\text{Fuel}} \bullet 0.08\,\frac{\text{Kg}-\text{Fuel}}{\text{min}} = 128\,\frac{\text{mg}-\text{BC}}{\text{min}}}$$

Similar to equation (5.3.1.1):

$$M_{BC} = C_{BC\text{-Exhaust}} \bullet Q_{Exh} \qquad (5.3.1.6)$$

Solving for the concentration of black carbon in the exhaust (C_{BC})

$$C_{BC\text{-Exh}} = \frac{128\,\frac{\text{mg}-\text{BC}}{\text{min}}}{9000\,\frac{\text{L}-\text{Exhaust}}{\text{min}}} \bullet \frac{1000\,\mu\text{g}}{1\text{mg}} \bullet \frac{1000\text{L}}{1\text{m}^3} \cong 14000\,\frac{\mu g}{m^3}$$

During Run 25, the percentage of air inside the cabin originating from the exhaust was about 0.03% (see Table 5.3.1.1). Hence, we estimated the concentration of black carbon inside the cabin to be about 4 micrograms per cubic meter.

$$C_{BC-Cabin} = 14000 \frac{\mu g}{m^3} \bullet 0.0003 \cong 4 \frac{\mu g}{m^3}$$

The mean of the real-time black carbon concentrations during Run 25 was about 9 $\mu g/m^3$, which is of the same order of magnitude as the estimated value of 4 $\mu g/m^3$. Agreement to within a factor of two between the calculated and measured black carbon concentrations tends to validate both the SF_6 and the black carbon data obtained during the fieldwork.

We emphasize again that the approach employed in the previous calculations was not intended to produce accurate values that can be extrapolated. The estimates were meant to be only first-order approximations aimed at validating the data gathered by two different instruments, as well as identifying major trends in our data set.

5.3.2 Effect of Window Position

The position of the windows was one of the most complex variables analyzed in this study. Window position contributed importantly to the ventilation rate inside the cabin leading to substantially different exposures. Assessment of the influence of the window position required consideration of other variables such as position of the sampling probe (inside versus outside) and its location inside the cabin (front versus rear) as discussed in Sections 5.3.6.3 and 5.3.6.2, respectively.

Due to the relevance of windows position, demonstrated in part during the pilot study, we performed three additional tests (Runs 9, 29, and 32) specifically designed to test the effect of the window position (See Section 4.1.7) on the concentrations inside the bus.

These experiments were performed on the I-405 freeway between 8 and 9 AM, traveling south from Wilshire Avenue (close to UCLA) to Century Blvd (close to LAX) and then returning north to UCLA on I-405. During these runs we experienced substantial differences in congestion in the north/south directions. Since differences in average speeds led to different ventilation rates, we effectively tested the effect of window position for different ventilation rates.

We utilized alternating cycles of about seven minutes. During the first cycle all windows were closed. During the second cycle every other window was opened about 10 inches on the driver's side and about two inches on the opposite side, where the instruments were located. Table 5.3.2.1 summarizes the characteristics of the window position experiments.

Table 5.3.2.1 Window position tests.

Run	Date	Time	Type of Bus	Week Number
9	05/08/02	AM	Conventional diesel	3
29	06/05/02	AM	Trap-outfitted	7
32	06/06/02	AM	Trap-outfitted	7

Inspection of the time series graphs for each run allowed identification of changes in the concentrations that could be correlated with the position of the windows and calculation of the 95% confidence intervals of the average concentrations demonstrated exposures inside the cabin depended significantly on the position of the windows. This finding was confirmed when we analyzed the cumulative frequency distribution of the data.

As mentioned in several sections of this report, the influence of the window position varied between pollutants. Directly emitted, diesel vehicle-related pollutants with low background concentrations, such as black carbon, PAH, and NO_2 exhibited a different behavior than those such as PC and $PM_{2.5}$ that had strong "background" contributions. Recent studies (Zhu et. al. 2002 a,b) suggest that $PM_{2.5}$ is a "background" pollutant substantially affected by regional sources, secondary formation and meteorological conditions and not a sensitive signature for direct emissions from vehicles. As mentioned before, our analyses of the particle count data refer only to the size range between 0.3 and 0.5 μm, near the peak of the accumulation mode for the particle size distributions by volume and mass (Hinds, 1999). Although number concentrations of ultra-fine particles are closely related to direct emissions from mobiles sources (Zhu et al., 2002 a,b), once these particles grow into the accumulation mode it is difficult to link them directly to emissions. In summary, the particle counts measured in the accumulation mode would tend to behave more like a "background" pollutant, and this is confirmed by the strong correlations we found between PC (0.3 to 0.5 μm) and $PM_{2.5.}$ For the following analysis, we selected one pollutant we will refer to as primarily directly emitted (e.g. black carbon or PAH), and one (e.g. $PM_{2.5}$ or fine particle counts) we will refer to as a "background" pollutant.

5.3.2.1 Black Carbon

The concentration time series for black carbon followed similar trends for all three window position tests. The concentrations inside the cabin tended to decrease when the windows were opened and tended to increase when the windows were closed. Figure 5.3.2.1 shows the ten-second data for black carbon concentrations during Run 32.

The results of the 95% confidence interval calculations were also consistent between the three runs. In all cases, the average concentrations were higher when the windows were closed (see Figure 5.3.2.2). However, the average concentrations during Run 9 (on a conventional diesel bus) were substantially higher than during Runs 29 and 32 (on the trap-outfitted diesel bus), due to the different bus types as discussed in Section 5.3.1.

Figure 5.3.2.3 shows the cumulative frequency distributions of the concentrations with the windows open and closed during Run 32, demonstrating the concentrations were higher when the windows were closed.

5.3.2.2 Particle Counts in the Accumulation Mode

For particle counts in the size range of 0.3 – 0.5 μm (PC), the concentration time series also followed similar trends for all three runs, but different to those observed for black carbon. The concentrations inside the cabin tended to increase when the windows were opened and tended to decrease when the windows were closed. Figure 5.3.2.1 shows the ten-second data for PC in a portion of the accumulation mode (0.3 to 0.5 um) during Run 32.

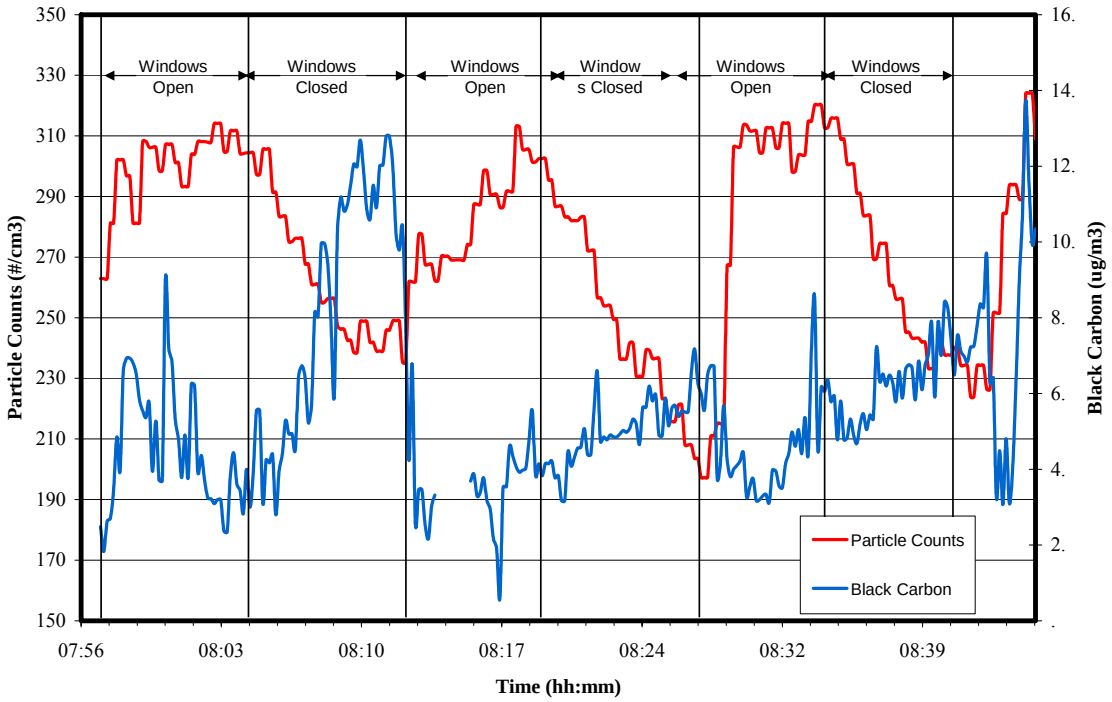


Figure 5.3.2.1 Black carbon and particle counts (0.3 to 0.5 µm) concentrations during Run 32 on the particle-trap outfitted diesel bus.

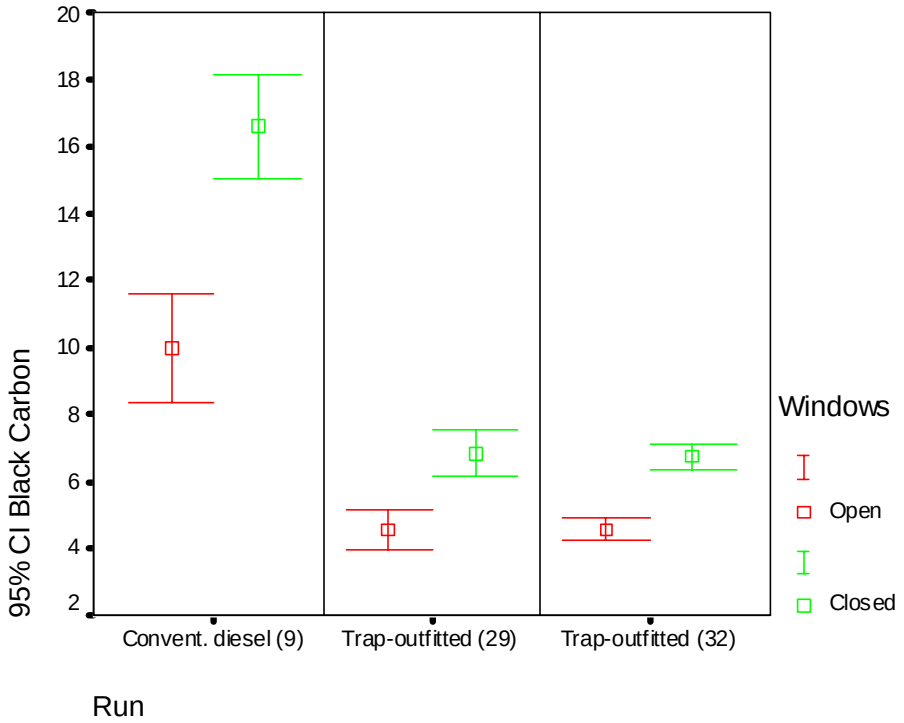


Figure 5.3.2.2 Mean concentrations and 95% confidence intervals of black carbon (µg/m^3).

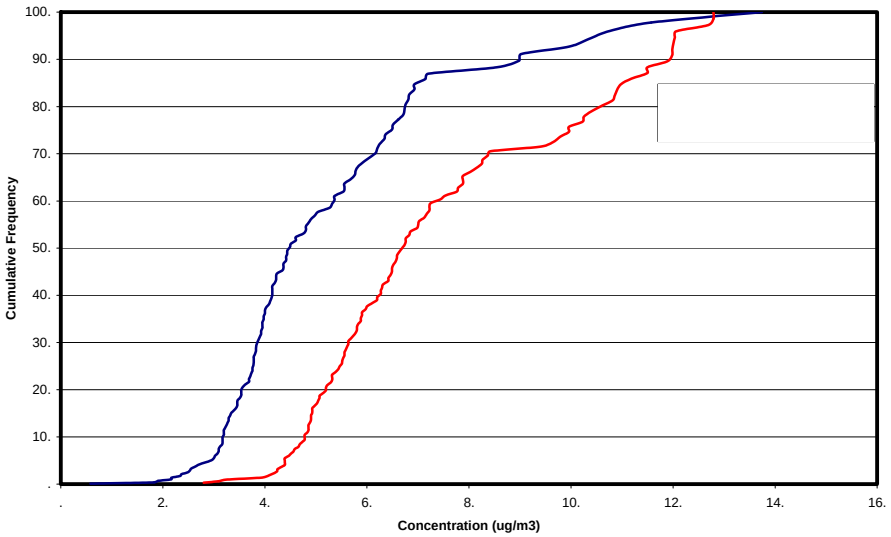


Figure 5.3.2.3 Cumulative frequency distributions for black carbon during Run 32.

The 95% confidence intervals for PC (0.3 to 0.5 um) were again consistent between the three runs but opposite of those for black carbon. In all cases, the average concentrations were higher when the windows were opened. Figure 5.3.2.4 depicts these results. The average concentrations during Run 9 were substantially lower than during Runs 29 and 32. These differences could be related to differences in background concentrations of particles in the accumulation mode.

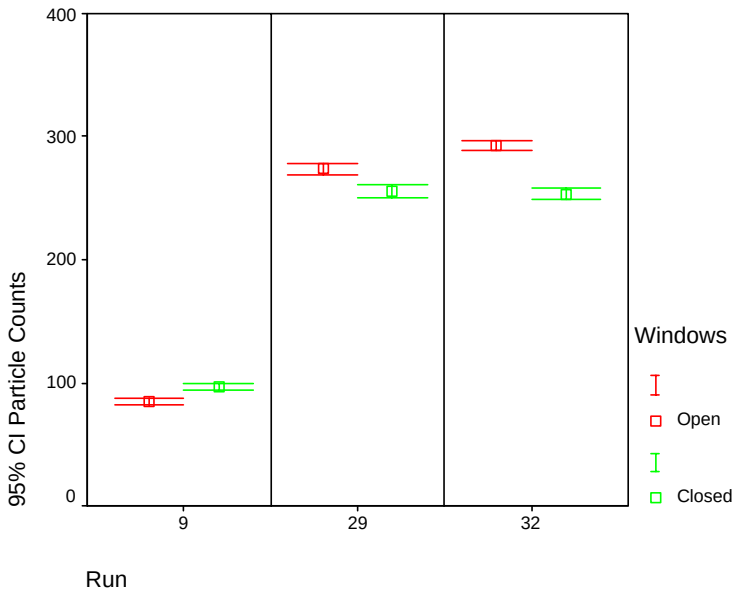


Figure 5.3.2.4 Mean particle counts and 95% confidence intervals in the accumulation mode from 0.3 to 0.5µm (#/cm^3).

Figure 5.3.2.5 shows the cumulative frequency distributions of the particle counts concentrations with the windows open and closed during Run 32, confirming the previous findings that the concentrations were higher when the windows were opened.

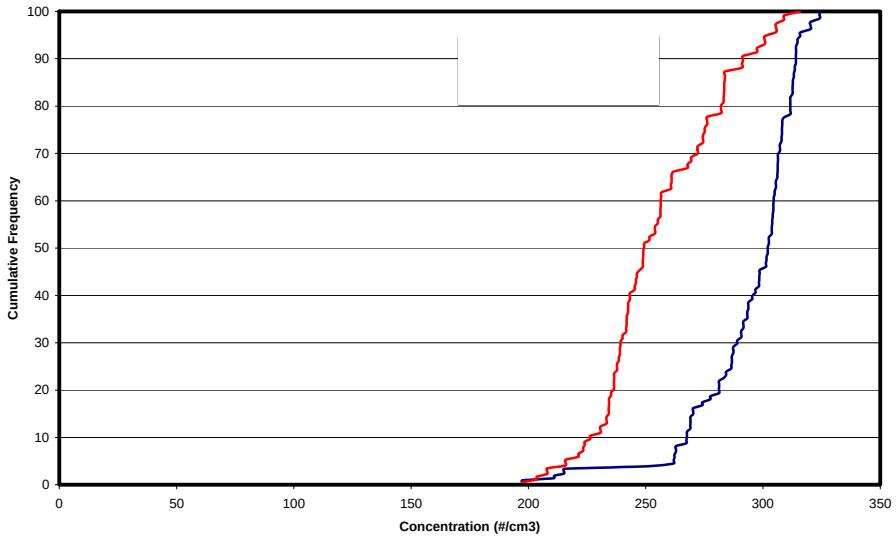


Figure 5.3.2.5 Cumulative frequency distributions for particle counts during Run 32.

5.3.2.3 Effect of Window Position on Directly Emitted Pollutants versus Background Pollutants

Figure 5.3.2.1 shows the inverse behavior between black carbon and particle counts. For directly-emitted, diesel-vehicle pollutants it appears in these runs the main source of pollution was the bus's own exhaust. Concentrations tended to increase when the buses were partially isolated from the outside, whereas increased ventilation rates caused dilution with the windows opened.

For "background" pollutants, concentrations were lower when windows were closed, slowing the entrance of pollutants into the cabin.

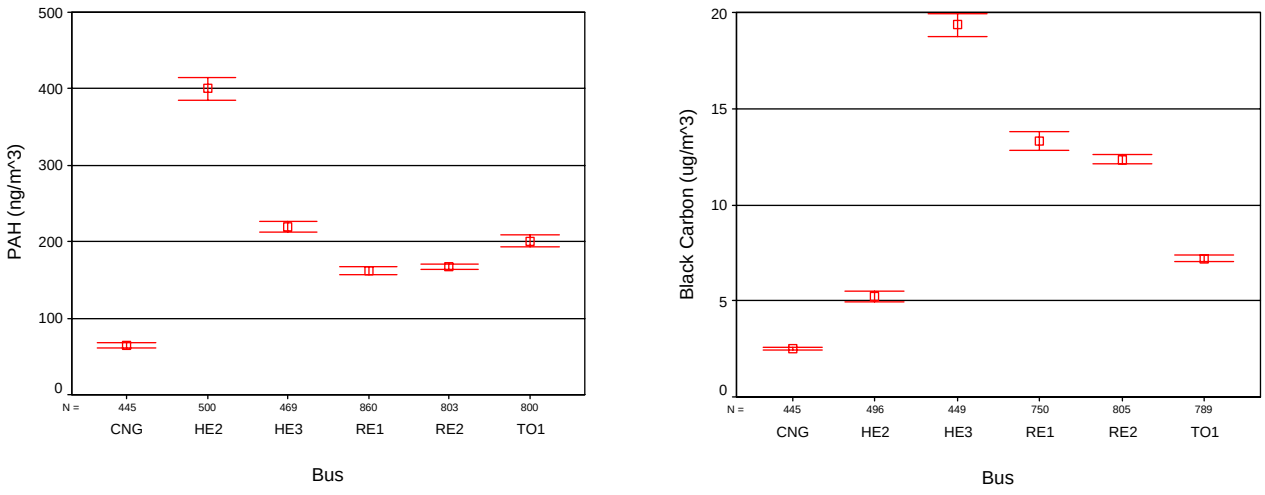
Both open and closed windows could lead to undesirable exposure scenarios, depending upon the pollutants involved and the degree of self pollution. For directly emitted pollutants, having the windows open could lead to short-term high exposures from other vehicles whereas closing the windows could lead to longer-term elevated exposures with higher average concentrations, particularly for dirty buses. On the other hand, for "background" pollutants closing the windows would yield lower average concentrations inside the cabin.

These findings emphasize the importance of transitioning as soon as possible to "cleaner" buses, such as the CNG bus we tested, to minimize pollution inside the cabin due to the bus's own exhaust, as well as to reduce impacts on and from other nearby vehicles, as discussed in Section 5.3.5.

5.3.3 Effect of Bus Type

In Section 5.2 we combined the results from two urban routes and the seven bus types to establish the overall average differences in exposure between the three microenvironments. However, the data also suggested that variables such as bus type and route type (see Section 5.3.4) played a significant role in determining the exposure inside the cabin. This section focuses on the effect of different bus types on concentrations inside the bus cabin when the windows of the bus were closed. However other variables, such as roadway type and the presence of other diesel vehicles, may have also influenced concentrations inside the cabin, and are discussed in other sections of this report (see Section 5.3.4 and 5.3.5).

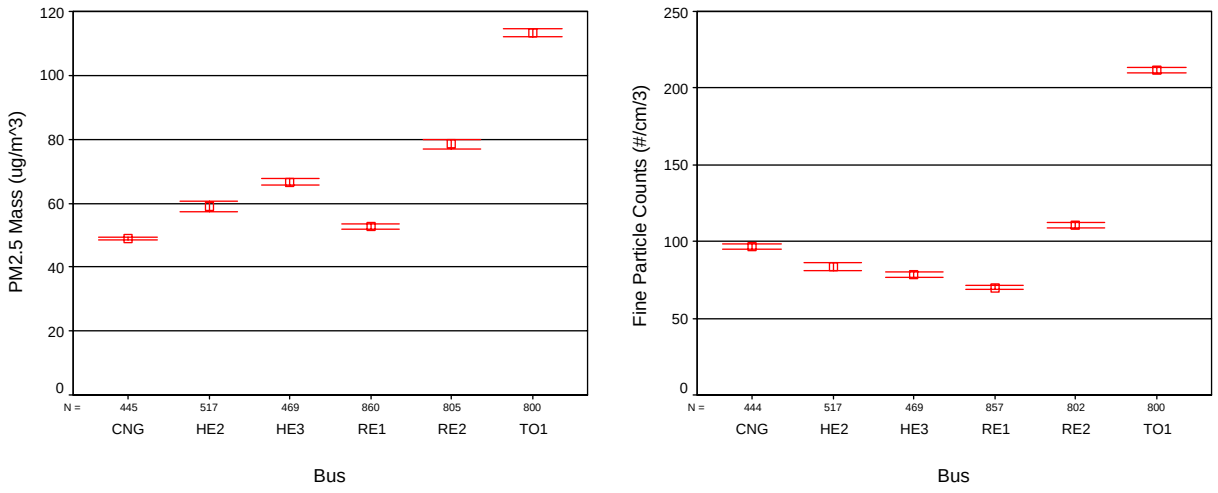
Different bus models and ages correspond to different construction and cabin designs and the results of our SF_6 analyses suggested these differences may result in a wide range of pollutant exposure across bus types. Figures 5.3.3.1 to 5.3.3.4 show the observed mean concentrations and 95% confidence intervals for the mean for various bus types investigated for several key pollutants. Confidence intervals were calculated using the standard error of the mean, and are not related to the precision estimates in Section 5.1.3. To minimize confounding factors, these plots were based only on the measurements in the rear/inside of the cabin for urban route one during commutes with windows on the bus closed. Again, as only one instrument, the rear instrument, was used for all analyses in the comparison between bus types, the results of the precision estimates in Section 5.1.3 do not apply here.



(a) PAH (b) Black carbon

Figure 5.3.3.1 Mean concentrations and 95% confidence intervals for PAH and black carbon across six bus types.

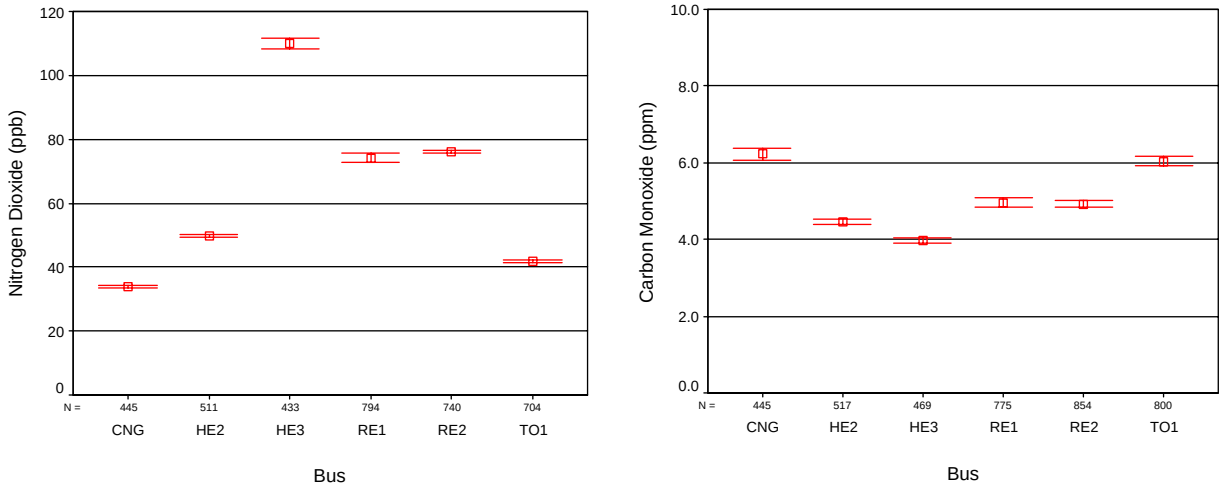
Figure 5.3.3.4 shows the results for the VOC data. Although the differences between buses appeared to be significant, the average concentrations were quite low relative to previously measured ambient total VOC concentrations (mostly below 200 ppb).



(a) $PM_{2.5}$
(Data collected with the DustTrak instrument)

(b) Fine Particle Counts (0.3 – 0.5 µm)

Figure 5.3.3.2 Mean concentrations and 95% confidence intervals for $PM_{2.5}$ and PC.



a) NO_2

(b) Carbon Monoxide

Figure 5.3.3.3 Mean concentrations and 95% confidence intervals for NO_2 and CO.

These figures illustrate the complexity of the data set, where differences across bus types are pollutant-dependent. To better understand the similarities and differences between pollutants, we summarized the findings presented in Figures 5.3.3.1 to 5.3.3.4 in Table 5.3.3.1 in terms of mean concentrations and 95% confidence intervals for the means, by bus type.

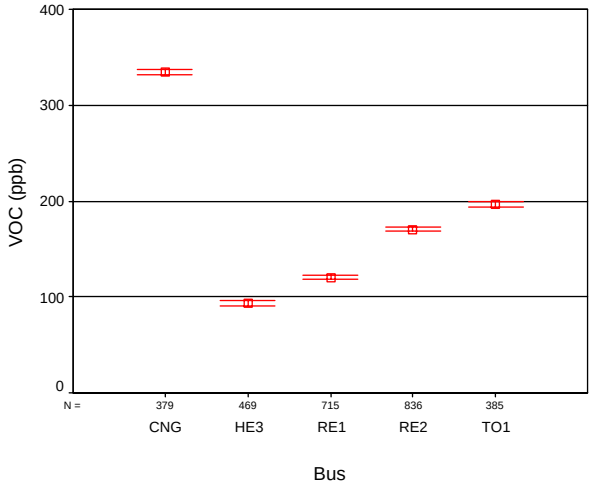


Figure 5.3.3.4 Mean concentrations and 95% confidence intervals for VOC.

Table 5.3.3.1 Summary of mean concentrations and 95% confidence intervals inside the cabin for commutes with windows closed (morning) by bus type[1,2].

	HE2	HE3	RE1	RE2	TO1	CNG
Black Carbon (ug/m^3)	5 ± 0.3	19 ± 0.6	13 ± 0.5	12 ± 0.2	7.2 ± 0.2	2.5 ± 0.1
PAH (ng/m^3)[3]	400 ± 15	220 ± 6.4	162 ± 5.8	167 ± 3.6	201 ± 7	64 ± 3.3
NO_2 (ppb)	50 ± 0.5	110 ± 1.6	74 ± 1.4	76 ± 0.3	42 ± 0.4	34 ± 0.4
CO (ppm)	4.5 ± 0.1	4 ± 0.1	5 ± 0.1	5 ± 0.1	6 ± 0.1	6 ± 0.2
PC (#/cm^3)[4]	83 ± 2.5	78 ± 1.7	70 ± 1.4	111 ± 2.1	211 ± 1.9	97 ± 1.4
VOC (ppb)	N/A	93 ± 2.8	121 ± 2.3	171 ± 1.7	197 ± 2.4	334 ± 2.5

[1]Conventional diesel buses = HE2, HE3, RE1 and RE2.
[2]This table groups two types of pollutants analyzed in this study: pollutants primarily or entirely emitted from combustion sources (BC, PAH and NO_2), and "background" pollutants considered to be dominated by atmospheric transformations or sources other than diesel (e.g. CO from light duty motor vehicles, "background" accumulation mode particles).
[3]Measurements of PAH inside conventional diesel buses may be biased low due to the instrument's maximum setting during these commutes. See Sections 4.1.2.3.5 and 5.3.3 for specific details.
[4]In size range from 0.3 – 0.5 µm.

According to the results found for emissions of PAH (with the exception of HE2), black carbon, and NO_2, in general, the buses that exhibited high opacity and a high percentage of air inside the cabin coming from the exhaust, also exhibited the highest average inside/rear concentrations. In other words, for BC, PAH, and NO_2, the average inside/rear exposure results could be explained by the snap-and-idle test and SF_6 data.

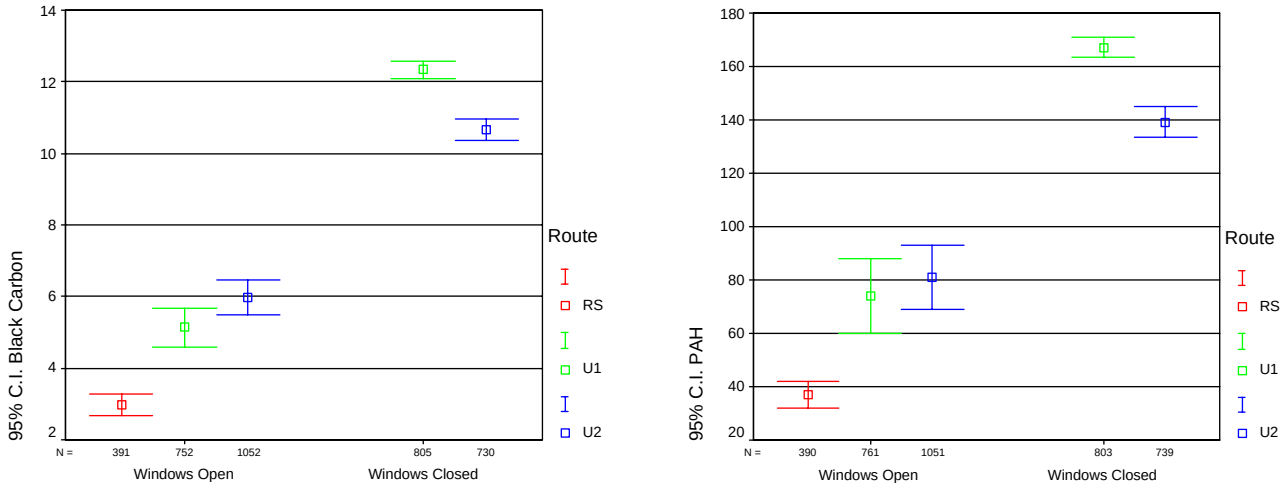
In contrast, the snap-and-idle test and the SF_6 data did not adequately explain the behavior of PC and $PM_{2.5}$ average concentrations inside the cabin. These results suggest that it is necessary to consider additional variables in order to explain the behavior of PC and $PM_{2.5,}$ such as the background contribution to concentrations of these pollutants inside the bus cabin.

In addition, as noted in Section 4.1.2.3.5, the PAH instrument's full scale was set to read a maximum of 500 ng/m^3 in the front, and 1000 ng/ m^3 in the rear during the first 17 runs of this study, all of which involved diesel buses. During Run 18 and all subsequent runs (including the TO1 and CNG buses), the scale was increased to read a maximum of 2000 ng/m^3 for both front and rear instruments. This created a potentially substantial bias in our comparison of PAH concentrations by bus type. PAH data for the diesel buses were biased low, because any concentrations higher than the maximum scale setting were recorded as the preset maximum of 500 or 1000 ng/m^3. In the later runs involving the trap-outfitted and CNG buses, the concentrations of PAH inside the bus cabin often exceeded 500 or 1000 ng/m^3. Thus, the differences we saw between the diesel buses versus the trap-outfitted or CNG buses likely understated the benefits of reduced PAH concentrations for the trap and CNG buses. We expect both higher peaks and higher average concentrations likely occurred inside the diesel buses during Runs 1 – 17 and therefore average PAH concentrations for diesel buses almost certainly would have been higher had the maximum scale been set to 2000 ng/m^3 as in the case of the trap-outfitted and CNG buses. Thus, if anything, the benefits of the trap and CNG technologies over the diesel with respect to PAH concentrations are under-estimated in our study.

Overall, the results presented in Table 5.3.3.1 demonstrate the importance of bus type for exposures inside the cabin, particularly for directly emitted pollutants such as black carbon and PAH. The ratios between the minimum and maximum average concentrations across bus type ranged from 1.6 to 6.3; hence, the differences in average exposure between buses were significant. For example, for black carbon, the average exposure associated with the HE3 bus was more than six times the exposure associated with the CNG bus.

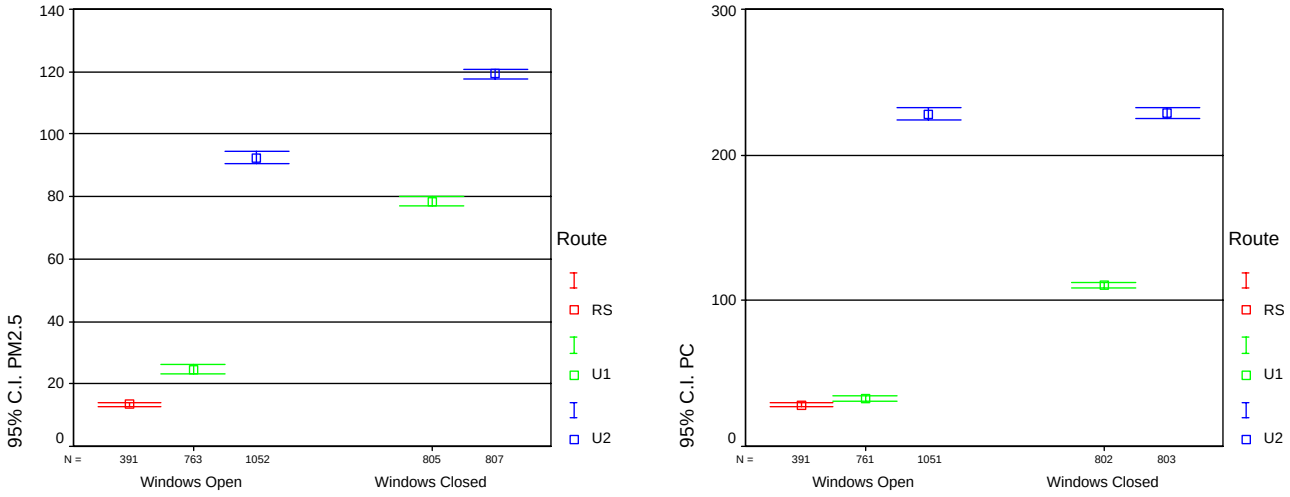
5.3.3 Effect of Route Type

Figures 5.3.4.1 and 5.3.4.2 show the observed mean pollutant concentrations and 95% confidence intervals for the three types of bus routes used during the main study (rural/suburban, urban route one, and urban route two). The confidence intervals were calculated based on the standard error of the mean. To avoid confounding factors only data from the RE2 bus were used since this was the only test bus used on all three routes, and only from measurements collected by sampling at the rear of the cabin.



(a) Black carbon (b) PAH

Figure 5.3.4.1 Mean concentrations and 95% confidence intervals for black carbon (μg/m^3) and PAH (ng/m^3).



(a) $PM_{2.5}$
(Data collected with the DustTrak instrument)

(b) PC

Figure 5.3.4.2 Mean concentrations and 95% confidence intervals for PC (#/cm^3) and $PM_{2.5}$ (μg/m^3) by bus route and time of day.

Figure 5.3.4.1 again shows similar behavior between black carbon and PAH. As expected, based in part on lower concentrations, the rural/suburban route (RS) exhibited significantly lower mean exposure than the urban routes. These differences could only be established for runs with windows open, since all rural/suburban runs were conducted in the afternoon with the windows open.

During the runs with windows open, the mean concentrations for black carbon and PAH were similar between the two urban routes (see Section 5.3.5.2.4). However, during runs with windows closed, U1 exhibited higher mean concentrations than U2. Differences in traffic conditions between the two urban routes could partially explain this result, where in general, the U2 route traveled trough less congested streets. As explained elsewhere this difference was only visible for morning runs where the windows were closed.

Figure 5.3.4.2 shows, when analyzing the type of route, PC and $PM_{2.5}$ also behaved similarly. The rural/suburban route and the U1 route exhibited similar average concentrations. The U2 route showed higher average concentrations than U1 during both morning and afternoon runs (in part due to higher speeds and greater ventilation on the freeway portion of U1, and greater density of diesel trucks on surface streets for U2).. These results demonstrate, again, the differences between directly emitted versus background pollutants.

5.3.5 Within Run Variables Investigated Using Real-Time Data and Videotape Analysis

The videotapes in conjunction with the activity logs recorded during each run were used to determine the influence of surrounding vehicles on black carbon and PAH concentrations inside the test buses during selected exposure runs. Black carbon and PAH were the focus of this analysis for three main reasons. The first was that these pollutants are both highly associated with diesel exhaust. Black carbon is typically used as a marker for diesel exhaust particulate (Rodes et. al., 1998; Fruin, 2003), and results from both the pilot and main studies found concentrations of black carbon and PAH were highly correlated (see Section 5.1.3). Second, we expected both pollutants resulted primarily from direct emissions, and had low background concentrations. This was important because high background concentrations could obscure our ability to identify the important variables contributing to exposure on the time scale of seconds and tens of seconds during the course of a commute. This also relates to the third major reason, which was the fast response of the instruments used to measure these pollutants allowed us to see changes in concentrations within seconds of an event. We used ten second data for all the videotape analyses.

All measurements of black carbon and PAH used in the following analyses were based on the measurements taken inside the cabin at the rear of the bus. For all runs in this study, the same instrument from each pair of instruments was placed at the rear of the bus (e.g. all runs had the same Aethalometer measuring black carbon at the rear of the bus). This is an important point because although in Section 5.1.3 we estimated the precision between the front and rear Aethalometers was low, the between-run comparisons in the videotape analyses which follow were based only on one instrument, the rear Aethalometer. Thus we were able to make relative comparisons between buses, without concerns of inadequate precision (for example between our front and rear Aethalometers), because these comparisons involved only one instrument for each pollutant (e.g. black carbon or PAH).

5.3.5.1 Selection of Runs for Videotape Analysis

The primary focus of this analysis was the effect of surrounding traffic on concentrations inside the bus, and less on the effect of the particular bus being tested (i.e. self pollution). In addition, because of the nature of the analysis, it was necessary to have highly variable concentrations inside the cabin over the course of a commute, in order to associate changes in

concentration with the events/characteristics that also changed during the commute. For these two reasons, we determined only afternoon runs, with windows partially open, were suitable for videotape analysis. Afternoon runs had generally low baseline pollutant concentrations compared with the morning runs (windows closed), but the greatest concentration variability, including high peaks. Figures 5.3.5.1 – 5.3.5.2 show examples of the types of time series obtained with morning and afternoon runs, respectively. The rapidly changing concentrations inside the bus from very low baseline concentrations to much higher peak concentrations found during afternoon runs provided the opportunity to investigate the factors influencing high concentrations inside the bus.

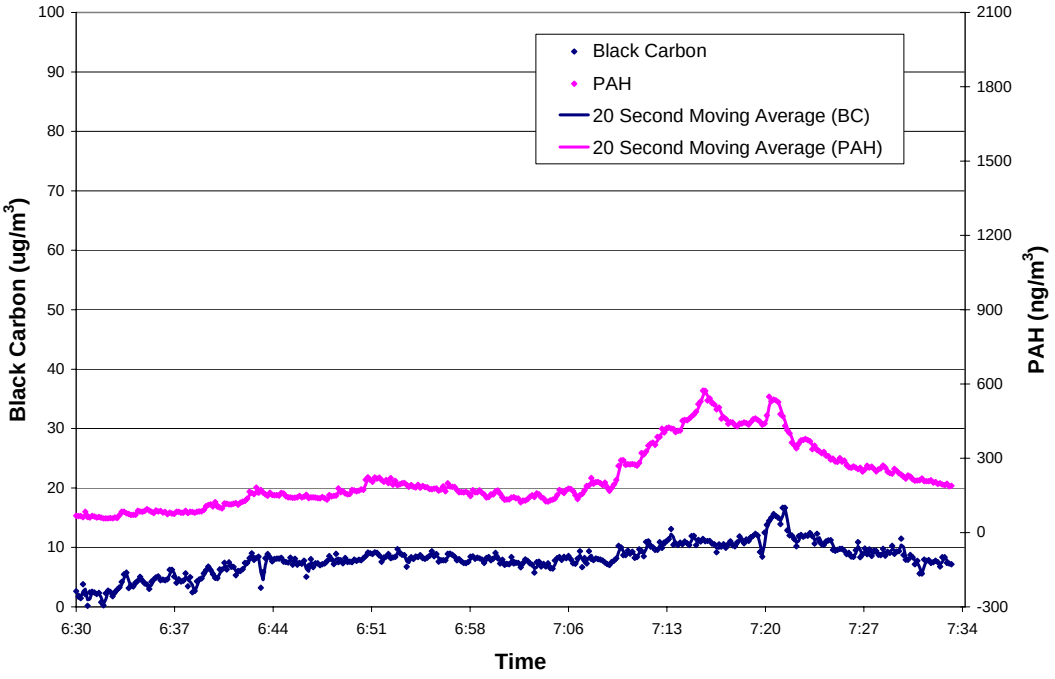


Figure 5.3.5.1 Run 31 (morning – windows closed) black carbon and PAH concentrations.

By contrast, during the morning runs with windows closed, pollutant concentrations inside the bus were less variable, and increased/decreased only gradually over much longer periods of time, making it more difficult to assess the factors causing those changes on a real-time basis.

In addition, focusing on afternoon runs for the videotape analysis increased our ability to assess the effects of surrounding traffic because these runs had higher ventilation rates and were more influenced by the outside air because the windows were open during all afternoon runs. As noted during the window position tests (see Section 5.3.2), when the windows were open, mean concentrations inside the bus tended to decrease, and were similar to concentrations outside the bus for directly emitted pollutants, such as black carbon and PAH. The high ventilation rate inside the bus cabin eliminated build up of concentrations inside the cabin over the course of the commute. High, but short-lived, peak concentrations inside the bus were observed only when the windows were open, indicating open windows allowed high concentrations from other

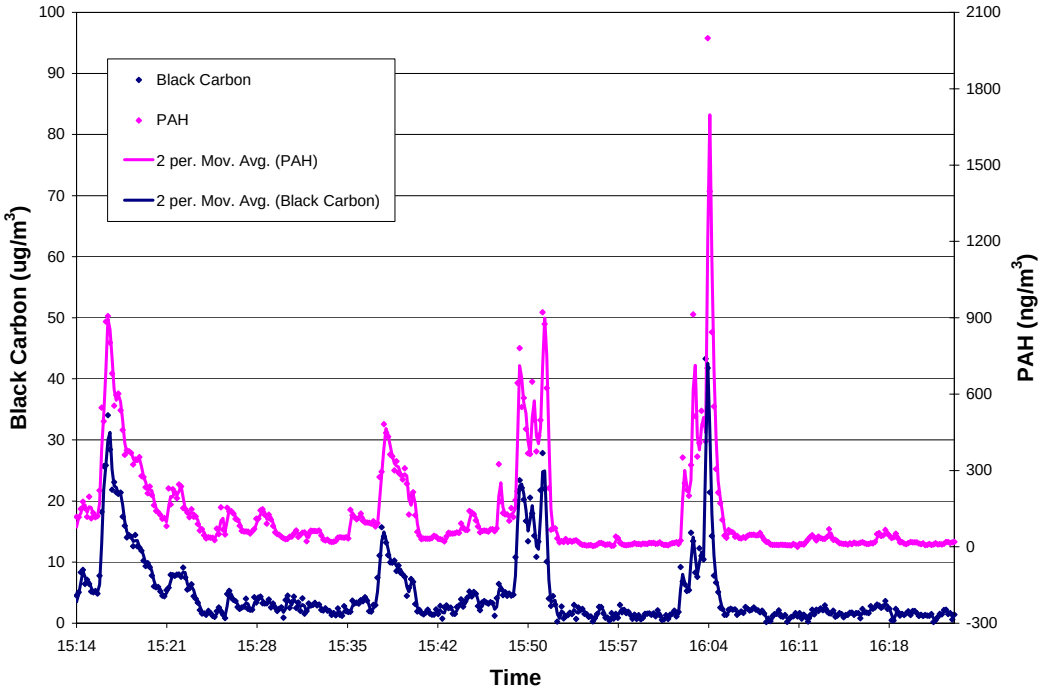


Figure 5.3.5.2 Run 33 (afternoon – windows open) black carbon and PAH concentrations.

sources to easily penetrate the bus cabin, although only for very short periods of time due to the high ventilation rates.

It is also important to note that meteorological conditions during the afternoon were consistent across all bus runs, with onshore flow conditions typical of the Los Angeles basin during May and June.

Out of a total of sixteen afternoon runs with windows open that were videotaped, seven were selected for complete videotape analysis, including a variety of bus types. Table 5.3.5.1 describes the runs analyzed using the videotapes. Four runs were on diesel buses, including Run 6 on a high emitter bus (HE2), Run 15 on a representative bus (RE1), Runs 20 and 26 on a second representative bus (RE2), Runs 30 and 33 on a particle-trap outfitted diesel bus (TO1), and Run 36 on a CNG bus (CNG). All of these runs were completed on the primary urban route, except one of the two runs on the representative diesel bus, RE2, which was done on the secondary urban route. Again, for all runs analyzed using the videotapes, the windows of the bus were open.

5.3.5.2 Results of the Videotape Analysis

Figures 5.3.5.3 – 5.3.5.4 show examples of time-series for black carbon and PAH, respectively, with events assigned to the concentration peaks based on the results of the videotape analysis. These figures are provided primarily to demonstrate the nature of the real-time data collected during this study, and the association between events and peak concentrations. However, all event assignments are not shown in these graphs. During the videotape analysis, every ten-second period within a run was given an event assignment according to the criteria described in Section 4.2.3.3. The event assignments shown in Figures 5.3.5.3–5.3.5.4 only represent the events occurring during major peak concentrations, and not all

events during a run, providing a qualitative demonstration of the value of the videotape analysis. However, all subsequent quantitative analyses include the event assignments for all ten-second periods during each run analyzed.

Table 5.3.5.1 Bus runs selected for videotape analysis.

Run Number	Bus ID	Bus Fuel Type	Bus Route	Time of Day	Average Bus Speed (km/hr)	Total Run Time (hr:min)	Average Wind Speed (km/hr)
6	HE2	Diesel	U1	PM	22.7	1:11	15
15	RE1	Diesel	U1	PM	23.8	0:57	9
20	RE2	Diesel	U1	PM	26.3	1:00	17
26	RE2	Diesel	U2	PM	21.4	1:21	9
30	TO1	Trap-Outfitted Diesel	U1	PM	24.8	0:52	14
33	TO1	Trap-Outfitted Diesel	U1	PM	23.1	1:10	10
36	CNG	CNG	U1	PM	19.8	1:24	15

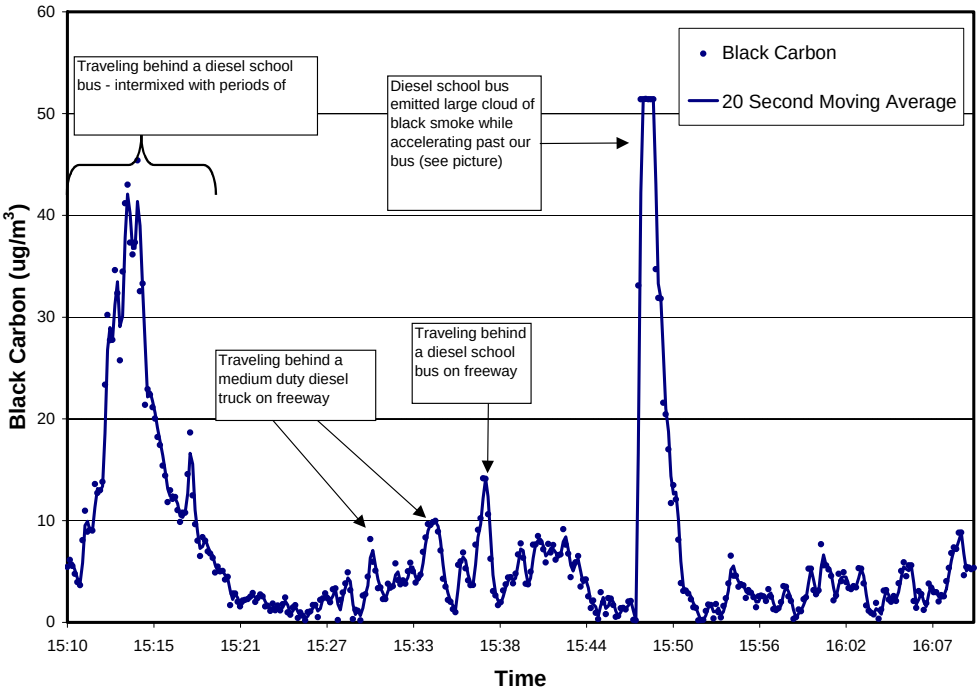


Figure 5.3.5.3 Run 20 black carbon concentrations with associated events.

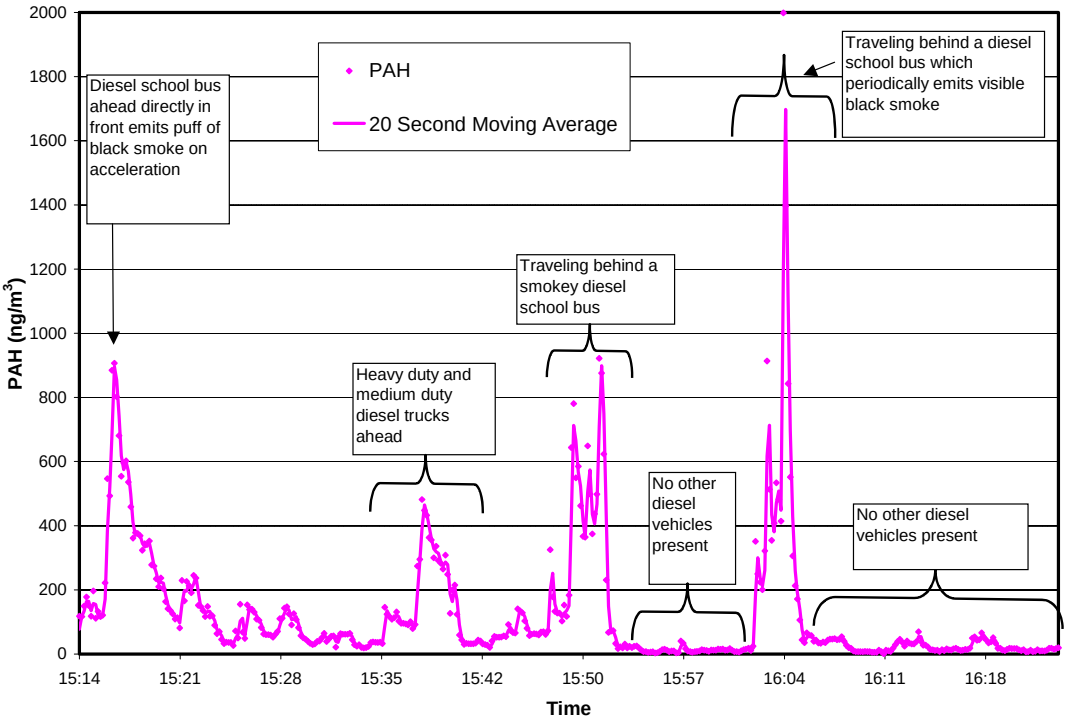


Figure 5.3.5.4 Run 33 PAH concentrations with associated events.

Small adjustments were made following the assignments of events based on further analysis of the data and inspection of the concentration time series. This was necessary because in a small number of instances, the criteria for assignment of an event described in Section 4.2.3.3 did not adequately characterize an association between pollutant concentrations inside the cabin and outside events. Inspection of the concentration time series revealed small portions of peak concentrations, typically the down-side of a peak, were not always identified with a given event, although they were likely associated with that event. Figure 5.3.5.5 provides an example, with an expanded time scale, during Run 36 where the original criteria did not result in the assignment of all elevated concentrations to the appropriate event. At the beginning of the period shown in Figure 5.3.5.5, a low baseline concentration of black carbon was observed, and no diesel vehicles were near the bus. At approximately 16:07, two events occurred nearly simultaneously. A passenger car with noticeable odors moved directly in front of the bus, and a diesel truck, which was ahead of the passenger car, came within about four to five car lengths of the bus. Although this distance was just outside our three car length limit (see Photograph 12), careful inspection of the black carbon time series during this period showed a dramatic increase in concentrations as soon as the truck and the passenger car both moved in front of the bus, with a further increase as the bus pulled closer to the diesel truck.

During the three minute period from about 16:07:30 to 16:10:00, we saw a series of peaks and valleys (although the valleys were still elevated well above the baseline) as the bus traveled behind the diesel truck. Although concentrations dipped slightly when the passenger car moved away at about 16:08:50, they increased again as the bus moved closer to the diesel truck. The truck moved in and out of a three car length distance ahead of the bus over the course of this time period, and the maximum concentration was reached when the diesel truck was within about one

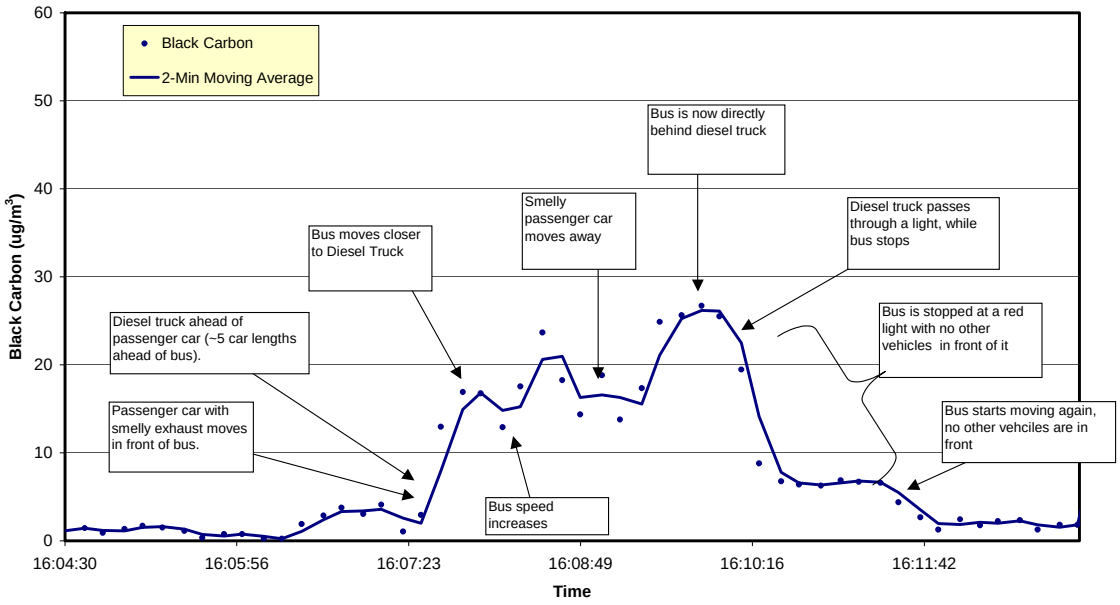


Figure 5.3.5.5 Events associated with increases/decreases in black carbon concentrations measured inside the CNG bus during Run 36.

Photograph 12. School bus ahead three-car lengths (maximum distance used for videotape analysis).

car length of the bus (at this point the passenger car was gone). Finally, the diesel truck passed through a traffic light, while the bus did not. At this point, according to our original criteria, the diesel truck was no longer associated with any subsequent time intervals. However, we saw from the black carbon time series that although the concentrations immediately started to decrease after the diesel truck was gone, it took approximately thirty seconds before the

concentrations were again close to the concentrations observed inside the bus prior to the encounter with the diesel truck. In this case, the highest concentrations were identified by our original criteria, but the elevated concentrations in the final tail of the peak, before background concentrations were again reached, were not. The event assignments in the few instances where this occurred were adjusted, and these periods of time were also assigned to the original event associated with the major peak. However, these adjustments accounted for only 3% of all vehicle assignments made during the videotape analysis for all seven runs. The great majority of the events during each run were identified solely on the basis of the original criteria described in Section 4.2.3.3.

5.3.5.2.1 The Effects of Following Diesel Vehicles With and Without Visible Exhaust

In order to assess the impact of diesel vehicles emitting visible exhaust, the time interval during which visible exhaust or odor from a diesel vehicle in front of, or adjacent to, the test bus was first noted, and the following one minute period after the exhaust was noted were assigned to the vehicle that emitted the exhaust. In order to determine the effect such an event had on concentrations inside the bus cabin, a comparison was done of all concentrations measured when no visible exhaust from surrounding vehicles was present, with the first ten second period during which exhaust was seen, and the one minute period following the emission of visible exhaust (see Photograph 13). The results of this comparison, stratified by the test bus type (diesel, trap-outfitted diesel and CNG) and including all seven runs analyzed with the videotapes, are shown in Table 5.3.5.2.

(a) (b)

Photograph 13. a) Smoky diesel school bus ahead on an arterial street.
b) Driving behind a heavy-duty diesel with high exhaust emitting visible smoke.

Table 5.3.5.2 The effect of visible exhaust from a diesel vehicle in front of or adjacent to the test bus on concentrations of black carbon and PAH inside the bus cabin.

	Mean Black Carbon Concentrations Inside the Test Buses (ug/m^3)			Mean PAH Concentrations Inside the Test Bus (ng/m^3)		
Presence of Visible Exhaust	Diesel Buses	Trap Outfitted Diesel Bus	CNG Bus	Diesel Buses	Trap Outfitted Diesel Bus	CNG Bus
No Visible Exhaust Present	5	3	2	76	78	70
Visible Exhaust First Emitted from a Diesel Vehicle in Front of the Test Bus	24	9	13	464	370	473
One Minute Period Following the Emission of Visible Exhaust from a Diesel Vehicle in Front of the Test Bus	28	11	14	586	362	517

Figures 5.3.5.6 and 5.3.5.7 show the boxplots of the same comparison for each of the bus types tested for black carbon and PAH, respectively. The boxplots show the median, quartiles, and extreme values. The box represents the interquartile range which contains 50% of the values. The whiskers are lines that extend from the box to the highest and lowest values. The

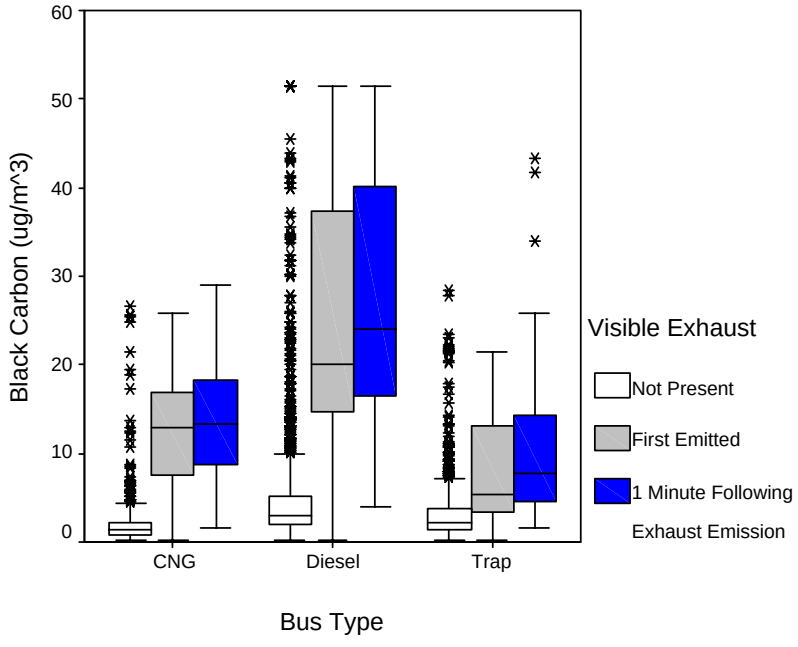


Figure 5.3.5.6 The effect of visible exhaust from a diesel vehicle in front of or adjacent to the test bus on concentrations of black carbon inside the bus cabin.

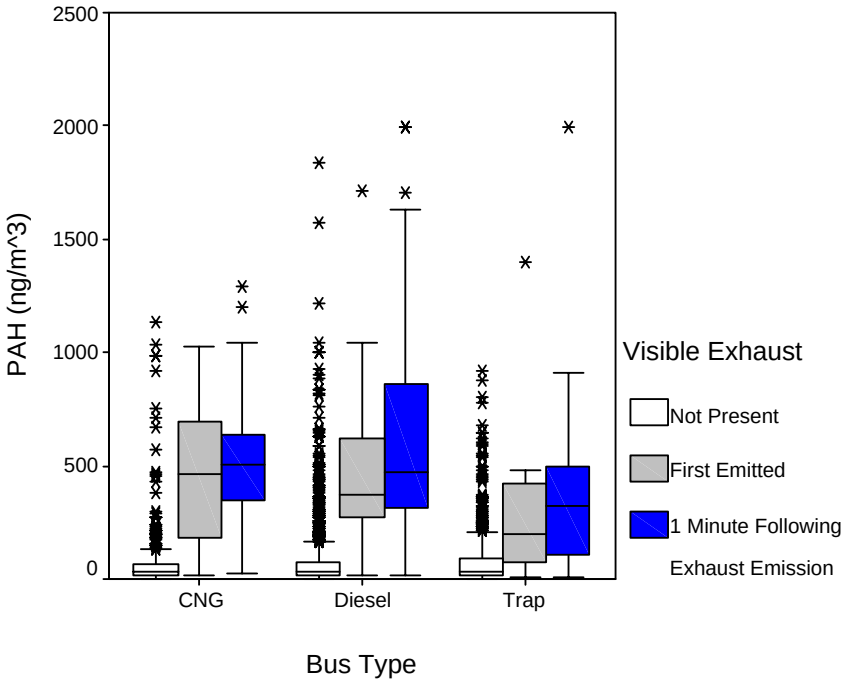


Figure 5.3.5.7 The effect of visible exhaust from a diesel vehicle (in front of, or adjacent to, the test bus) on concentrations of PAH inside the bus cabin.

line across the box indicates the median, and outliers or extreme values are included as individual points. Mean black carbon and PAH concentrations inside the test bus increased four to eight times when a diesel vehicle in front of the test bus emitted visible black smoke. These results indicated the presence of visible exhaust from a diesel vehicle near the test bus was a major predictor of high concentrations of black carbon and PAH inside the cabin.

In an effort to further characterize the effect of the presence of other diesel vehicles in front of, or adjacent to, the test bus (see Photograph 14) during the course of a bus commute, a comparison was done of the observed concentrations inside the test bus while following different vehicle types, with or without visible exhaust. All seven runs (including the three different bus types) were included in this analysis. Five categories of vehicles being followed were used: diesel school buses with visible exhaust; other diesel vehicles with visible exhaust; diesel school buses without visible exhaust; other diesel vehicles without visible exhaust; and gasoline vehicles or no target. The category of other diesel vehicles included both medium and heavy-duty vehicles (2,3 or 5 axles). Although during the video analysis separate categories for medium and heavy duty diesel vehicles were used, for this analysis these two categories were collapsed into one. The reasons for doing this were first, to increase the sample size, because the number of times these vehicles were encountered was low, and second, because the distinction between medium and heavy duty diesel vehicles was not always apparent from the videotapes. The assignment to one category or the other was often difficult to make and collapsing the categories eliminated possible misclassification between the two categories. Diesel transit buses were not included because the number of times they were encountered during these bus commutes was low (N<20 for all runs combined). Most of the transit buses encountered were fueled by CNG.

Photograph 14. Medium duty diesel vehicle ahead on the freeway.

Table 5.3.5.3 shows the results of this comparison. Mean black carbon and PAH concentrations inside the test buses were highest when following a diesel school bus which emitted visible exhaust, and lowest when following a gasoline vehicle or no target. When following a smoky diesel school bus, the concentrations measured inside the bus cabin were

Table 5.3.5.3 Concentrations measured inside the bus cabin while traveling behind different types of vehicles.

Type of Vehicle Being Followed:	Mean Concentration Inside the Test Buses[1]	
	Black Carbon (ug/m3)	PAH (ng/m^3)
Diesel School Bus Emitting Visible Exhaust	23	552
Other Diesel Vehicle Emitting Visible Exhaust[2]	13[3]	429[c]
Diesel School Bus	12	271
Other Diesel Vehicle[2,4]	4	104
Gasoline Vehicle or No Target	3	48

Notes:

[1]Selected exposure runs included 4 runs on 3 diesel buses, 2 runs on a trap-outfitted diesel bus, and 1 run on a CNG bus. Only afternoon runs with windows open were included in this analysis.

[2]Diesel vehicles with 2, 3 or 5 axels were combined into one category to increase sample size.

[3]Small sample size (N<50).

[4]Transit buses were not assessed due to small sample size (N<10).

eight and twelve times greater for black carbon and PAH, respectively, compared with following a gasoline vehicle or no target. When following a diesel school bus that was not emitting visible exhaust, black carbon and PAH concentrations inside the test buses increased four and six times, respectively, compared with following a gasoline vehicle or no target.

Figures 5.3.5.8 and 5.3.5.9 show the boxplots of the black carbon and PAH concentrations, respectively, measured inside the test buses while following different types of vehicles. The type of diesel vehicle encountered most frequently during all the bus runs was a diesel school bus. On average, during more than one-quarter of a bus commute, another diesel vehicle was within three car lengths in front of or adjacent to, our bus, with diesel school buses responsible for over sixty percent of that time. The highest concentrations inside all the test buses were observed when following, or adjacent to, a diesel school bus. This demonstrated that the type of school bus a child rides on is not the only determinant of exposure. Another major factor is the type of vehicles encountered during the commute. Under conditions similar to our study conditions (e.g. high density urban setting, with many children commuting to school by bus), the impact of other diesel school buses in the surrounding area on the exposure of a commuting child would be significant, regardless of the type of bus that child was riding on. This was a particularly important finding as we observed during the course of the study that school buses tended to caravan with each other, especially when leaving the school at the same time in the afternoon.

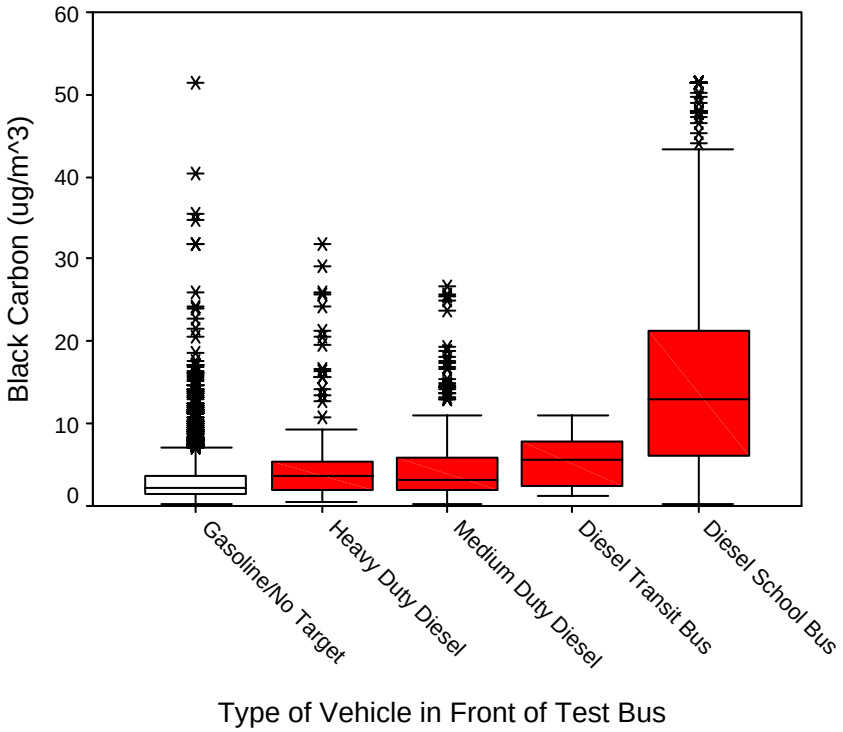


Figure 5.3.5.8 Concentrations of black carbon inside the test buses while following or adjacent to different types of vehicles.

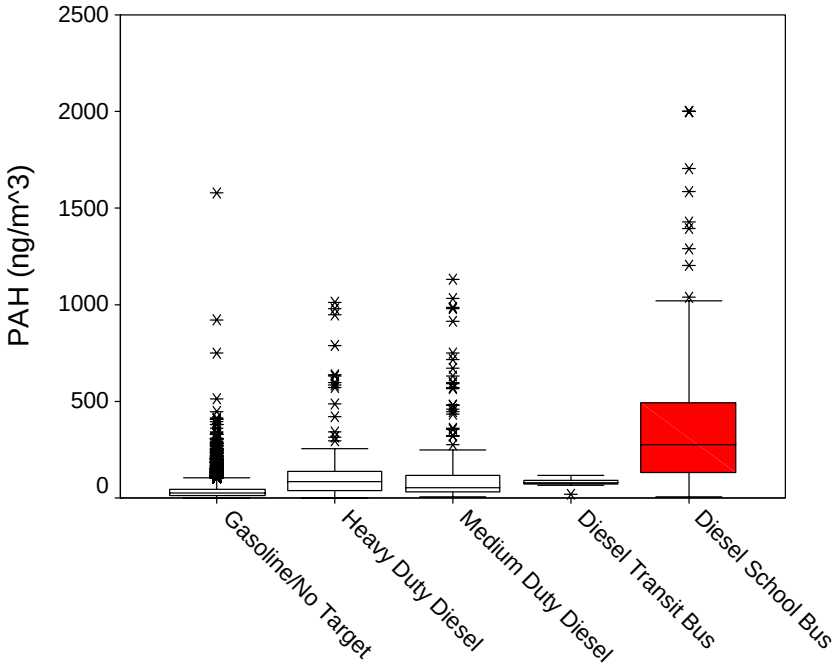


Figure 5.3.5.9 Concentrations of PAH inside the test buses while following or adjacent to different types of vehicles.

5.3.5.2.2 The Effect of Bus Fuel Type or After-Treatment Technology

Although the primary purpose of the videotape analysis was to investigate the effects of surrounding traffic and outside events on concentrations inside the bus, we were also able to investigate differences between bus fuel types and after-treatment technology. As stated previously, we tested three different bus types: conventional diesel, trap-outfitted diesel, and CNG. For these analyses, we used only the six runs on the primary urban route. Despite the effect of open windows and consequently high ventilation rates in the bus during these afternoon runs, which reduced build up of any of the bus's own emissions inside the bus, we were still able to see differences in concentrations during the commutes based solely on the type of test bus, irrespective of surrounding events and traffic conditions. In order to show this, we first investigated differences in traffic congestion levels, frequency of encounters with other diesel vehicles, and average wind speeds, to verify these conditions were similar between runs and thus not responsible for any differences observed between bus types.

One major indicator of traffic congestion levels (Photograph 15) on these commutes was the average speed and the total time to complete the run. The run on the second urban route was not included in the comparisons between test bus types because any differences due to the route may have confounded the analysis. Consequently, all runs considered in this section followed the same route, our primary urban route, and were performed at approximately the same time of day, on either Wednesday (four runs) or Thursday (two runs), over the course of seven consecutive weeks. We would expect similar average speeds and total time to complete the route

between runs if traffic congestion levels were similar. Table 5.3.5.1 shows the average speeds and total time to complete the route for each of the six runs on the primary urban route. Table 5.3.5.4 shows the mean black carbon and NO_2 concentrations, categorized into periods of idling when no other diesel vehicles were present for these six runs. Run 36, the CNG bus, had the slowest average speed, and the longest total commute time, indicating traffic congestion levels were heavier on this run than on all other runs. Yet, the black carbon concentrations over an entire commute were almost twice as high on the diesel bus with the highest average speed (Run 20).

Photograph 15. Typical congestion on the freeway during an afternoon run.

When the black carbon concentrations were broken down into periods of idling with no other diesel vehicles present, we also saw the CNG bus again had the lowest mean concentration of black carbon, while the conventional diesel buses had concentrations that were two to four times higher than either the CNG or trap-outfitted diesel buses. Both the trap-outfitted diesel and conventional diesel buses had concentrations of NO_2 that were up to two to three times higher, respectively, than the CNG bus.

Figures 5.3.5.10 – 5.3.5.12 and 5.3.5.13 – 5.3.5.15 show the boxplots of the breakdown of black carbon and PAH concentrations, respectively, by idling with or without other diesel vehicles present, for each of the three bus types. These graphs demonstrate the importance of the presence of a diesel vehicle in front of the test bus in determining the highest concentrations inside the bus cabin. However, we also saw a difference between test bus types as well. For the conventional diesel buses, the highest peak concentrations were obtained when the test bus was idling behind another diesel vehicle. But high peak concentrations were also obtained during periods of idling, without other diesel vehicles present, and even during periods when the bus was not idling and no other diesel vehicles were present.

Table 5.3.5.4 Mean black carbon and NO_2 concentrations inside the test buses under selected conditions.

		Black Carbon (ug/m^3)			NO_2 (ppb)		
Run Number	**Bus Fuel Type**	**Entire Run**	**Not Idling NO Vehicle in Front**	**Idling with NO Vehicle in Front**	**Entire Run**	**Not Idling NO Vehicle in Front**	**Idling with NO Vehicle in Front**
6	Conventional Diesel	5.1	2.5	7	54	50	50
15	Conventional Diesel	11.5	3	3.4	122	106	102
20	Conventional Diesel	7.2	4.1	6.2	88	83	79
26	Conventional Diesel	6.6	3.9	4.7	50	44	51
30	Trap-Outfitted Diesel	2.7	2.2	1.9	82	81	82
33	Trap-Outfitted Diesel	4.5	2.6	1.7	90	85	80
36	CNG	3.9	1.5	1.7	40	38	33

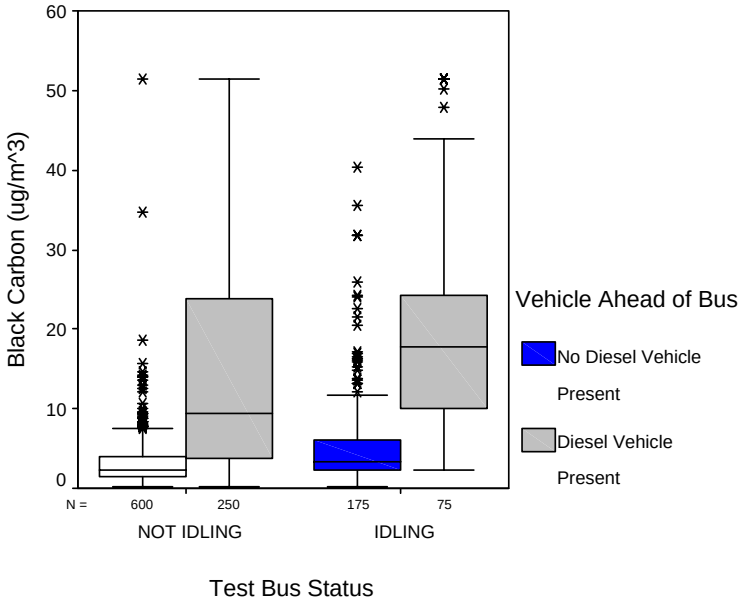


Figure 5.3.5.10 Concentrations of black carbon inside the conventional diesel school buses while idling and/or following another diesel vehicle.

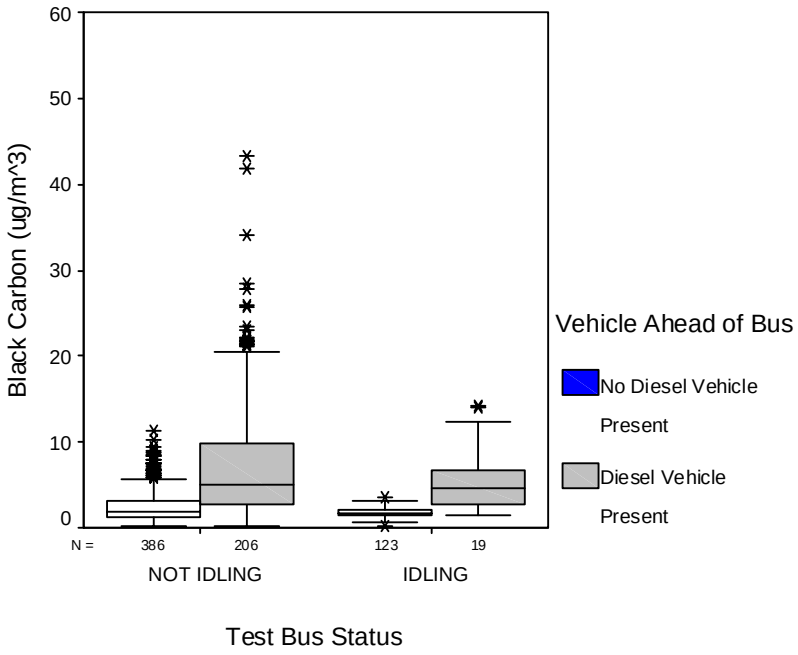


Figure 5.3.5.11 Concentrations of black carbon inside the trap-outfitted diesel school bus while idling and/or following another diesel vehicle.

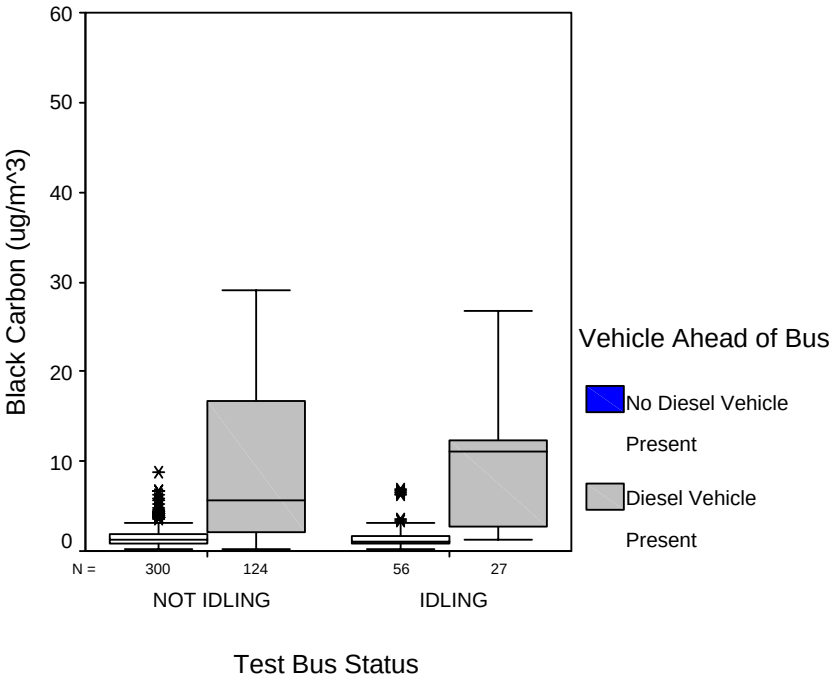


Figure 5.3.5.12 Concentrations of black carbon inside the CNG school bus while idling and/or following another diesel vehicle.

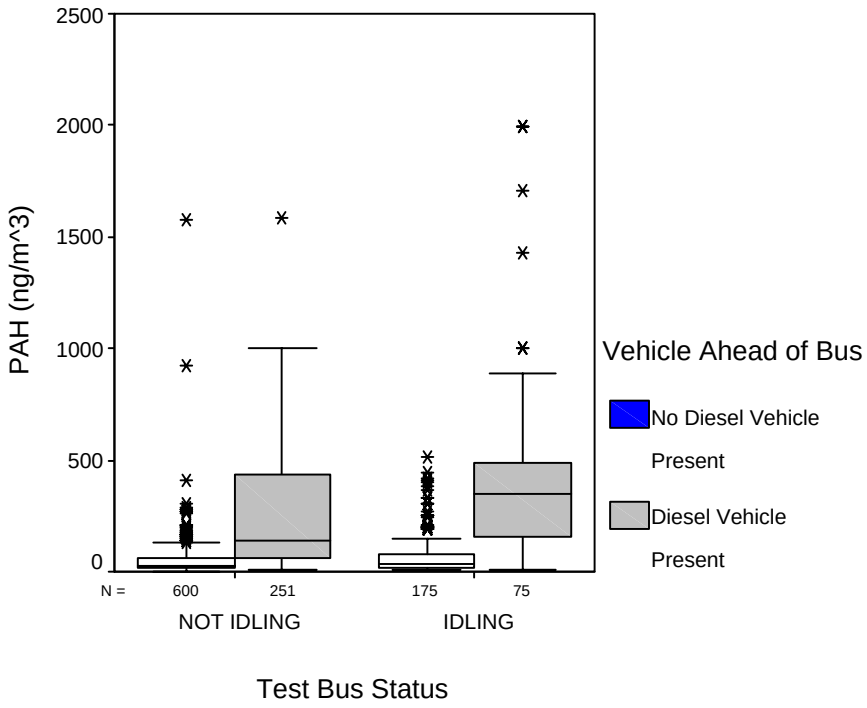


Figure 5.3.5.13 Concentrations of PAH inside the conventional diesel school buses while idling and/or following another diesel vehicle.

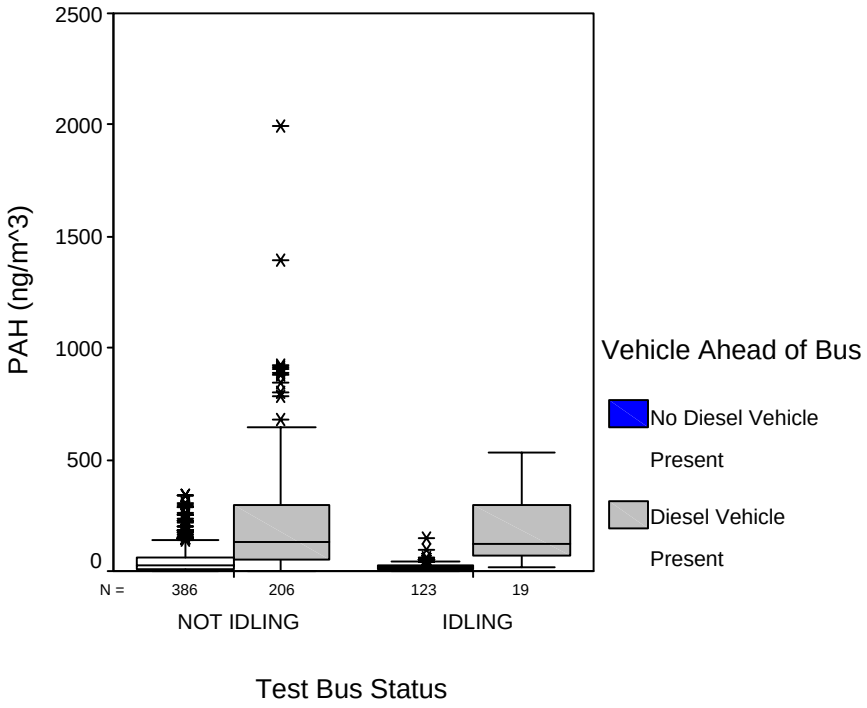


Figure 5.3.5.14 Concentrations of PAH inside the trap-outfitted diesel school bus while idling and/or following another diesel vehicle

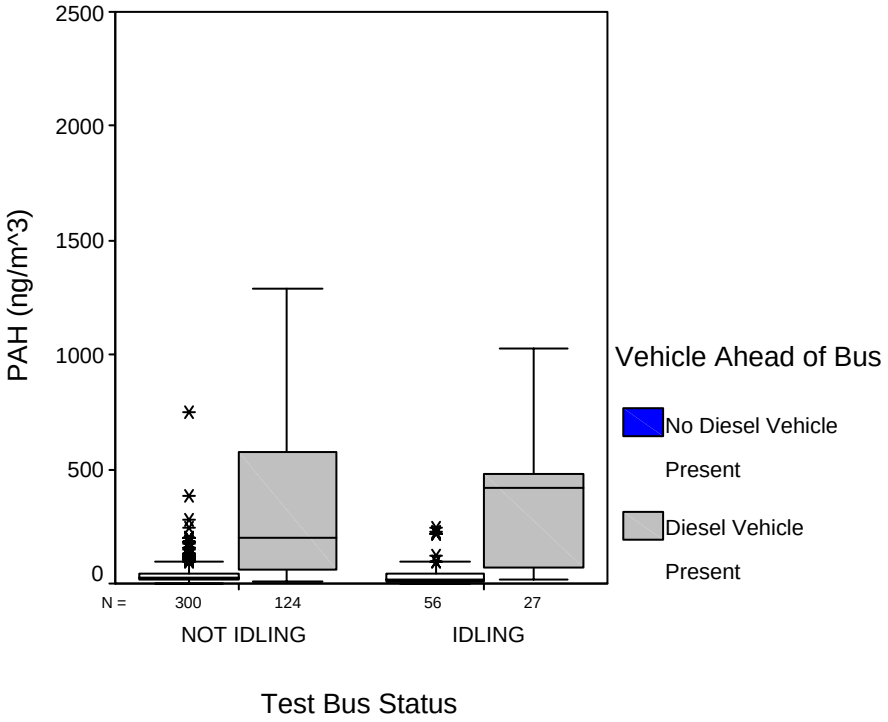


Figure 5.3.5.15 Concentrations of PAH inside the CNG school bus while idling and/or following another diesel vehicle.

By comparison, the trap-outfitted diesel bus and CNG bus had high peak concentrations only while traveling behind a diesel vehicle. In addition, the highest peaks obtained on the CNG bus (which occurred while traveling behind a diesel vehicle) were less than 30 $\mu g/m^3$, while on the conventional diesel buses, the highest peaks reached were over 50 $\mu g/m^3$, almost twice as high. Concentrations over 50 $\mu g/m^3$were observed inside the conventional diesel test buses not only when following another diesel vehicle, but also during periods when no other diesel vehicles were present. This indicated even with the high ventilation rates and low background concentrations observed during the afternoon runs with the windows open, the type of bus had a significant impact on concentrations inside the bus cabin, in particular, on peak concentrations.

Moreover, the differences found between bus types could not be attributed to differences in average wind speed between runs. Table 5.3.5.1 shows the average wind speeds, measured at the nearest AQMD stations (see Section 4.1.2.3.13), on all runs used in this analysis. The averages by bus type ranged from 12 km/hr (trap-outfitted diesel bus) to 15 km/hr (CNG bus), with the average speed for the diesel buses at 14 km/hr.

Finally, we also calculated the amount of time spent behind or adjacent to other diesel vehicles, to determine if there were differences between bus types based on this variable. We found that for each of the bus types tested, the amount of time spent behind or adjacent to other diesel vehicles was 25%, 30%, and 31% for the conventional diesel, CNG, and trap-outfitted diesel buses, respectively. Thus, we concluded differences found between bus types were not due to differences in the number of encounters with other diesel vehicles during the commutes.

In fact, the runs with the least number of encounters with other diesel vehicles were on the diesel test buses, the buses with the highest concentrations of black carbon and PAH concentrations inside the cabin.

5.3.5.2.2.1 Concentrations Inside Buses While Idling at Bus Stops

Investigation of concentrations inside the buses while idling at bus stops provided another opportunity to evaluate differences between bus types. Because the bus stops along the primary urban route were all located on small residential streets, usually in front of elementary schools, there were few other vehicles in the surrounding area during the afternoon runs (see Photograph 16). Thus, we expected concentrations inside the bus cabin during bus stops were dominated by the bus's own exhaust, with a minimal contribution from other vehicles.

Photograph 16. View out the front window of the test bus while idling at a bus stop (at Vermont Avenue School) during an afternoon run.

Table 5.3.5.5 shows the mean black carbon, and PAH, and NO_2 concentrations measured inside the test buses while idling at bus stops, categorized by bus type. On average, black carbon

Table 5.3.5.5. Mean black carbon, PAH and NO_2 concentrations measured inside the test buses while idling at bus stops in residential neighborhoods (urban route one).

Test Bus Type	Black Carbon (ug/m3)	PAH (ng/m3)	NO_2 (ppb)
Conventional Diesel	8.6 +/- 1.9	106 +/- 26	64 +/- 6
Trap-Outfitted Diesel[1]	1.9 +/- 0.2	20 +/- 4	78 +/- 6
CNG	1.2 +/- 0.2	27 +/- 16	37 +/- 3

[1] Trap outfitted diesel bus was the only bus to encounter another diesel vehicle at a bus stop for runs included here.

concentrations were four and six times higher inside the diesel buses at bus stops compared with the trap outfitted diesel bus and the CNG bus, respectively, while PAH concentrations were four to five times higher inside the conventional diesel buses. Concentrations of NO_2 were twice as high for both the conventional diesel and trap-outfitted buses than for the CNG bus.

Inspection of the boxplots in Figures 5.3.5.16 and 5.3.5.17 for black carbon and PAH concentrations, respectively, during bus stops revealed a much wider range of differences for the diesel buses tested. These figures showed both the CNG and the trap-outfitted diesel buses had relatively low concentrations and no high peaks while idling at bus stops. For these two bus types, the concentrations measured at the bus stops were lower than the average concentrations measured on these buses in congested traffic conditions when other diesel vehicles were not present. This finding indicated the contribution to black carbon and PAH from the CNG and diesel trap bus's own exhaust was minimal.

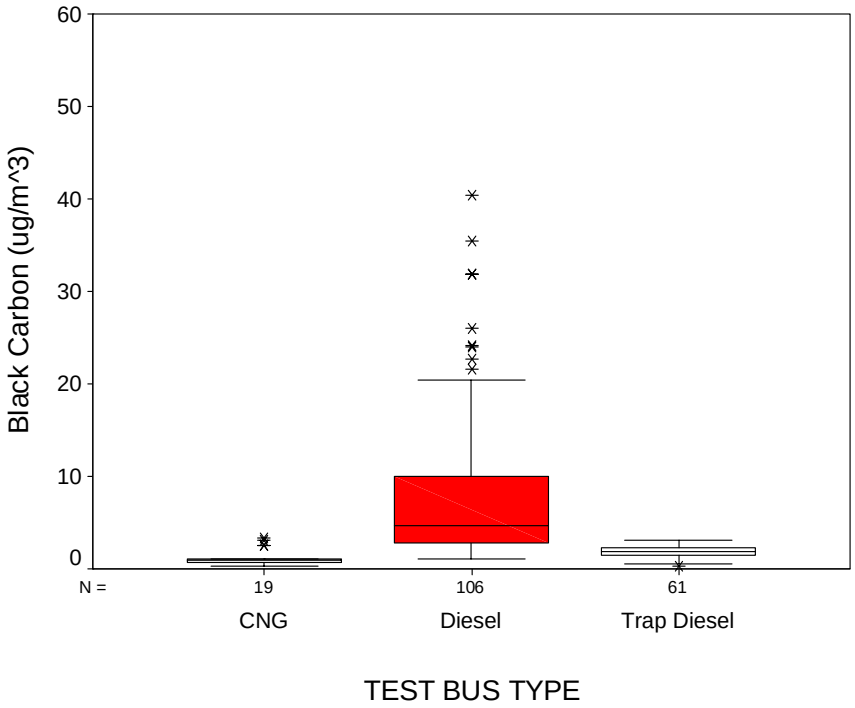


Figure 5.3.5.16 Concentrations of black carbon inside the cabin for conventional diesel, trap-outfitted diesel, and CNG school buses while idling at bus stops.

However, for the conventional diesel buses, we found a different result. First, we found a wider range of concentrations measured inside the bus cabin at bus stops. For black carbon, the range of concentrations was similar to what was observed during other portions of the bus commute. This was an interesting finding since the period of time spent idling at the bus stops had the least impact from traffic congestion and the presence of other diesel vehicles. In fact, only the trap-outfitted diesel bus had any other diesel vehicles present at any of the bus stops, and that was only for a short thirty-second period. Thus, the concentrations measured at the bus stops were not due to other vehicles or general traffic congestion. The differences seen between

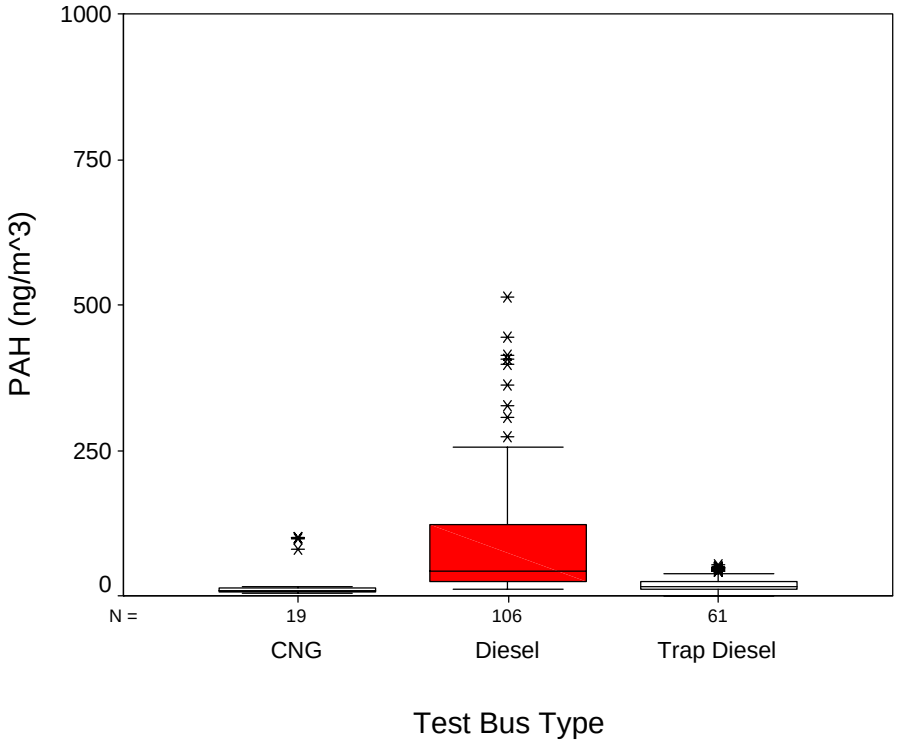


Figure 5.3.5.17 Concentrations of PAH inside the cabin for conventional diesel, trap-outfitted diesel, and CNG school buses while idling at bus stops.

bus types clearly indicated the conventional diesel buses contributed higher concentrations of black carbon and PAH to concentrations inside the bus cabin compared with either trap-outfitted diesel or CNG buses. These results also suggested the exhaust from the diesel test buses was a significant source of black carbon and PAH inside the cabin during the commute. In addition, both the conventional diesel and trap-outfitted diesel buses contributed approximately twice as much NO_2 to concentrations inside the bus cabin compared with the CNG bus.

5.3.5.2.3 The Effect of Roadway Type: Freeway versus Surface Streets

Again, only those runs on the primary urban route (U1) were included in the analysis of the effect of roadway type (freeway versus surface streets) on concentrations inside the bus. The second urban route (U2) consisted of all surface streets, and no freeway travel, so no comparison of roadway type was possible for that route. Mean concentrations of black carbon were approximately three times higher on surface streets compared with the freeway for diesel buses and the CNG bus, while for the trap-outfitted diesel bus, concentrations were higher on the freeway. In order to explain these differences, it was necessary to account for other potentially influential variables, such as the type of vehicle in front of the bus (see below).

Table 5.3.5.6 shows the black carbon concentrations for surface streets and freeways on the three different test bus types, grouped into periods with or without a diesel vehicle in front of the bus. The most interesting result we found was that the presence of a diesel vehicle in front of the bus was a more dominant determinant of bus cabin concentrations than roadway type. First,

we found that the trap-outfitted diesel bus had fewer encounters with other diesel vehicles on surface streets than on the freeway, while the opposite was true for the diesel buses and the CNG bus, which had more encounters with other diesel vehicles on surface streets. As encounters with other diesel vehicles appear to be the dominant determinant of high peak concentrations inside the test buses, this result helped explain why when looking only at mean concentrations on each roadway type without further separation by presence of other diesel vehicles, the diesel buses and the CNG bus were higher on surface streets (where they had more encounters with other diesel vehicles), while the trap-outfitted diesel was higher on the freeway (where it had more encounters with other diesel vehicles).

Table 5.3.5.6 Mean black carbon concentrations inside the test buses on different roadway types, stratified by the presence of a diesel vehicle in front of the bus.

Roadway Type	Diesel Vehicle Ahead of Test Bus	Mean Black Carbon Concentrations Inside the Test Buses (ug/m^3)		
		Diesel Buses	Trap Outfitted Diesel Bus	CNG Bus
SURFACE STREETS	**NO**	4.3	1.7	1.5
FREEWAY	**NO**	3.2	3.4	1.6
SURFACE STREETS	**YES[1]**	23.2 (19%, 93%)	10.5 (10%, 91%)	13.1 (19%, 56%)
FREEWAY	**YES[1]**	4.6 (10%, 26%)	5.9 (20%, 44%)	2.4 (10%, 32%)

Notes:
1: First number in parenthesis is the percent of the total commute time represented by this category.
Second number in parenthesis is the percentage of the vehicle encounters in this category that were diesel school buses.

In addition, the concentrations measured inside the buses were highest for all three bus types when diesel vehicles were encountered on surface streets. This was likely due to the type of diesel vehicle typically encountered on surface streets versus freeway. For all the test bus types, the majority of encounters with other diesel vehicles on surface streets were with diesel school buses (between 56 and 93% of all encounters on surface streets), while the majority of encounters on the freeway were with medium or heavy duty diesel trucks. This finding again demonstrated the potential of diesel school buses to contribute high concentrations of black

carbon (and by implication, other diesel-related pollutants) to a commuting child's exposure regardless of the type of vehicle the child may be riding in. Exhaust pipe location may also have played a role here, as most diesel school buses encountered had low exhaust pipes, while most heavy duty diesel trucks had high exhaust pipes. Other studies have found the position of a diesel vehicle's exhaust pipe was an important determinant of pollutant concentrations in a passenger car traveling behind a diesel vehicle (Fruin, 2003).

Mixed results between bus types were obtained for the comparison of roadway type when no diesel vehicles were present (Table 5.3.5.6, Figures 5.3.5.18 – 5.3.5.20). Statistical tests of significance were done to compare surface streets versus freeways when no other diesel vehicles were present for each of the test bus types. The CNG bus had no significant difference between the two roadway types, while both the diesel buses and the trap-outfitted bus were significantly different, although in the opposite direction (e.g. surface streets were higher on the diesel buses, while the freeway was higher on the trap bus).

One explanation for this difference between buses was likely due in part to differences in average speeds on the different roadway types in combination with the different contribution from each bus's own exhaust, depending on bus type. Notice that the mean concentrations of black carbon measured inside both the diesel buses and the trap-outfitted diesel bus while traveling on the freeway when other diesel vehicles were not present was approximately the same (3.2 and 3.4 ug/m3, respectively). For both of these bus types, the average speed on the freeway was about double the average speed on surface streets. Thus, we would expect much higher ventilation rates, and as the windows were open, we would expect the outside air, or the

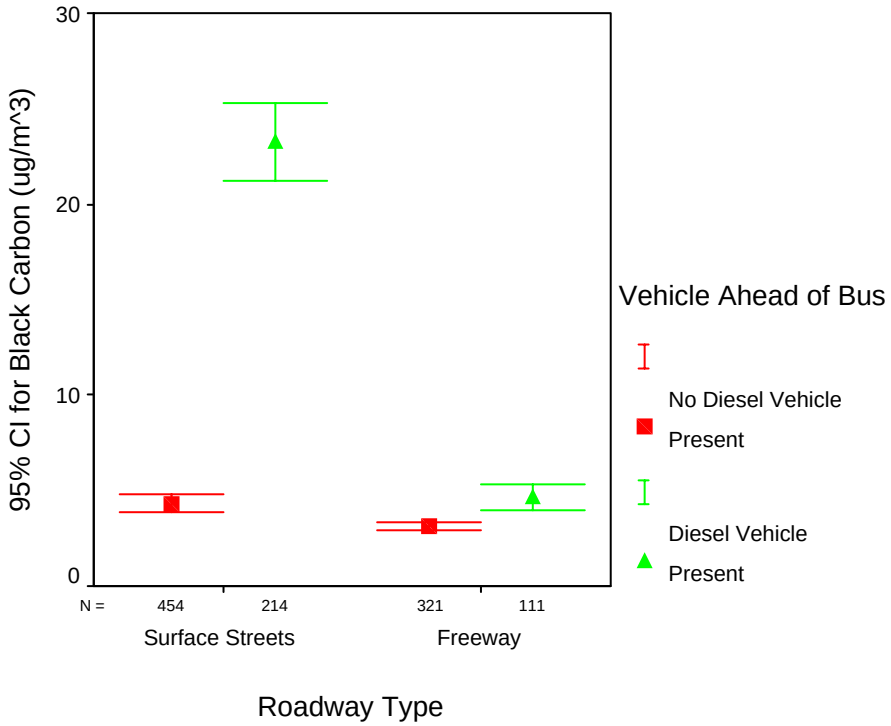


Figure 5.3.5.18 Mean and 95% confidence intervals for black carbon inside diesel school buses on different roadway types with or without other diesel vehicles present.

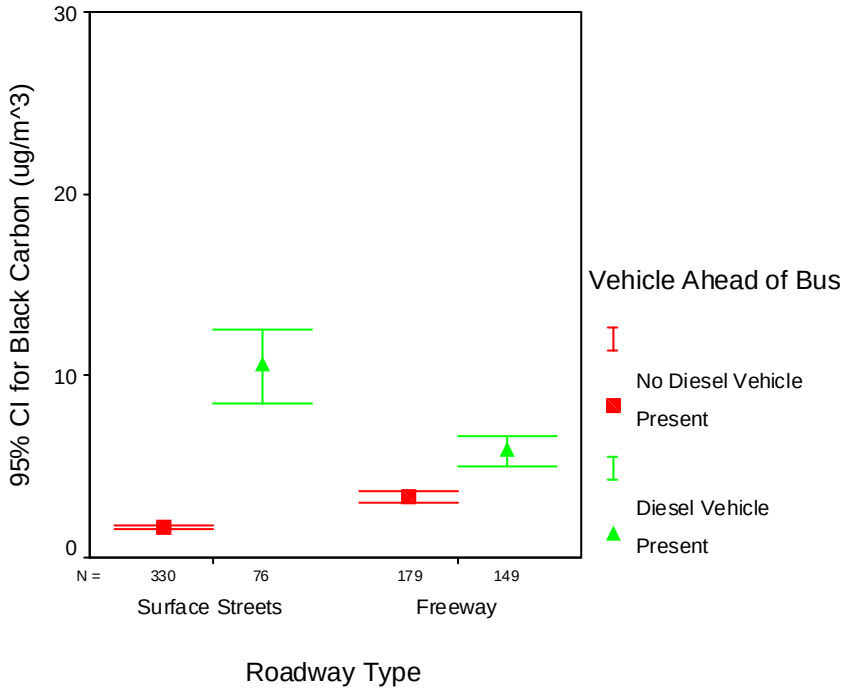


Figure 5.3.5.19 Mean and 95% confidence intervals for black carbon inside the trap-outfitted diesel school bus for different roadway types with or without other diesel vehicles present

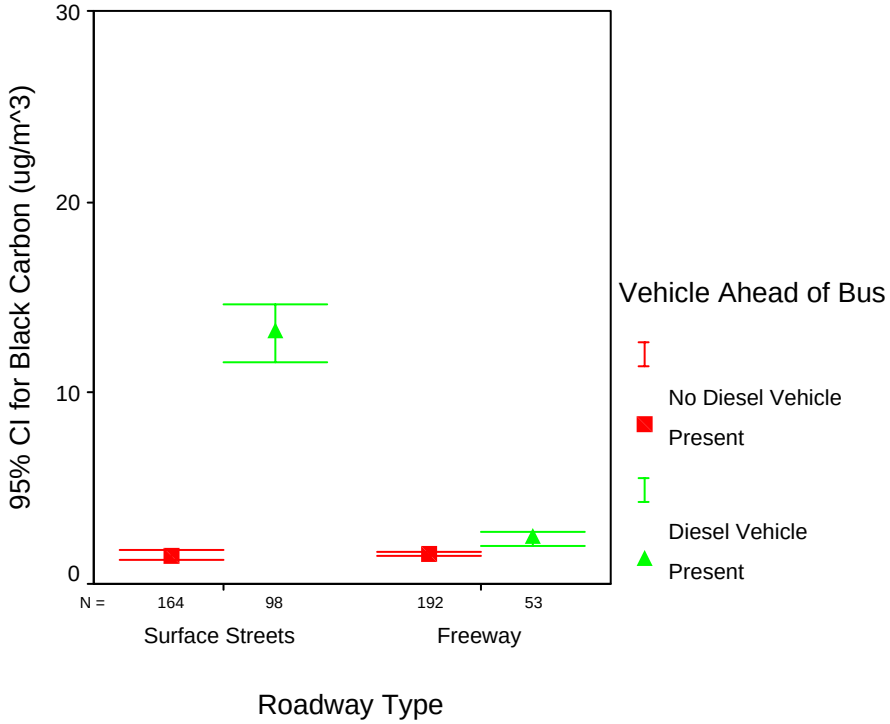


Figure 5.3.5.20 Mean and 95% confidence intervals for black carbon inside the CNG school bus for different roadway types with or without other diesel vehicles present.

average concentration in the line source, was the dominant determinant of concentrations inside the bus, when other diesel vehicles were not present and when the buses were moving at relatively high speeds (greater than 30 km/hr). Under these conditions, the dominant determinant of concentrations inside the bus cabin was the average concentration in the line source.

By contrast, on surface streets, with much lower speeds and more periods of idling, the concentrations inside the bus, when other diesel vehicles were not present, were dominated both by average concentrations in the road, and by the contribution of the bus's own exhaust. The trap-outfitted diesel bus likely had lower concentrations on surface streets, under the conditions described above, because the contribution from the bus itself was lower. The diesel buses' own exhaust contributed more black carbon to the concentrations inside the cabin while on surface streets than did the trap-outfitted or CNG bus, resulting in higher concentrations on surface streets inside the bus cabin. The CNG bus, which showed no significant difference between surface streets and freeways also had similar average speeds on both roadway types (17 and 22 km/hr), indicating similar traffic congestion levels, and thus similar line source concentrations.

These results indicated the importance of roadway type on concentrations inside the bus cabin depended on the type of bus. For relatively clean buses (e.g. CNG, trap-outfitted diesel), traveling in a line source with higher concentrations, such as on freeways, tended to increase concentrations inside the bus. For a dirtier buses (e.g. diesel), because the bus's own exhaust was a significant contributor to concentrations inside the bus, increasing the ventilation rate inside the bus, which was more likely to occur on freeways, tended to decrease concentrations inside the bus.

5.3.5.2.4 Effect of Route Type

Two different urban routes were driven by the same diesel bus (RE2) during two consecutive weeks. For the comparison of routes, only these runs (Run 20 - driven on the primary urban route (U1) and Run 26 - driven on urban route 2 (U2)) were used. By maintaining the same bus on both routes, we hoped to eliminate any confounding due to differences between buses.

Table 5.3.5.7 shows the comparison between U1 and U2 for variables such as distribution of road types (freeway, major/minor arterials and small residential streets), average speed, percent of time spent following other diesel vehicles and behind diesel school buses, number of bus stops, etc. One of the major differences between these two routes was that U2 involved only surface streets, while U1 was a mix of surface streets and freeways. In Section 5.3.5.2.3 we concluded for diesel buses, freeway travel which allowed higher speeds and increased ventilation in the bus cabin resulted in lower concentrations of black carbon. Given that result, we might expect a bus route with no freeway travel to have higher concentrations.

Table 5.3.5.7 Comparison between urban route one and urban route two.

	Urban Route One (U1)	**Urban Route Two (U2)**
Distribution of Roadway Types:		
Freeway	42%	0
Major/Minor Arterials	37%	56%
Small Residential Streets	21%	44%
Speed:		
Average Speed - Entire Commute (km/hr)	26.3	21.4
Average Speed - Freeway	39.5	N/A
Average Speed - Major/Minor Arterials	20.4	27.4
Average Speed - Residential Streets	11.4	13.6
Other:		
Number of Bus Stops	5	10
Time Spent at Bus Stops	7%	14%
Time Spent Behind Other Diesel Vehicles	28%	13%
Time Spent Behind Other Diesel School Buses	18%	11%
Time Spent Idling	19%	36%

Tables 5.3.5.8 and 5.3.5.9 show the concentrations of black carbon and PAH measured inside the bus on U1 and U2. Figures 5.3.5.21 and 5.3.5.22 show the 95% confidence intervals and boxplots for the concentrations measured on the entire routes. We found the mean concentrations for the entire runs were not significantly different between routes. We also found similar ranges of concentrations on both routes (Figure 5.3.5.22). However it was also interesting to compare the concentrations measured on different roadway types for each route (see Tables 5.3.5.8-5.3.5.9 and Figure 5.3.5.23). First we found the freeway portion of Run 20 on U1 had lower concentrations compared to the other two roadway types, as we would expect based on our results from Section 5.3.5.2.3 in which we found lower concentrations on the freeway versus surface streets for diesel buses. Next, concentrations of both black carbon and PAH were similar on major/minor arterial roads on U1 and U2, although slightly higher on U1. The major/minor arterial streets on both routes had similar mixes of vehicles, but when we examined average speeds on the different road types, we found slightly reduced average speeds for the same roadway types on U1 compared with U2, indicating higher concentrations of traffic congestion on U1. This is consistent with the fact that although both routes were in urban areas, U1 had higher density neighborhoods compared to U2. This also explains why although both routes traveled through small residential streets for a portion of the run, U1 had higher

Table 5.3.5.8 Mean black carbon concentrations measured inside the bus on urban route one and urban route two (Runs 20 and 26). Mean black carbon (ug/m^3).

	Total Run	Freeway	Major/Minor Arterials	Small Residential Streets	Not Idling No Vehicle in Front	Idling No Vehicle in Front	Diesel Vehicle in Front
Urban Route One (U1)	7.2	4.1	10.8	6.9	4.1	6.2	19.3
Urban Route Two (U2)	7.3	N/A	10.2	3.4	3.9	4.7	22.8

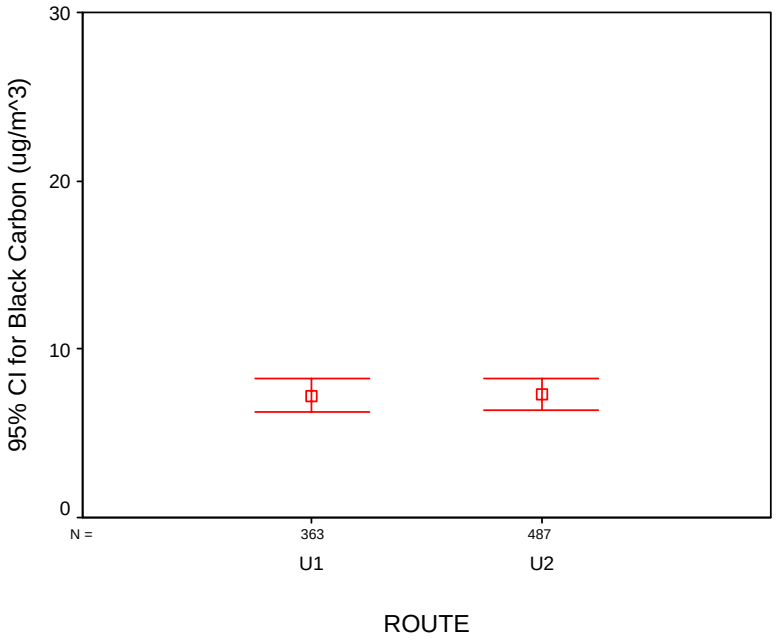
Table 5.3.5.9 Mean PAH concentrations inside the bus on urban route one and urban route two (Runs 20 and 26). PAH (ng/m^3).

	Total Run	Freeway	Major/Minor Arterials	Small Residential Streets	Not Idling No Vehicle in Front	Idling No Vehicle in Front	Vehicle in Front
Urban Route One (U1)	113	46	188	111	55	70	365
Urban Route Two (U2)	122	N/A	192	26	41	51	508

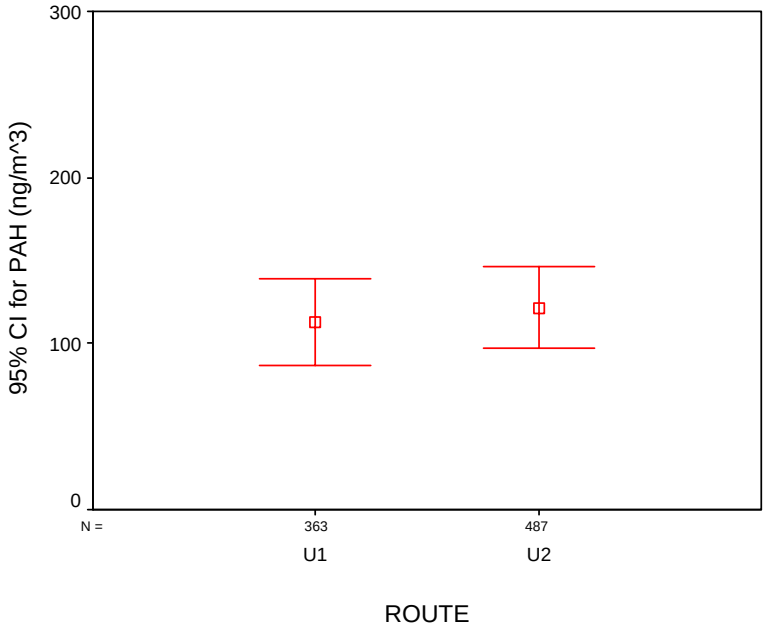
concentrations on these streets. Again, average speeds were slightly lower on residential streets on U1, and the residential streets were generally located between highly congested arterials. By contrast, on U2, the majority of the residential streets were in neighborhoods with low density housing and few other vehicles, and were separated from congested arterials by greater distances compared with U1.

Despite these minor differences, however, our results show that both urban routes were comparable in terms of in-bus concentrations over the course of the commute. These results also emphasize our previous finding that freeway travel, which increases the average speed of the bus, tended to reduce concentrations inside diesel buses. In the case of U1, the reduced concentrations on the freeway portion of the run were offset by higher concentrations on the surface streets. U2, which had no freeway travel, had slightly lower surface street concentrations compared with U1, but these concentrations were still higher than those experienced on U1 while on the freeway.

In addition, from the comparisons in Table 5.3.5.7, we also found that for both routes, concentrations inside the bus cabin increased five to six times when following another diesel vehicle. Encounters with other diesel vehicles were less frequent on U2 than on U1 as a percentage of the total run. However, 82% of all encounters with other diesel vehicles on U2 were with diesel school buses, while on U1, there were more total encounters with other diesel vehicles, but only 63% of these were with other diesel school buses. For both routes, a similar

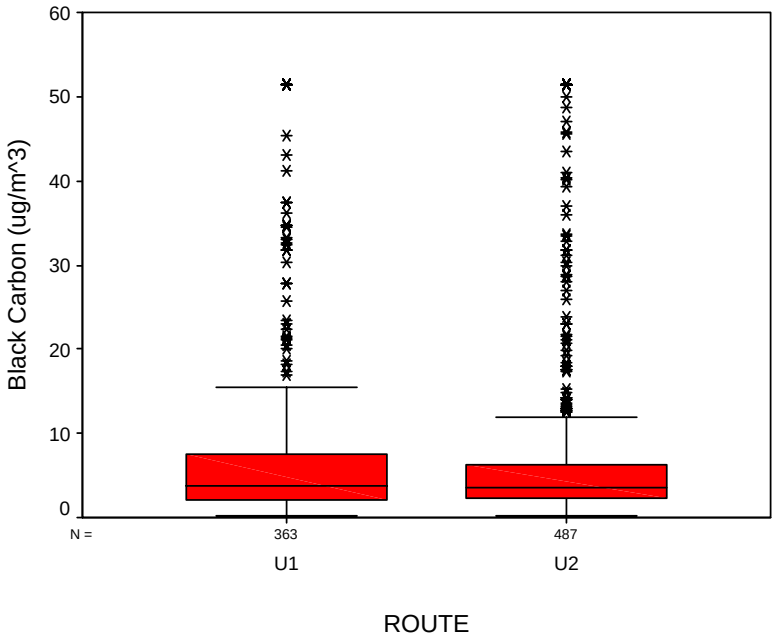


(a)

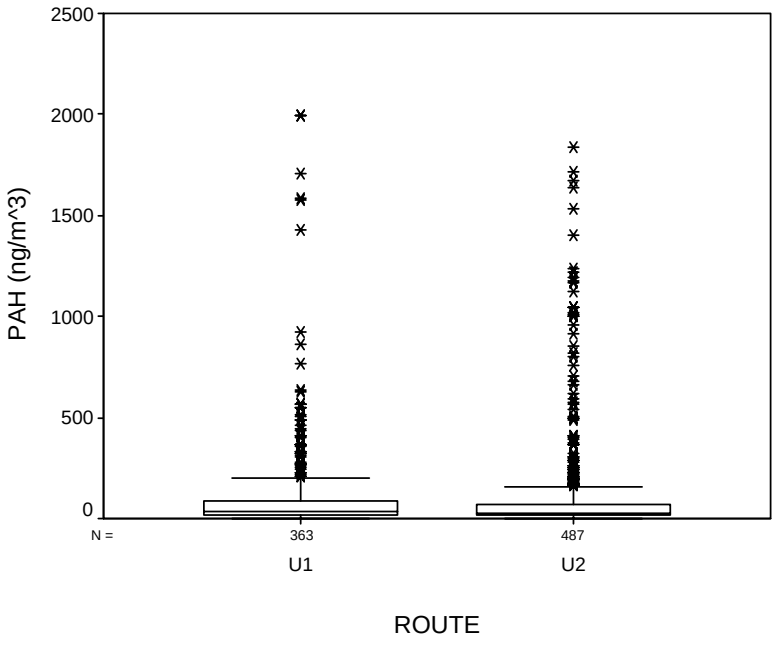


(b)

Figure 5.3.5.21 Means and 95% confidence intervals for black carbon (a) and PAH (b) concentrations measured inside school bus RE2 while traveling on two different urban routes.

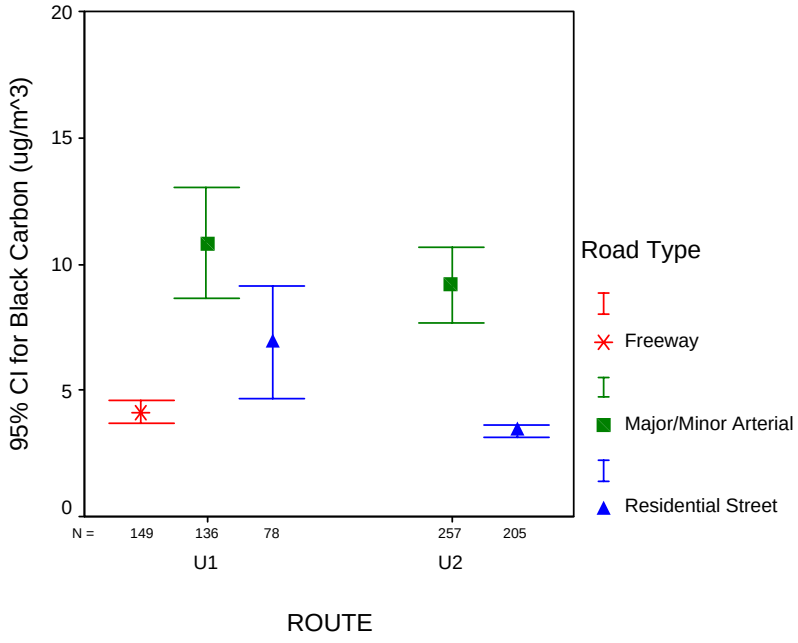


(a)

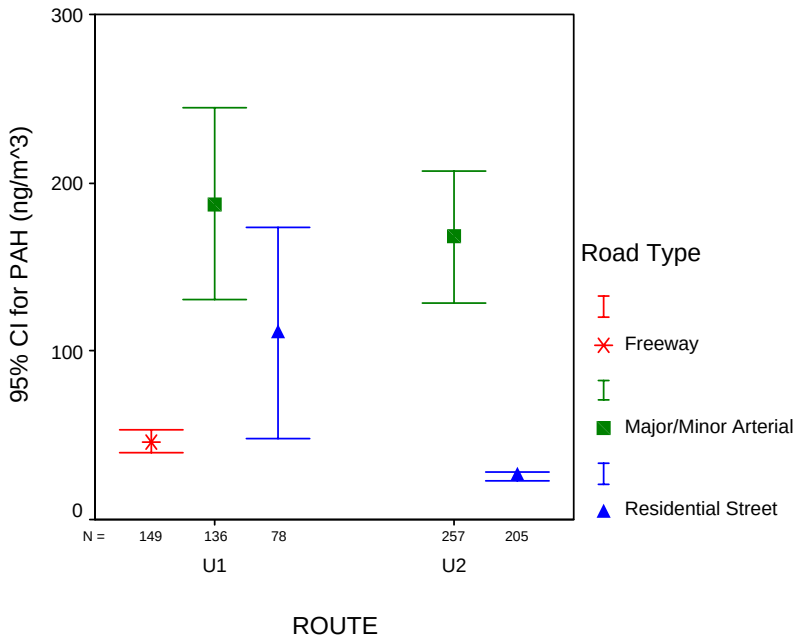


(b)

Figure 5.3.5.22 Boxplots of black carbon (a) and PAH (b) concentrations measured inside school bus RE2 while traveling on two different urban routes.



(a)



(b)

Figure 5.3.5.23 Means and 95% confidence intervals for black carbon (a) and PAH (b) concentrations measured inside school bus RE2 for two urban bus routes, divided by roadway type.

amount of time was spent traveling behind or adjacent to diesel school buses, with approximately ten and eight minutes, respectively, of U1 and U2 spent behind or adjacent to another diesel school bus.

The most interesting aspect of the comparison of the two urban routes was that we found both routes to have similar concentrations inside the bus, both in terms of the ranges of concentrations experienced, and the average over the commute. This despite the fact that both routes had a variety of different features, including the mix of roadway types and traffic congestion scenarios. Features which were similar between routes, such as encounters with other diesel vehicles, particularly diesel school buses, dominated the highest peak concentrations on both routes.

5.3.6 Variables Investigated Using Paired Instruments

For a selected number of runs, two sets of instruments were used to simultaneously measure the front and rear of the bus. Section 5.1.3 of this report describes the results of collocation measurements taken each week during the study for the two sets of instruments.

5.3.6.1 Differences in Concentration Between the Front and the Rear of the Cabin

Pollutant concentrations at the front and rear of the cabin were measured during 10 of the 36 exposure runs, including five afternoon and five morning runs, all on urban route one. Three conventional diesel buses were used for these measurements: HE2, RE1, and RE2. Table 5.3.6.1 summarizes the characteristics of these runs.

We used two approaches to analyze the differences between front/rear concentrations: a study of the time series graphs and the calculation of the 95% confidence intervals of the average concentrations.

Table 5.3.6.1 Front versus rear comparison runs.

Run	Date	Time	Type of Bus	Type of Route	Window Position	Week Number
5	05/01/02	AM	HE2	U1	Closed	2
6	05/01/02	PM	HE2	U1	Open	2
12	05/14/02	AM	RE1	U1	Closed	4
13	05/14/02	PM	RE1	U1	Open	4
14	05/16/02	AM	RE1	U1	Closed	4
15	05/16/02	PM	RE1	U1	Open	4
17	05/21/02	AM	RE2	U1	Closed	5
18	05/21/02	PM	RE2	U1	Open	5
19	05/22/02	AM	RE2	U1	Closed	5
20	05/22/02	PM	RE2	U1	Open	5

5.3.6.1.1 PAH

In general, during the five morning runs, the front/rear time series tended to track each other, however, concentrations at the rear of the cabin showed more frequent and higher peaks, leading to slightly higher average concentrations. Run 14 was an exception in that the two time series did not track each other and the baseline of the time series for the rear of the cabin was

substantially higher. Figure 5.3.6.1 shows the PAH measurements during Run 12 (RE1-U1-AM), an example of a representative morning run. As discussed below, exhaust entered into the cabin creating the concentration gradient. Since the rear of the cabin was closer to the engine and its exhaust, the time series for this location exhibited a more dynamic behavior compared with the front.

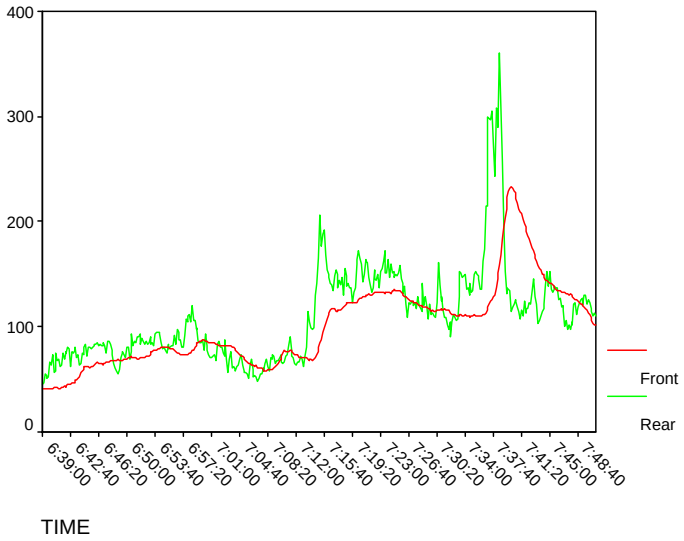


Figure 5.3.6.1 PAH concentrations (ng/m^3) inside the cabin during Run 12.

In general, the five afternoon runs showed similar trends, with the time series tracking each other (with the exception of a few concentration spikes). Figure 5.3.6.2 shows an example of a representative afternoon run, the 10-seconds data for PAH during Run 13 (RE1-U1-PM).

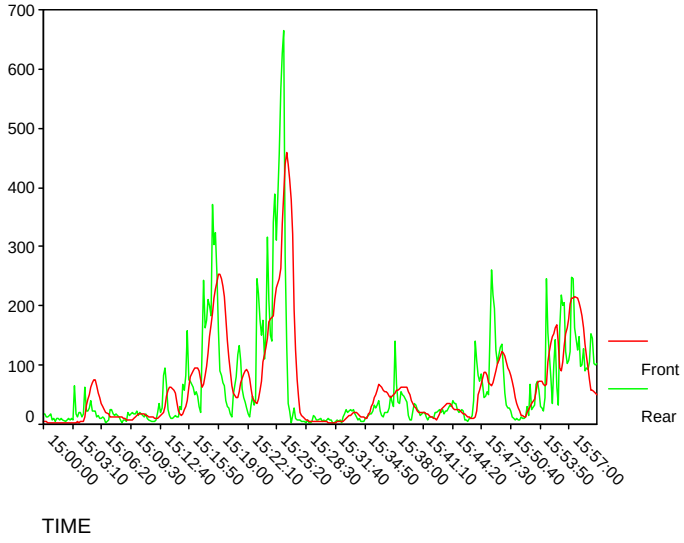


Figure 5.3.6.2 PAH concentrations (ng/m^3) inside the cabin during Run 13.

Figure 5.3.6.3 shows the means and 95% confidence intervals of the average PAH concentrations during the selected morning runs, with the windows closed. In general, the concentrations at the rear of the cabin were significantly higher than those at the front of the cabin for the conventional diesel buses.

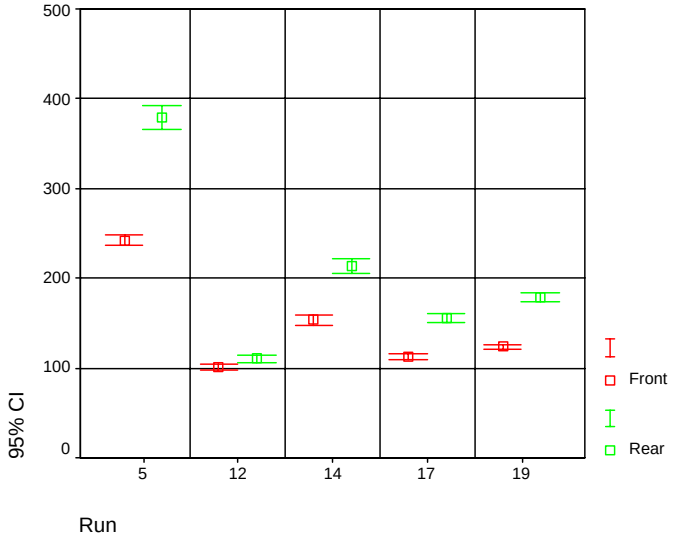


Figure 5.3.6.3 Means and 95% confidence intervals of the average PAH concentrations during morning runs (ng/m^3).

Figure 5.3.6.4 shows the 95% confidence intervals of the average PAH concentrations during the selected afternoon runs, with the windows opened. In general, the differences between the concentrations measured at the two locations were less substantial than during morning runs and in most cases those differences were not statistically significant.

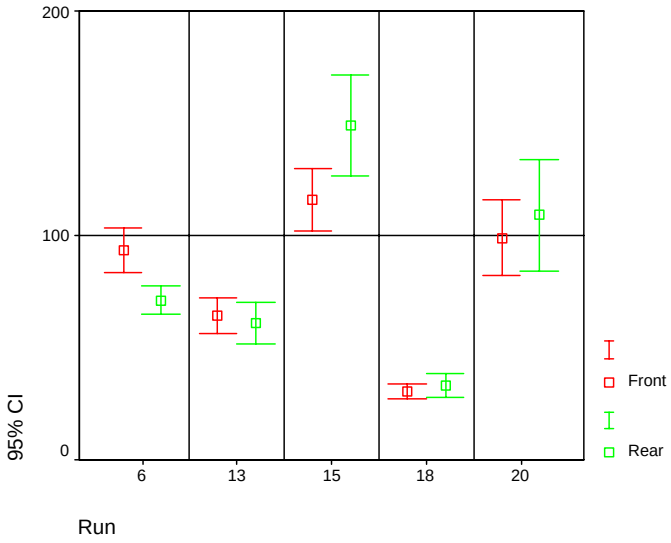


Figure 5.3.6.4 Means and 95% confidence intervals of the average PAH concentrations during afternoon runs (ng/m^3).

5.3.6.1.2 Particle Counts

During the morning runs, for particle counts, the pairs of time series followed similar trends and exhibited a similar number of peaks. However, the time series for the rear of the cabin showed higher baselines and higher peaks, yielding higher average concentrations (except for Run 17 which between the two time series did not show a high degree of correlation). Figure 5.3.6.5 shows the 10-seconds data for PC during Run 5 (HE2-U1-AM).

The pairs of time series for PC (0.3 to 0.5 μm) to the five afternoon runs exhibited strong overlap, as shown in Figure 5.3.6.6, the 10-second data for Run 13 (RE1-U1-PM).

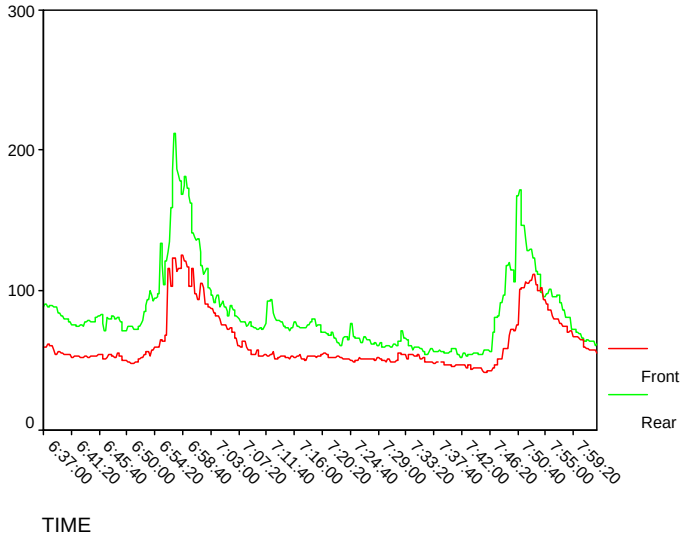


Figure 5.3.6.5 Concentrations (#/cm^3) of PC (0.3 to 0.5μm) during Run 5.

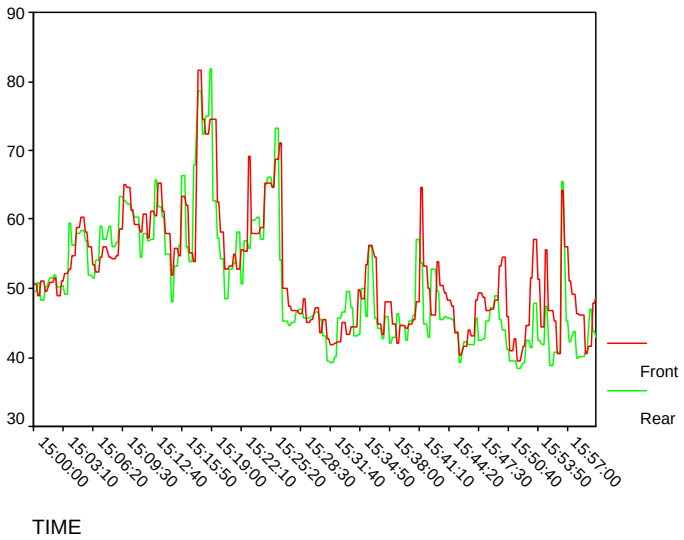


Figure 5.3.6.6 Concentrations (#/cm^3) of PC (0.3 to 0.5μm) during Run 13.

Figure 5.3.6.7 shows the means and 95% confidence intervals of the average PC concentrations during the selected morning runs, with the windows closed. In general, the concentrations at the rear of the cabin were significantly higher than those at the front of the cabin, for all bus types.

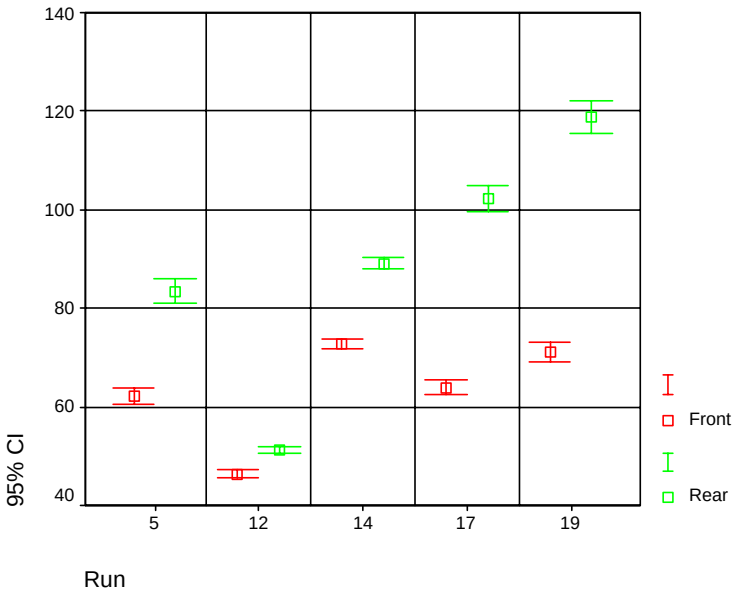


Figure 5.3.6.7 Means and 95% confidence intervals of the average PC concentrations (0.3 to 0.5 µm) at front and rear of cabin during morning runs (#/cm^3).

Figure 5.3.6.8 shows the means and 95% confidence intervals of the average PC concentrations (0.3 to 0.5 µm) during the selected afternoon runs, with the windows partially opened. In general, the differences between the concentrations measured at front versus rear were not significant (except for Run 20 where the concentrations at the rear of the cabin were significantly higher than those at the front). After reviewing the real time data for this run, we determined this was due to a few of high concentrations peaks the instruments at the front did not capture, probably caused by a momentary malfunction.

In summary, both pollutants PAH and PC exhibited higher concentrations at the rear of the cabin when the windows were closed (morning runs), due to exhaust leaking into the cabin as demonstrated by the analysis of the SF_6 data (see Section 5.3.1).

For the runs where the windows were partially opened (afternoon), the front/rear concentrations were similar for the two pollutants, consistent with other analyses reported here suggesting that when the windows were opened, higher ventilation rates promoted uniform mixing within the cabin

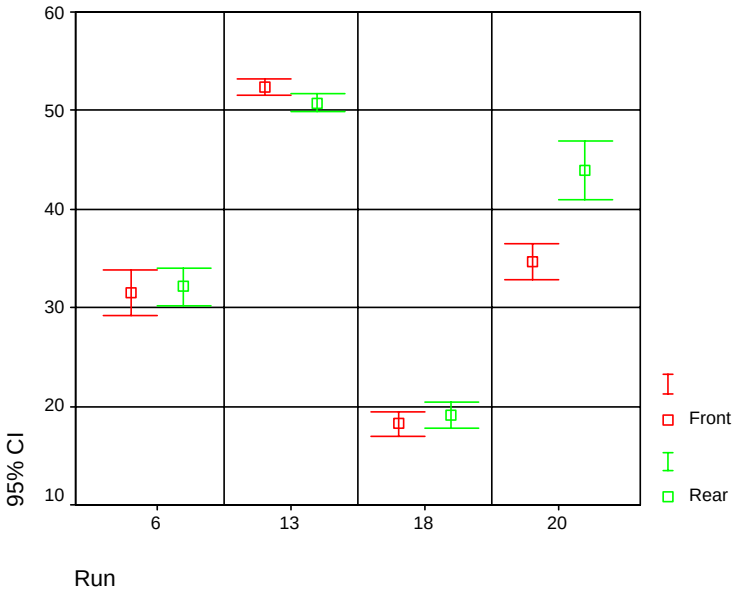


Figure 5.3.6.8 Means and 95% confidence intervals of the average PC concentrations (0.3 to 0.5 μm) at front and rear of cabin during afternoon runs (#/cm^3).

5.3.6.1.3 Correlation Between Front and Rear Concentrations

As described in previous sections we could quantify the differences between the front and rear using the ratio of the average concentrations. Table 5.3.6.2 summarizes these results. The ratios were calculated using the 10-second data of the selected runs. In addition, the 95% confidence intervals of the average of those ratios were computed and also shown.

Table 5.3.6.2 Observed rear-to-front ratios.

	PAH Ratio (rear to front)	PC Ratio (rear to front)
Morning (windows closed)	1.4 ± 0.07	1.4 ± 0.03
Afternoon (windows open)	1.3 ± 0.17	1.1 ± 0.06

5.3.6.2 Differences in Concentration Inside and Immediately Outside the Bus

Pollutant concentrations inside and immediately outside the bus were measured during 10 of the 36 exposure runs. These experiments consisted of four afternoon runs with windows open (using U1), three morning runs with windows closed (using U1), and three runs using the rural/suburban route (all of them with windows open). Three bus types were used for these measurements, including a conventional diesel bus (HE3), the particle trap-outfitted diesel bus (TO1) and the CNG bus. Table 5.3.6.3 summarizes the characteristics of these runs. Because of

a problem with the sampling line, outside measurements were not made on the conventional diesel bus (HE3) during any runs with the windows closed.

Table 5.3.6.3 Inside/outside comparison runs

Run	Date	Time	Type of Bus	Type of Route	Window Position	Week Number
7	05/07/02	PM	HE3	RS	Open	3
10	05/08/02	PM	HE3	U1	Open	3
27	06/04/02	PM	TO1	RS	Open	7
28	06/05/02	AM	TO1	U1	Closed	7
30	06/05/02	PM	TO1	U1	Open	7
31	06/06/02	AM	TO1	U1	Closed	7
33	06/06/02	PM	TO1	U1	Open	7
34	06/11/02	PM	CNG	RS	Open	8
35	06/12/02	AM	CNG	U1	Closed	8
36	06/12/02	PM	CNG	U1	Open	8

We used two methods to analyze this set of runs. First, we produced time series graphs for each run to study the concentration variations within runs and also to examine whether the two series (inside and outside) tracked each other. Second, we calculated the 95% confidence intervals of the concentration means to establish if the differences between the average concentrations (inside versus outside) were statistically significant.

As discussed in Section 5.1.3.3, the PC and the PAH analyzers exhibited the highest precision between paired instruments used in the study. In the same section, we discussed the strong correlations between PAH and black carbon and between PC and $PM_{2.5}$. Hence, we focused on PC and PAH measurements when performing the comparisons between the inside and outside concentrations. Based on the analysis of the correlation coefficients, we assumed $PM_{2.5}$ and black carbon mimic the behavior of PC and PAH, respectively.

5.3.6.2.1 PAH

During the three rural/suburban runs with the windows open, the inside versus outside concentration time series tended to track each other: when the outside values were high, the inside values were also high and vice versa. However, the outside time series exhibited a more dynamic behavior, with more frequent and higher peaks. The four runs on U1 with the windows open showed similar trends, with the outside time series exhibiting more variability.

During the three morning runs on U1 with the windows closed (on the particle-trap outfitted diesel and the CNG buses), the inside time series exhibited few peaks but the outside time series again showed more dynamic behavior with a variety of peaks. As an example, Figure 5.3.6.9 shows 10-second resolution data for PAH during Run 28 (AM-TO1-U1).

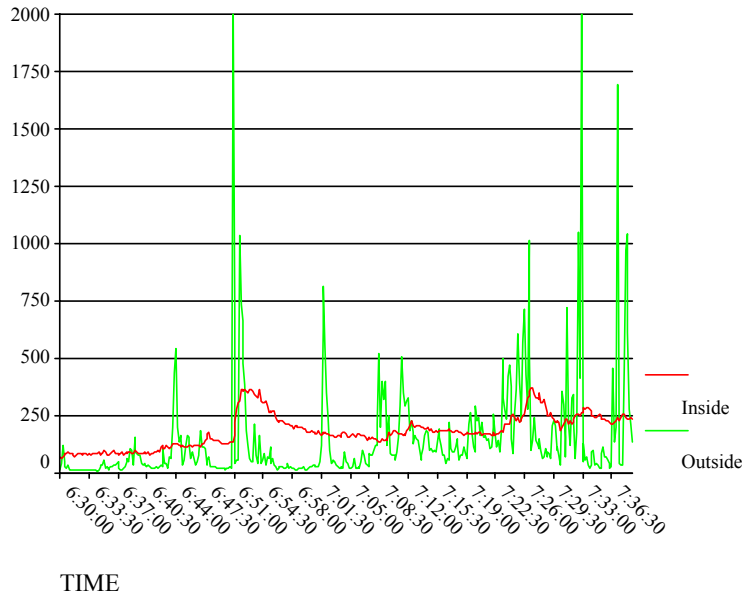
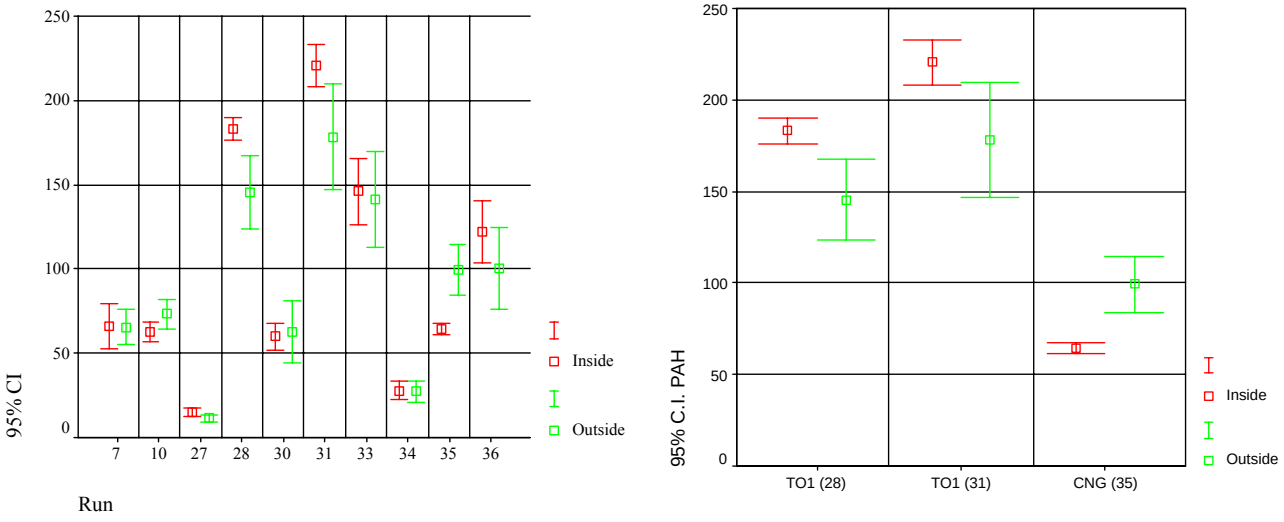


Figure 5.3.6.9 Inside (rear) and outside concentrations (ng/m^3) of PAH during Run 28 (urban, morning, windows closed).

Figure 5.3.6.10 shows the means and 95% confidence intervals of the average PAH concentrations for selected runs. During Runs 28 and 31 on the particle trap-outfitted diesel bus with the windows closed, the concentrations inside the cabin were clearly higher than outside. However, for the run on the CNG bus with the windows closed (Run 35), concentrations outside the bus were higher, indicating lower contribution from self pollution inside the cabin of the CNG bus.



(a) All available runs (see table 5.3.6.3) (b) Runs with windows closed only

Figure 5.3.6.10 Means and 95% confidence intervals of the average PAH concentrations (ng/m^3) for selected runs.

For the remaining afternoon runs with the windows open, differences in concentrations between inside and outside were not significant.

5.3.6.2.2 Particle Counts

For PC, for the three rural/suburban runs with the windows open, the inside and outside concentration time series behaved similar to each other, following the same trends and with similar magnitudes of peak concentrations.

The four afternoon runs for U1 with the windows open exhibited almost the same trends as those for the rural/suburban route, with inside/outside time series following similar patterns except that, in general, the outside time series showed slightly more dynamic behavior for U1.

During the three morning runs on U1 with the windows closed (includes the particle trap-outfitted diesel and the CNG buses), the pairs of time series again followed similar trends; however, the outside concentrations (both baseline and peaks) were higher than the concentrations inside the cabin, the opposite of what we found for PAH.

Figure 5.3.6.11 shows an example of a time series based on 10-second resolution data for PC during Run 33 (PM-TO1-U1-windows open).

Figure 5.3.6.12 shows the 95% confidence intervals of the average PC concentrations during the selected runs. For the morning runs with windows closed (28, 31, 35), the differences between the inside and outside concentrations were not only significant but also substantial (outside higher).

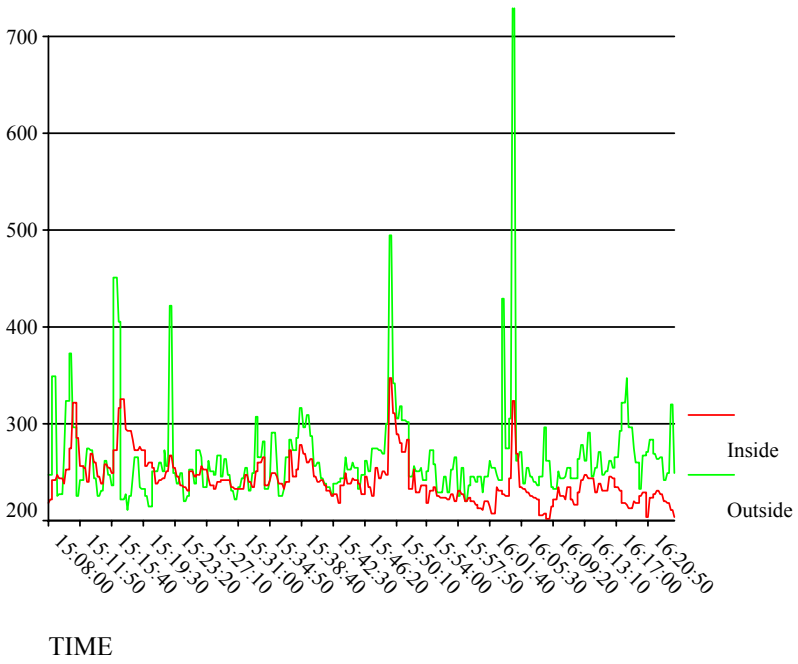


Figure 5.3.6.11 Concentrations (#/cm^3) of PC (0.3 to 0.5µm) during Run 33.

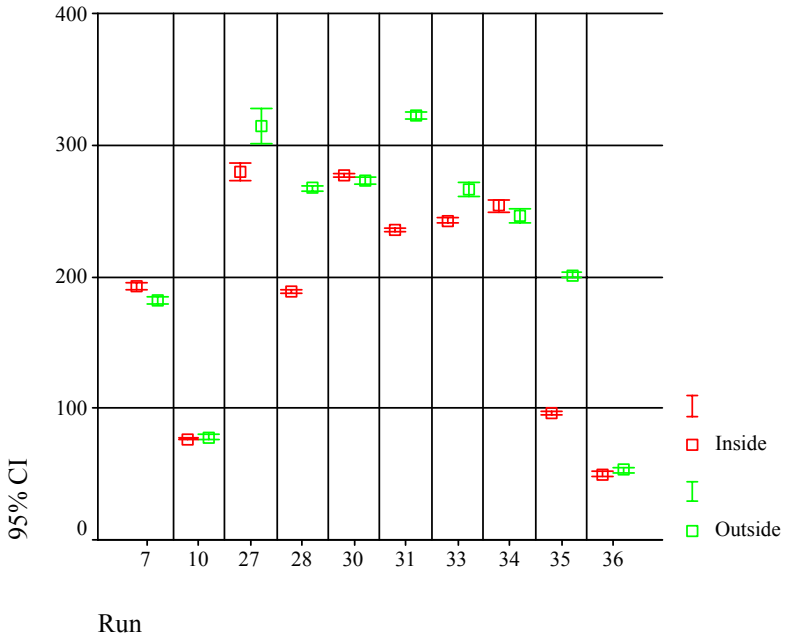


Figure 5.3.6.12 Mean and 95% confidence intervals for PC (#/cm^3).

During the afternoon runs with windows open, the differences between inside versus outside were, in general, not statistically significant, except for Runs 27 and 33, where the concentrations outside the cabin were significantly higher.

5.3.6.2.3. Implications of the Results

Table 5.3.6.4 summarizes the major findings of the analyses described in this section.

Table 5.3.6.4 Summary of major findings of inside versus outside comparisons.

	PAH (Directly emitted pollutant)	
	Inside	Outside
Windows closed (morning)	Flat time series Higher average concentrations (except CNG)	Dynamic time series Lower average concentrations
Windows open (afternoon)	Time series similar to outside Average concentrations similar to outside	Time series slightly more dynamic Average concentrations similar to inside
	PC in the accumulation mode (Background pollutant)	
	Inside	Outside
Windows closed (morning)	Time series similar to outside Average concentrations lower than outside	Time series slightly more dynamic Average concentrations higher than inside
Windows open (afternoon)	Time series similar to outside Average concentrations similar to outside	Time series slightly more dynamic Average concentrations similar to inside

These results also suggest there is a significant difference in behavior between directly emitted pollutants, such as BC, PAH, and NO_2, and "background" pollutants such as $PM_{2.5}$ and PC (in the accumulation mode), consistent with analyses presented in other sections of this report (5.1.3, 5.2.1, and 5.3.1) (although we emphasize that both $PM_{2.5}$ and PC can have a direct emissions component). These findings demonstrate the complex implications for effects of windows position, as explained in Section 5.2.2.3.

5.3.6.2.4 Correlation between Inside and Outside Concentrations

We have described the key factors governing the differences in pollutant concentrations inside and outside the cabin. However, the analysis above did not include the quantification of those differences.

Outdoor-to-indoor ratios are widely used in the analysis of indoor air pollutant data (Fischer et al., 2000; Janssen et al, 2001; Chaloulakoua and Mavroidis, 2002). We used an analogous parameter defined as the ratio of outside-to-inside bus cabin concentrations.

$$\alpha_j = \frac{C_{Outside,j}}{C_{Inside,j}} \qquad (5.3.6.2.1)$$

where α_j is the outside-to-inside concentrations ratio for the pollutant j; $C_{Outside,j}$ is the concentration of pollutant j measured immediately outside the bus; and $C_{inside,j}$ is the concentration of pollutant j measured inside the cabin.

These ratios were calculated using the 10-seconds data of Runs 28, 31, and 35 (U1 with windows closed on TO1 and CNG), and Runs 30, 33, and 36 (U1 with windows open on TO1 and CNG). In addition, the 95% confidence intervals of the mean of these ratios were computed. Table 5.3.6.5 summarizes these results and demonstrates the outside-to-inside ratio is a function of several factors, including type of pollutant, bus type, and window position. These quantitative results support the findings described in the preceding sections.

Table 5.3.6.5 Observed outside-to-inside concentration ratios for PAH and PC for CNG and particle trap-outfitted buses.

	PAH (ng/m^3)	PC ($\#/cm^3$)
Windows closed (morning)	0.75 ± 0.2 (TO1) 1.54 ± 0.2 (CNG)	1.40 ± 0.04 (TO1) 2.14 ± 0.04 (CNG)
Windows open (afternoon)	0.99 ± 0.2 (TO1)	1.06 ± 0.08 (TO1)

5.4 Comparison of Pollutant Concentrations Measured Inside the Buses and at Nearby AQMD Monitoring Sites

5.4.1 Carbon Monoxide and Nitrogen Dioxide

CO and NO_2 ambient monitoring data were obtained from the South Coast Air Quality Management District (SCAQMD) for comparisons between inside bus concentrations and ambient concentrations. The Central LA and West LA SCAQMD stations (see Photograph 17) were selected as the closest monitoring sites to urban route one. We took the average of the data from the two monitoring sites as best representing ambient concentrations near urban route one. We chose AQMD data corresponding to the date and time of each run. Sections 5.4.1.1 and 5.4.1.2 discuss results for CO and NO_2, respectively.

Photograph 17. West L.A. SCAQMD Station

5.4.1.1 Carbon Monoxide

Tables 5.4.1.1 and 5.4.1.2 show the mean CO concentrations measured inside the buses and at the two nearest AQMD stations for morning (windows closed) and afternoon (windows open) runs on urban route one, respectively.

Mean CO concentrations were compared separately, morning versus afternoon, since earlier studies showed in-vehicle/fixed site ratios were higher for afternoon/evening hours than for the morning hours, with fixed-site monitors exhibiting lower concentrations in the afternoon (Petersen and Allen 1982; Fernandez-Bremauntz and Ashmore 1995b). However, in the present study we found no difference between morning and afternoon in-vehicle/fixed-site ratios, with on average the same ratio of 0.7. However, as shown in Tables 5.4.1.1 and 5.4.1.2, the absolute concentrations of CO were significantly different between morning and afternoon, with higher concentrations both inside the buses and at the AQMD stations in the morning. The overall mean CO concentration inside buses in the morning was 5.2 ppm versus 2.4 ppm in the afternoon, while the overall mean of the two AQMD stations in the morning was 6.9 ppm versus 3.6 ppm in the afternoon.

Table 5.4.1.1 Mean CO concentrations (ppm) inside buses and at two nearby AQMD stations for the morning runs (windows closed) on urban route one.

Date	Run	Week	Bus	Inside bus	West LA	Central LA	Mean[1]	Ratio[2]
05/01/02	5	2	HE2	4.5	6.5	6.5	6.5	0.7
05/08/02	8	3	HE3	4.0	4.0	5.5	4.8	0.8
05/14/02	12	4	RE1	5.8	9.5	7.0	8.3	0.7
05/16/02	14	4	RE1	4.3	2.5	4.5	3.5	1.2
05/21/02	17	5	RE2	6.0	4.0	6.0	5.0	1.2
05/22/02	19	5	RE2	3.7	7.0	8.0	7.5	0.5
06/05/02	28	7	TO1	5.7	6.0	12.0	9.0	0.6
06/06/02	31	7	TO1	6.4	8.5	9.0	8.8	0.7
06/12/02	35	8	CNG	6.2	3.0	15.0	9.0	0.7
			Mean	5.2			6.9	0.7

1. Arithmetic mean of west LA and central LA stations.
2. Ratio = (mean of inside bus) / (mean of two stations)

Table 5.4.1.2 Mean CO concentrations (ppm) inside buses and at two nearby AQMD stations for the afternoon runs (windows open) on urban route one.

Date	Run	Week	Bus	Inside bus	West LA	Central LA	Mean[1]	Ratio[2]
05/01/02	6	2	HE2	2.1	2.0	3.0	2.5	0.8
05/08/02	10	3	HE3	N/A	2.0	4.0	3.0	
05/14/02	13	4	RE1	2.6	2.5	6.0	4.3	0.6
05/16/02	15	4	RE1	2.1	2.0	5.0	3.5	0.6
05/21/02	18	5	RE2	2.0	2.0	3.0	2.5	0.8
05/22/02	20	5	RE2	1.9	2.0	3.0	2.5	0.8
06/05/02	30	7	TO1	2.9	3.0	8.5	5.8	0.5
06/06/02	33	7	TO1	2.9	3.5	7.0	5.3	0.6
06/12/02	36	8	CNG	2.8	1.5	4.0	2.8	1.0
			Mean	2.4			3.6	0.7

1. Arithmetic mean of west LA and central LA stations.
2. Ratio = (mean of inside bus) / (mean of two stations)

Interestingly, the trends of CO concentrations inside the buses and at the ambient sites tracked each other over the main study period (see Figures 5.4.1.1 (a) and (b) for morning and afternoon, respectively). Figure 5.4.1.1 shows CO concentrations inside the buses were consistently lower than those at the ambient sites except for Run 14 and 17 in the morning and Run 36 in the afternoon.

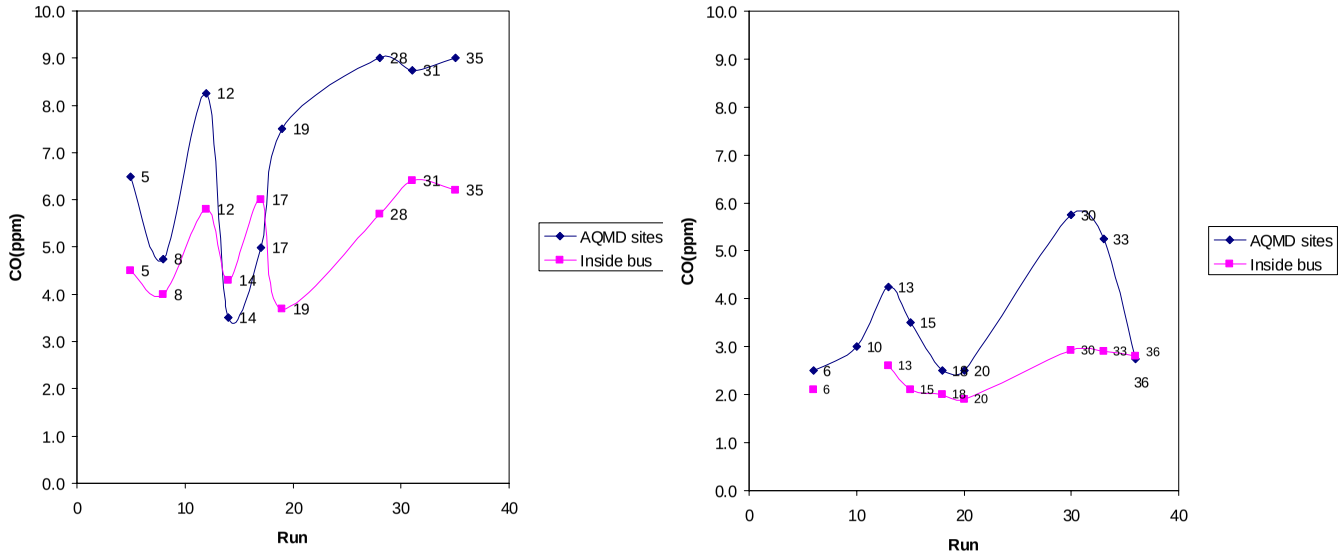


Figure 5.4.1.1 CO concentrations (ppm) inside buses and at the selected AQMD stations.

5.4.1.2 Nitrogen Dioxide

Tables 5.4.1.3 and 5.4.1.4 present mean NO_2 concentrations measured inside buses and at the Central LA and WLA stations for morning and afternoon runs, respectively. Mean NO_2 concentrations inside the buses were consistently higher than those measured at the ambient sites. For morning runs, the overall mean concentration inside the buses was 64 ppb versus 28 ppb at the AQMD stations. On average, NO_2 concentrations inside the buses were more than two times higher than the ambient concentrations in the morning during the main study. The overall mean concentration inside bus in the afternoon was 76 ppb, higher than inside the buses in the morning and more than three times higher than the overall ambient level of about 22 ppb.

Exposure to NO_2 inside buses employed in the main study ranged from 34 to 110 ppb in the morning and from 39 to 115 ppb in the afternoon, significantly higher than in earlier studies of other microenvironments. Levy et al. (1998) reported NO_2 concentrations measured in eighteen cities in fifteen countries around the world. The range of personal exposures was from 11 ppb to 51.5 ppb. The highest NO_2 concentration reported in their study was 106 ppb in workplace exposure in Mexico City. Stevenson et al. (2001) also reported the average NO_2 concentration during 1997 in UK was 23 ppb at curbside locations. They defined points within one to five meters from the curb of a busy road as curbside locations, reflecting NO_2 concentrations on a busy street.

Table 5.4.1.3 Mean NO_2 concentrations (ppb) inside buses and at two nearby AQMD stations for the morning runs with windows closed on urban route one.

Date	Run	Week	Bus	Inside bus	West LA	Central LA	Mean[1]	Ratio[2]
05/01/02	5	2	HE2	49.8	28.5	28.5	28.5	1.7
05/08/02	8	3	HE3	109.9	17	29.5	23.3	4.7
05/14/02	12	4	RE1	55.4	40.5	33.5	37	1.5
05/16/02	14	4	RE1	91.8	14	28.5	21.3	4.3
05/21/02	17	5	RE2	74	19.5	25.5	22.5	3.3
05/22/02	19	5	RE2	78.2	30.5	25	27.8	2.8
06/05/02	28	7	TO1	38.7	25	45	35	1.1
06/06/02	31	7	TO1	45.9	30	35	32.5	1.4
06/12/02	35	8	CNG	33.8	25	30	27.5	1.2
			Mean	64.2			28.4	2.3

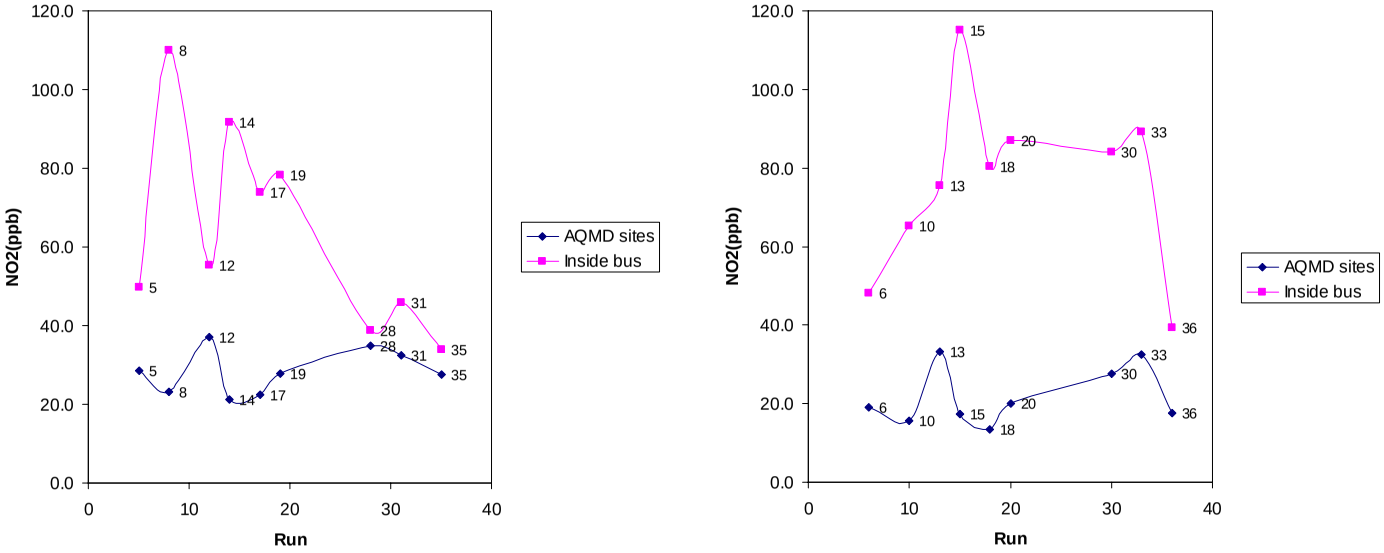
[1]Arithmetic mean of west LA and central LA stations.
[2]Ratio = (mean of inside bus) / (mean of two stations).

Table 5.4.1.4 Mean NO_2 concentrations (ppb) inside buses and at two nearby AQMD stations for the afternoon runs with windows open on urban route one.

Date	Run	Week	Bus	Inside bus	West LA	Central LA	Mean[1]	Ratio[2]
05/01/02	6	2	HE2	48.1	28.5	9.5	19	2.5
05/08/02	10	3	HE3	65.2	17	14.5	15.8	4.1
05/14/02	13	4	RE1	75.6	40.5	26	33.3	2.3
05/16/02	15	4	RE1	115.1	14	20.5	17.3	6.7
05/21/02	18	5	RE2	80.4	19.5	7.5	13.5	6.0
05/22/02	20	5	RE2	86.9	30.5	9.5	20	4.3
06/05/02	30	7	TO1	84	25	30	27.5	3.1
06/06/02	33	7	TO1	89.2	30	35	32.5	2.7
06/12/02	36	8	CNG	39.4	25	10	17.5	2.3
			Mean	76			21.8	3.5

[1]Arithmetic mean of west LA and central LA stations.
[2]Ratio = (mean of inside bus) / (mean of two stations).

Figure 5.4.1.2 shows clear differences in NO_2 concentrations between inside the buses and at the ambient sites. It is noteworthy that the CNG bus had the lowest NO_2 concentrations among bus types in the morning and afternoon runs. However, the trends in concentrations over the study period less closely tracked each other compared with CO (as discussed in the previous section). Furthermore, there was little difference in average NO_2 concentrations between morning and afternoon runs (64 versus 76 ppb inside buses and 28 versus 22 ppb at the ambient sites), while CO concentrations in the morning were about two times higher than those in the afternoon in both microenvironments.



(a) Morning (b) Afternoon

Figure 5.4.1.2 NO_2 concentrations (ppb) inside buses and at the selected AQMD stations.

5.4.2 Integrated Samples

5.4.2.1 Aromatic Hydrocarbons

During the main study we measured the concentrations of several aromatic hydrocarbons both inside the school buses and at the site of the AQMD station located in west Los Angeles (WLA). The mean concentrations of target pollutants are shown for morning runs and afternoon runs in Tables 5.4.2.1 and 5.4.2.2, respectively. Ratios of mean concentrations between inside the bus and the WLA AQMD site varied by pollutant. It is also interesting that the ratios for each pollutant varied between morning and afternoon sampling periods.

Several other studies have also compared concentrations of aromatic hydrocarbons inside vehicles versus fixed monitoring sites. Rodes et al. (1998) measured concentrations of all of the above-mentioned pollutants (except n-octane) inside a typical California diesel school bus in Sacramento and then compared those concentrations to ambient concentrations. The ratios between inside the school bus and the ambient site were available only for m and p-xylene (ratio = 2.5) and toluene (ratio = 2) in the morning and only for toluene (ratio = 1.2) in the afternoon. The higher ratios found for the same pollutants in the present study are presumably due to differences in traffic densities and vehicle mixes between Sacramento and Los Angeles as found by Rodes et al. (1998) and Fruin (2003).

Jo and Park (1999) also reported in-diesel bus/fixed site ratios for aromatic hydrocarbons: benzene (3), ethylbenzene (4.2), m and p-xylene (5), o-xylene (3.8), toluene (2). They conducted measurements of those compounds during morning and evening rush hours in Taegu, the third largest city in Korea. Ratios for benzene, ethylbenzene, and m and p-xylene were comparable to ratios we found in the present study, while ratios for o-xylene and toluene were much lower than our results.

Table 5.4.2.1 Mean concentrations (ppb) of aromatic hydrocarbons during morning runs (windows closed).

Compound	Inside bus	West LA AQMD site	Ratio[1]
Benzene	2.9	0.5	5.8
Ethylbenzene	1.1	0.1	11
m and p-Xylenes	4.4	0.7	6.3
o-Xylene	1.7	0.2	8.5
n-Octane[2]	1.5	1.0	1.5
Toluene	17.7	2.3	7.7

1. Ratio = (mean concentration inside bus) / (mean concentration at WLA AQMD site)
2. n-octane is not an aromatic hydrocarbons but is reported for completeness.
3. Detection limit for all compounds except toluene was 0.01 ppb.

Table 5.4.2.2 Mean concentrations (ppb) of aromatic hydrocarbons during afternoon runs (windows open).

Compound	Inside bus	West LA AQMD	Ratio[1]
Benzene	0.9	0.6	1.5
Ethylbenzene	0.4	0.1	4
m and p-Xylenes	2	0.2	10
o-Xylene	0.6	0.05	12
n-Octane[2]	1.9	0.1	19
Toluene	14.4	0.7	20.6

1. Ratio = (mean concentration inside bus) / (mean concentration at WLA AQMD site)
2. n-octane is not an aromatic hydrocarbons but is reported for completeness.
3. Detection limit for all compounds except toluene was 0.01 ppb.

5.4.2.2 1,3-Butadiene

Concentrations of 1,3-butadiene were also measured inside the school buses and at the west LA AQMD site during the main study. Several other studies have also investigated in-vehicle concentration of 1,3-butadiene (Chan et al., 1991a; Duffy and Nelson 1997; Rodes et al., 1998). However, 1,3-butadiene has high photochemical reactivity and will undergo rapid photooxidation during the day (Carter and Atkinson, 1989). Thus, in the study by Duffy and Nelson (1997), significant concentrations of 1,3-butadiene were only observed inside vehicles during peak traffic hours, while concentrations of 1,3-butadiene in the samples collected during non-peak traffic times or in ambient air were near, or below, the detection limit of 0.1 ppb. Chan et al. (1991a) also reported 30% of in-vehicle samples and 100% of fixed-site and sidewalk samples below the detection limit of 1 ppb. In the present study, 70% of the 1,3-butadiene data collected during the bus commute runs were below the detection limit of 0.5 ppb.

Table 5.4.2.3 shows concentrations of 1,3-butadiene inside the buses and at the west LA AQMD site. In-cabin concentrations of 1,3-butadiene were above the detection limit of 0.5 ppb for only eight morning runs out of thirty-one bus commute runs. For the west LA AQMD site, only two values of 0.5 ppb and 0.6 ppb were at, or above, the detection limit of 0.5 ppb, both in the afternoon.

Table 5.4.2.3 Concentrations of 1,3-butadiene (ppb) during the main study.

Date	Run	Time	Location	Concentration
4/23/02	2	AM	Bus	1.3
5/8/02	8	AM	Bus	0.9
5/14/02	12	AM	Bus	0.8
5/16/02	14	AM	Bus	0.8
5/21/02	17	AM	Bus	0.8
5/22/02	19	AM	Bus	0.6
6/6/02	31	AM	Bus	1.2
6/12/02	35	AM	Bus	1.1
5/21/02		PM	WLA AQMD site	0.5
5/29/02		PM	WLA AQMD site	0.6

Table 5.4.2.4 summarizes in-vehicle mean concentrations of 1,3-butadiene measured in the present study as well as in earlier studies. Two studies compared in-vehicle concentrations to ambient concentrations: the present study (0.9 ppb average inside bus and 0.5 ppb or less average at the west LA AQMD station) and Chan et al. (1.5 ppb in-vehicle and 0.5 ppb at a fixed-site). According to Seiber (1996), the statewide population-weighted exposure to ambient airborne 1,3-butadiene for California was estimated to be 0.37 ppb. Consequently, on average, children on the school buses employed in the present study would have been exposed to 1,3-butadiene at about two and a half times the statewide population-weighted level during the times of their commute.

Table 5.4.2.4 Studies measuring 1,3-butadiene (ppb) inside vehicles.

Reference	Year	Location	Vehicle type	Concentration
This study	2002	Los Angeles, CA	School bus (diesel)	0.9 ± 0.2
Chan et al. (1991)	1988	Raleigh, NC	Passenger car (gasoline)	1.5 ± 1.0
Duffy and Nelson (1997)	1996	Sydney, Australia	Bus (diesel)	2.2
Rodes et al. (1998)	1997	Sacramento, CA	School bus (diesel)	0.4

5.4.2.3 Gaseous Aldehydes and Ketones

Gaseous aldehydes and ketones (i.e., carbonyls) emitted by vehicles have been identified as hazardous air pollutants, with compounds such as formaldehyde being a known human carcinogen (Kean et al. 2001). Grosjean et al. (2001) showed formaldehyde, acetaldehyde, and acetone were the three most abundant carbonyls emitted by vehicles. However, there are few data on children's exposure to speciated carbonyls inside school buses powered by diesel fuel. During the main study the concentrations of fourteen carbonyl species including formaldehyde, acetaldehyde, and acetone, were measured inside the buses. For comparisons, these fourteen carbonyls were also measured concurrently at the west LA AQMD site.

Tables 5.4.2.5 and 5.4.2.6 show the mean concentrations of each individual species for morning runs and afternoon runs, respectively. Acrolein and methacrolein were not reported because all the measurements of both species were below the detection limit of 0.03 ug/sample.

Table 5.4.2.5 Mean concentrations (μg/m^3) of carbonyls during morning runs (windows closed).

	Inside bus cabin	WLA AQMD Site	Ratio[1]
Formaldehyde	2.1	0.4	5.3
Acetaldehyde	28.2	6.7	4.2
Acetone	18.6	9.5	2
Propanal	4.8	1.9	2.5
Crotonaldehyde	2.7	0.7	3.9
Methylethylketone[2]	32.7	3.3	9.9
Butanal[3]	4.6	3	1.5
Benzaldehyde	0.9	0.8	1.1
Pentanal	1.5	1.4	1.1
m-Tolualdehyde	1.4	0.5	2.8
Hexanal	4.4	2.2	2
Pentanones	1.2	N/A	N/A

[1]Ratio = (Inside school buses) / (WLA AQMD site)
[2]methylethylketone (MEK) = 2-butanone
[3]butanal = butyraldehyde
[4]N/A: not available.

Table 5.4.2.6 Mean concentrations (ug/m^3) of carbonyls during afternoon runs (windows open).

	Inside bus cabin	At West LA AQMD Site	Ratio[1]
Formaldehyde	1.1	0.4	2.8
Acetaldehyde	62.5	15.1	4.1
Acetone	20.4	9.6	2.1
Propanal	10.9	3.1	3.5
Crotonaldehyde	2.8	1	2.8
Methylethylketone[2]	76.3	5	15.3
Butanal[3]	10.7	5.1	2.1
Benzaldehyde	0.5	0.6	0.8
Pentanal	3.5	3.1	1.1
m-Tolualdehyde	1.4	1.3	1.1
Hexanal	5.2	4.1	1.3
Pentanones	2.5	3.1	0.8

[1]Ratio = (Inside school buses) / (WLA AQMD site)
[2]methylethylketone (MEK) = 2-butanone
[3]butanal = butyraldehyde

As shown in Table 5.4.2.5, the mean in-cabin concentrations of acetaldehyde, acetone and methylethylketone were substantially higher than for the other compounds. Moreover, the mean concentration of each carbonyl inside the bus cabin divided by the corresponding mean concentration at the WLA AQMD site, varied widely by carbonyl species in the morning runs. Benzaldehyde, pentanal, and butanal showed little difference, while the concentration of methylethylketone inside the bus cabin was on average about ten times higher compared to the west LA AQMD station. Formaldehyde and acetaldehyde also showed much higher concentrations inside the buses than at the WLA AQMD site with ratios of five and four for formaldehyde and acetaldehyde, respectively. The remaining compounds showed average

concentrations inside the bus cabin two to four times higher than in ambient air measured at the WLA AQMD site.

For the afternoon runs, the mean concentrations of methylethylketone (MEK) and acetaldehyde were much higher inside the buses compared with other species. As in the morning runs, the ratio of mean MEK concentrations inside the bus cabin to the mean concentration in ambient air was much higher than for the other compounds and about 50% higher than the morning ratio. For most other compounds there was reasonable consistency between the morning and afternoon in the ratios of the inside cabin/WLA site mean concentrations, given the uncertainties in these measurements. An exception was formaldehyde for which the afternoon inside cabin concentration was half the morning concentration while the mean concentration at the WLA AQMD site was the same in the morning and afternoon.

6.0 CONCLUSIONS AND RECOMMENDATIONS

6.1 Importance of the School Bus Microenvironment in Children's Exposure During School Bus Commute-Related Activities

Across all pollutants, concentrations measured during bus commutes on urban routes were higher than background concentrations measured during precision experiments or at the bus stop (Table 6.1). For example, the highest mean concentrations of black carbon and PAH measured during a bus commute were nearly factors of ten and twenty, respectively, greater than mean concentrations measured for these pollutants during instrument precision assessments made near the West Los Angeles AQMD site. The mean bus commute concentrations over all bus runs (in parentheses) were approximately a factor of two or more (except for NO_2) higher than mean concentrations at the bus stops studied.

Table 6.1. Mean concentrations measured in two microenvironments and in background air.

	Ambient air[1]	Bus stop	Bus commutes[2,3]
Black Carbon (ug/m^3)	2 ± 0.1	4 ± 0.4	3 - 19 (8)
PAH (ng/m^3)	27 ± 1.5	44 ± 4.5	64 - 400 (134)
NO_2 (ppb)	49 ± 1.0	54 ± 1.9	34 - 110 (73)
PC ($\#/cm^3$)[5]	83 ± 3.1	62 ± 1.8	70 - 236 (130)
$PM_{2.5}$ (ug/m^3)	23[4]	26	21 - 62 (43)

[1]Measured during precision experiments near the West Los Angeles AQMD station.
[2]Ranges include all bus types (see Table 5.3.3.1).
[3]Values within parentheses are the mean for all bus commutes.
[4]2001 annual arithmetic mean from the Central Los Angeles AQMD station.
[5]In 0.3 – 0.5um size range.

Exposure factors, calculated by multiplying the mean concentration in a given microenvironment by the average time spent in that microenvironment, were highest for the bus commute (Table 6.2). Bus stop microenvironment exposures were up to thirteen times greater than loading/unloading zone microenvironment exposures, while the urban bus commute exposures were between 50 and 400 times greater than those due to the loading/unloading microenvironment. The urban bus commutes led to between 20 and 100 times higher exposures than at the bus stops, depending upon the pollutant. Therefore, the bus commute microenvironment dominated school bus commute-related exposures, and the bus stop and loading/unloading zone microenvironments were nearly insignificant contributors to commute-related exposures.

Table 6.2. Exposure factors in three school bus commute-related microenvironments.

	Exposure Factors		
	Loading/ unloading[1]	Bus stops	Urban commutes
BC ($ug/m^3 * min$)	5	20	600
PAH ($ng/m^3 * min$)	45	230	10000
NO_2 ($ppb * min$)	105	270	5500
PC ($\#/cm^3 * min$)	25	310	10000
$PM_{2.5}$ ($ug/m^3 * min$)	N/A	130	3500

[1]Exposure factor calculations for the loading/unloading zone were based on measurements taken during the pilot study under winter conditions.

6.2 Contribution of Bus's Own Exhaust to Concentrations Inside the Cabin

Our study used an SF_6 tracer gas metered into the bus's exhaust system to determine the percentage of air in the bus cabin originating from the bus's own exhaust and this percentage varied significantly between buses and also depended on window position. For all buses tested, the amount of the bus's own exhaust inside the cabin was substantially higher with all windows closed compared with the windows partially opened. Moreover, older buses showed a larger percentage of their own exhaust entering into the cabin compared with newer buses.

SF_6 concentrations inside the bus cabin correlated reasonably well with normalized black carbon concentrations measured inside the conventional diesel and trap-outfitted diesel bus cabins. This correlation was stronger than that between SF_6 and $PM_{2.5}$ mass when the windows were closed.

Different bus models and ages correspond to different construction and cabin designs, and the results of our SF_6 analyses suggested these differences may result in a wide range of pollutant exposures across bus types, depending on the pollutant. In general, the buses that exhibited high opacity during snap-and-idle tests (i.e., conventional diesel buses) and a high percentage of SF_6 inside the cabin, also exhibited the highest average inside/rear concentrations of black carbon, PAH and NO_2 in runs with the windows closed. Although the higher inside/rear exposures of these pollutants could be predicted by the snap-and-idle tests and SF_6 data, this was not the case for particle counts (in the size range from 0.3 to 0.5 μm) or $PM_{2.5}$ mass average concentrations inside the cabin.

6.3. Conditions Associated with Highest Concentrations Inside the Bus Cabin

6.3.1 Within-Bus Variables

Window Position. The influence of window position depended on whether pollutants were primarily directly-emitted (e.g., black carbon and PAH) or had strong "background" contributions (e.g., $PM_{2.5}$). Concentrations of black carbon inside the cabin tended to decrease when the windows were partially opened but increase when the windows were closed. In contrast, for particle counts within the size range 0.3 to 0.5 μm, concentrations inside the cabin tended to increase when the windows were partially opened but tended to decrease when the windows were closed. Thus, for directly-emitted pollutants it appears increased ventilation rates with the windows opened caused dilution of the "self pollution," while for "background" pollutants closed windows inhibited entrance of these pollutants into the cabin.

Location Inside the Bus Cabin. With the windows closed the concentrations of PAH and particle counts (in the range of 0.3 – 0.5 μm) at the rear of the cabin were statistically significantly higher than those at the front of the cabin across all conventional diesel buses tested. With partially opened windows, there were no statistically significant differences between measurements at the front and rear of the cabin. These results are consistent with the preceding analysis suggesting that when the windows were open, higher ventilation rates promoted uniform mixing within the cabin.

6.3.2 External Variables

Type of Vehicle in Front of, or Adjacent to, the Test Bus. Using the videotapes of the bus runs with windows open, we found black carbon and PAH concentrations inside the test buses were highest when following a diesel school bus which emitted visible exhaust, and lowest when following a gasoline vehicle or no target. When following a smoky diesel school bus, concentrations inside the cabin were on average eight and twelve times higher for black carbon and PAH, respectively, compared with following a gasoline vehicle or no target. When following a diesel school bus that was not emitting visible exhaust, black carbon and PAH concentrations inside the test buses were on average four and six times higher, respectively, compared with following a gasoline vehicle or no target. These results were found during commutes when the windows of the bus were open.

This finding is important because the type of diesel vehicle we encountered most frequently during all bus runs was a diesel school bus. On average in the afternoon runs analyzed using the videotapes, another diesel vehicle was within three car lengths in front of or adjacent to, our bus, during more than one-quarter the commute, with diesel school buses responsible for over sixty percent of these encounters. This high incidence of following other diesel school buses was in part due to caravanning behind other buses leaving the BSMS.

These results demonstrated that the type of school bus a child rides on is not the only determinant of exposure and that conventional diesel school buses can have a double exposure impact on commuting children: first, the influence of the bus's own exhaust on concentrations inside the cabin and second, exposures to the exhaust from other nearby conventional diesel school buses. These encounters may cause high concentrations of black carbon and PAH inside the cabin of the school bus, particularly if the bus windows are open.

Roadway Type The degree of influence of roadway type on pollutant concentrations inside the bus cabin depended on the type of bus, the level of traffic congestion and whether the bus windows were open or closed. For relatively clean buses (i.e., the CNG bus, and the trap-outfitted diesel bus), traveling on congested freeways with higher pollutant concentrations tended to increase concentrations inside the bus. For relatively dirtier buses (i.e., conventional diesel), because the bus's own exhaust was a significant contributor to concentrations inside the bus, increasing the ventilation rate inside the bus, which was more likely to occur on freeways when the windows were open, tended to decrease concentrations inside the bus.

Route Type Two different urban routes, U1 and U2, (as well as an rural/suburban route) were driven by the same "representative" conventional diesel bus on two consecutive weeks and for the comparison of urban routes U1 and U2, only these runs were used. We found the concentrations of black carbon and PAH averaged over the entire runs were not significantly different between the two different urban routes while the rural/suburban route exhibited lower concentrations of both black carbon and PAH. This was interesting since U1 spent approximately 40% of the time on the freeway and 60% of the time on surface streets, while U2 was entirely on surface streets. Features which were similar between routes, such as encounters with other diesel vehicles, particularly diesel school buses, dominated the highest peak concentrations of black carbon and PAH on both routes.

For PC (in the size range from 0.3 to 0.5 um) and $PM_{2.5}$ mass, the rural/suburban route and U1 exhibited similar average within-bus concentrations, while U2 showed higher average concentrations than U1 during both morning and afternoon commutes with windows closed and open, respectively. These results demonstrate, again, the differences between the influences of directly emitted versus "background" pollutants on concentrations inside the bus cabin.

Table 6.3 presents the mean pollutant concentrations by run type for the urban and rural/suburban routes, under open and closed window conditions. Both window position and surrounding traffic density can be seen to have strongly affected on-board concentrations for vehicle-related pollutants, but for pollutants with a strong background component, such as PM2.5 and fine particle counts (0.3-0.5 μm), differences between routes and differences due to window position were less distinct.

Table 6.3. Concentrations means and range of means by route and by window position[1].

Pollutant	Urban Route[1]				Rural/Suburban Route	
	Windows Closed (morning)[2]		Windows Open (afternoon)[2]		Windows Open (afternoon)[2]	
	Mean	Range of Means	Mean	Range of Means	Mean	Range of Means
Black Carbon ($\mu g/m^3$)	10	2.5 – 19	5.2	2.9 – 9.1	2.7	0.9 – 4.8
PM-bound PAH (ng/m^3)	198	64 – 400	96	33 – 147	36	14 – 66
NO_2 (ppb)	64	34 – 110	73	39 – 120	45	23 – 68
Formaldehyde ($\mu g/m^3$)	2.1	0.89 – 4.8	1.1	0.55 – 2.1	0.93	0.34 – 2.0
Particle Counts ($\#/cm^3$)	113	51 – 235	96	19 – 276	159	29 – 253
Carbon Monoxide (ppm)	5.1	3.7 – 6.4	2.4	1.9 – 2.9	na[3]	na[3]

[1]Includes commutes on urban route one only.
[2]Windows closed in morning and open in afternoon, similar to conditions uniformly observed in other buses in Los Angeles for April through June.
[3]No measurements conducted.

6.4 Variations in Exposure Across Fuel Type and After-Treatment Technology

6.4.1 Windows Closed

For directly-emitted pollutants such as black carbon and PAH, with the windows closed we found the lowest mean concentrations inside the CNG bus, and the highest concentrations inside the conventional diesel buses. Compared with the CNG bus, concentrations of black carbon and PAH were two-to-three times higher inside the trap-outfitted diesel bus, and two-to-six times higher inside the conventional diesel buses. We also found the conventional diesel buses had the highest average NO_2 concentrations. Thus, in our study, the highest exposures to black carbon, PAH and NO_2 occurred while riding on a conventional diesel school bus, followed next by the trap-outfitted diesel bus, with the lowest exposures to these pollutants occurring on the CNG bus.

For pollutants such as particle counts (in the size range from 0.3 to 0.5 μm) and $PM_{2.5}$ mass, differences observed between bus types were not easily explained, although these pollutants tend to have a higher "background" influence.

Finally, the type of bus we tested did not appear to be an important variable in determining in-cabin concentrations of CO, with comparable, low in-bus concentrations measured in conventional diesel, trap-outfitted diesel, and CNG buses.

6.4.2 Windows Partially Open

During afternoon runs, the windows in our tests buses were partially open, consistent with the behavior of a majority of school buses we observed in the afternoon in Los Angeles. As discussed in Section 5.3.5 and 6.3.2, we found the dominant variable associated with high concentrations of black carbon, PAH and NO_2 inside the bus cabin when the windows of the bus were open was the presence of another diesel vehicle, while the contribution from the bus's own exhaust was reduced under these conditions.

During periods of idling, with windows partially open and with no other diesel vehicles present, the CNG bus again had the lowest mean concentrations of black carbon and PAH, while the conventional diesel buses had concentrations that were two-to-four times higher than either the CNG or trap-outfitted diesel buses. For NO_2, the trap-outfitted and conventional diesel buses both had mean concentrations nearly double those measured on the CNG bus. No real time data were available for formaldehyde, so we were not able to examine subsets of the data.

For all buses tested, we found the highest peak concentrations were observed when the test bus was idling behind another diesel vehicle. The trap-outfitted diesel bus and CNG bus generally exhibited high peak concentrations only while traveling behind a diesel vehicle, while the conventional diesel buses exhibited high peaks both when following other diesel vehicles and while idling without another diesel vehicle in front of the bus.

6.4.3 Summary of Effect of Bus Type

In summary, during commutes when the windows of the bus were closed, we found substantial differences in concentrations measured inside the bus cabin depending on the fuel type or after-treatment technology of the test bus, while the impact of outside sources was less important. In contrast, when the windows of the bus were open, differences between buses were dominated by the influence of outside sources (e.g. other diesel vehicles in front of the test bus). However, for commutes with windows open, investigation of concentrations inside the bus during periods of idling on small residential streets, when no other diesel vehicles were present, showed significant differences between buses based on bus type. Overall, however, the magnitudes of the differences were markedly reduced compared with windows closed.

Table 6.4 provides a summary of the mean concentrations measured inside the buses during commutes on urban route one. For commutes with windows open, the first number represents the mean concentration from the entire commute, while the number in parentheses represents the mean concentration during periods of the commute when the buses were idling in residential areas, and no other diesel vehicles were present (See Section 5.3.5). Both numbers were included to demonstrate the importance of outside sources when the windows of the bus were open. When the windows of the bus were closed, outside sources such as other diesel vehicles, did not have a discernable influence on concentrations inside the test bus.

Table 6.4 Mean concentrations and 95% confidence intervals during commutes by bus type[1,2].

	Windows Closed (morning)			Windows Open (afternoon)[3,4]		
	CNG Bus	Trap-Outfitted Diesel Bus	Conventional Diesel Buses	CNG Bus	Trap-Outfitted Diesel Bus	Conventional Diesel Buses
Black Carbon (ug/m^3)	2.5 ± 0.1	7.2 ± 0.2	12.5 ± 0.3	3.5 ± 0.4 (1.2 ± 0.2)	4.2 ± 0.3 (1.9 ± 0.2)	5.9 ± 0.3 (8.6 ± 1.9)
PAH (ng/m^3)[5]	64 ± 3	201 ± 7	219 ± 5	119 ± 18 (27 ± 16)	105 ± 12 (20 ± 4)	87 ± 7 (106 ± 26)
NO_2 (ppb)	34 ± 0.4	42 ± 0.4	76 ± 1	39 ± 1 (37 ± 3)	86 ± 2 (78 ± 6)	77 ± 1 (64 ± 6)
Formaldehyde[6] (ug/m^3)	4.8	2.2	1.4	1.2	1.0	1.0

[1]Includes bus commutes on urban route one only.
[2]The number of buses tested in each category included: one CNG bus; one particle trap-outfitted diesel bus; five conventional diesel buses.
[3]With windows open, concentrations inside bus were dominated by outside sources (see Section 5.3.5 and 6.4.2), thus reducing the influence of the bus type.
[4]Numbers in parenthesis represent mean concentrations during periods of the commute when the bus was idling in residential neighborhoods with windows open, and no other diesel vehicles were present. Includes selected afternoon runs on HE2, RE1, RE2, TO1 and CNG (see Section 5.3.5.1).
[5]Measurements of PAH inside conventional diesel buses may be biased low due to the instrument's maximum setting during these commutes. See Sections 4.1.2.3.5 and 5.3.3 for specific details.
[6]Confidence intervals are not presented for formaldehyde because of small sample size.

Across all of our various measurements, the highest concentrations of black carbon and PAH were consistently found inside the conventional diesel test buses compared to the trap-outfitted diesel or CNG buses, with the CNG bus having the lowest concentrations of these pollutants. The conventional diesel and trap-outfitted buses also had the highest concentrations of NO_2. For formaldehyde the CNG bus exhibited the highest integrated concentration of any bus type, twice the average concentration for the trap-outfitted bus and three times the average for the conventional diesel buses when the windows of the bus were closed. Results for the trap-outfitted bus were generally in between the conventional diesel buses and the CNG bus. However, based on emission data reported for other trap-equipped diesel vehicles (Johnson, 2001); diesel-related pollutant concentrations on board our specific trap-outfitted bus appeared to be higher than expected.

In addition, it is essential to consider that bus-to-bus variations were large, so to fully characterize the differences in exposures due to bus fuel type would require a much larger number of buses.

6.5 Comparison of Exposure to Air Pollutants During Bus Commutes with Ambient Air Concentrations Measured at Nearby AQMD Monitoring Sites

In the course of the main study, we compared measurements taken inside the bus cabin for runs on U1 with measurements taken simultaneously at the SCAQMD ambient air monitoring sites in downtown and west Los Angeles (WLA). Different results were obtained for different pollutants as shown in Table 6.5. On average, ambient concentrations of CO measured at the nearby monitoring stations were slightly higher than those measured inside the bus cabin, but within-cabin CO concentrations were relatively low during all runs (2-3 ppm in the afternoon and 4-6 ppm in the morning). Commutes on diesel school buses do not appear to result in higher exposures to CO than would be encountered in nearby ambient air.

In contrast, NO_2 concentrations inside the buses were, on average, more than two and three times higher than the ambient concentrations measured at the SCAQMD sites in the morning and afternoon, respectively. Concentrations of aromatic hydrocarbons, including benzene, were four to ten times higher inside the buses we tested than at the WLA ambient monitoring site during morning commutes. Other aromatic hydrocarbons were three to twelve times higher inside the buses and within-cabin concentration of 1,3-butadiene (only found above detection limits for about 30% of the bus commutes) were higher than those measured at the WLA ambient site.

During morning runs, concentrations of aldehydes and ketones measured inside the bus cabins were generally higher than those measured at the WLA ambient monitoring site. Formaldehyde, acetaldehyde and methylethylketone were five, four and nine times, respectively, higher inside the buses compared with ambient concentrations. Similar ratios were obtained during the afternoon (except for formaldehyde).

These results indicate children riding on school buses may be exposed to elevated concentrations of NO_2, aromatic hydrocarbons, 1,3-butadiene, and a variety of aldehydes and ketones (including formaldehyde) and that estimates of children's exposure to these pollutants based on ambient air concentrations alone are unlikely to account for the elevated exposures occurring on bus commutes. Table 6.5 summarizes the results for the comparison of in-bus and ambient air concentrations during morning commutes on urban route one.

Table 6.5 Morning mean concentrations: buses versus ambient air at West Los Angeles monitoring site.

	Bus Cabin Concentration	West Los Angeles Monitoring Site Concentration	Bus-to-Ambient Ratio
CO (ppm)	5.1 ± 0.1	6.9	0.7
NO_2 (ppb)	64 ± 1	28	2.3
Benzene (ppb)	2.9	0.5	5.8
Formaldehyde (μg/m3)	2.1	0.4	5.4

6.6 Recommendations

Although the present study was ambitious in its scope and approach, it was necessarily limited to a single school, a relatively small number of buses, and a given region of California (i.e., the SoCAB), as well as a limited number of bus routes and bus stops. However, based on the results of this study it is apparent several steps can be taken to reduce exposures of the public in general and commuting children in particular due to conventional diesel school buses. First, reducing the number of conventional diesel school buses, by replacing them with cleaner buses, such as CNG-fueled or trap-outfitted diesel buses, will substantially reduce exposures of children, commuters and the general public to the directly-emitted pollutants measured in this study, such as black carbon, PAHs and NO_2. Moreover, commuting children would be doubly benefited by phasing out current diesel engines since both pollutants from the bus's own exhaust and from nearby buses would be minimized. However, as long as a significant number of conventional diesel school buses remain in service, the cleanest buses should be assigned to the longest routes.

Next, school bus drivers should be instructed to avoid other diesel school buses, and especially to avoid caravanning. This would provide an immediate reduction of children's exposures to diesel exhaust. In particular, at schools which have more than one bus transporting children to and from school, instituting a policy of staggered departure times of buses at that school would significantly reduce children's exposures to diesel-related pollutants. With such a policy, bus drivers would be required to wait a minute or two after the bus ahead of them has departed before they pull away from the school.

If conventional diesel buses are not full, children should be encouraged to sit nearer the front of the bus than the rear. In addition, if strategies are available to shorten commute times for children such strategies should be pursued, since on average exposure is proportional to commute time.

Another major recommendation is that in-use diesel school bus engines be maintained properly by school districts to eliminate visible exhaust, since the presence of diesel school buses with visible exhaust resulted in the highest peak concentrations observed inside the test buses during this study. Moreover, in the course of this study we estimated that between one fourth to one third of all diesel school buses we encountered emitted black smoke or visible exhaust. Thus, we believe the prevalence of "dirty" buses in Southern California is high.

Finally, we want to reinforce school district policies which minimize children's time spent on sidewalks in front of schools when diesel school buses are arriving or departing and urge enforcement of the policy that drivers immediately turn off their engines on arrival at a school and do not turn their engines on until ready to depart a school. In the course of the present study we received anecdotal evidence that such policies are not always strictly enforced.

7.0 RECOMMENDATIONS FOR FUTURE RESEARCH

Further research could be undertaken to interpret the measurements from the present study. For example, the laborious analysis of the data obtained with the SEMS instruments at the smallest particle size ranges, from 30 to 100 nm, was beyond the scope of the present project. However, such data would provide important information about roadway and in-bus exposure to ultra-fine particles in this size range, and this analysis should be undertaken as soon as possible.

More extensive comparisons could be made between the data obtained in the present study and recent data obtained, or being obtained, for near-roadway and in-roadway measurements of black carbon and other vehicle-related species. For example, some of the current measurements by the Southern California Particle Center and Supersite (SCPCS) researchers as a function of distance from freeways would be of interest relative to certain of our in-roadway and within bus-cabin measurements. Additional comparisons could also be made between the data generated in the current project for gaseous and particulate pollutants and corresponding ambient air measurements made away from heavily traveled arterials for the same pollutants.

A follow-on school bus field study modeled on the present project but using on-board particle concentrators and a focused set of pollutant analyzers should be conducted for the express purpose of collecting particulate and gas phase samples for toxicological analyses. Such a study could be built around the expertise of the SCPCS, including the Center's development and application of a range of toxicological assays.

Finally, as we have acknowledged elsewhere in this report, due to resource constraints the present study was unable to test across a full range of possible commutes, meteorological conditions, bus manufacturers, model years, school districts, geographic locations in California, etc. Our results are "representative" only to the extent the commutes, buses, conditions, areas, etc. we studied are representative of southern California school districts. Additional field studies could be conducted to broaden the range of conditions investigated, although substantial resources are required to conduct studies of this magnitude. Potential cost savings include eliminating the PM filter samples (and elemental analysis) and the integrated VOC samples, as they were frequently at low concentrations and provided relatively little information compared to the other study measurements.

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9.0 INVENTIONS REPORTED AND COPYRIGHTED MATERIALS PRODUCED

None.

10.0 GLOSSARY OF TERMS, ABBREVIATIONS AND SYMBOLS

AQMD	South Coast Air Quality Management District
ARB	California Air Resources Board
ATCM	airborne toxic control measure
BC	black carbon
BSMS	Brentwood Science Magnet School
CE-CERT	College of Engineering – Center of Environmental Research and Technology
CNG	compressed natural gas
CNC	condensation nucleus counter
CO	carbon monoxide
DPM	diesel particulate matter
EC	elemental carbon
EPA	Environmental Protection Agency (U.S.)
FRM	Federal reference method
GC	gas chromatography
HE1	high emitter conventional diesel bus number one
HE2	high emitter conventional diesel bus number two
HE3	high emitter conventional diesel bus number three
ID`	inner diameter
LAUSD	Los Angeles Unified School District
NO_2	nitrogen dioxide
NO_x	oxides of nitrogen ($NO + NO_2$)
NRDC	Natural Resources Defences Council
OD	outer diameter
OPC	optical particle counter
PAH	polycyclic aromatic hydrocarbons
PAN	peroxyacetyl nitrate
PC	fine particle counts (0.3 – 0.5 μm diameter)
PM	particulate matter

$PM_{2.5}$	particulate matter less than 2.5 μm in diameter
PM_{10}	particulate matter less than 10 μm in diameter
RE1	representative conventional diesel bus number one
RE2	representative conventional diesel bus number two
SEMS	scanning electrical mobility particle sizer
SF_6	sulfur hexafluoride
SVOC	semi-volatile organic compound
TAC	toxic air contaminant
TO1	particle trap-outfitted diesel bus
U1	urban bus route one
U2	urban bus route two
UCLA	University of California, Los Angeles
UCR	University of California, Riverside
VOC	volatile organic compound