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Characterizing Ultrafine Particles and Other Air Pollutants In and Around School Buses

Yifang Zhu and Qunfang Zhang

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with a Critique by the HEI Health Review Committee

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ABOUT HEI

The Health Effects Institute is a nonprofit corporation chartered in 1980 as an independent research organization to provide high-quality, impartial, and relevant science on the effects of air pollution on health. To accomplish its mission, the institute

- Identifies the highest-priority areas for health effects research;
- Competitively funds and oversees research projects;
- Provides intensive independent review of HEI-supported studies and related research;
- Integrates HEI's research results with those of other institutions into broader evaluations; and
- Communicates the results of HEI's research and analyses to public and private decision makers.

HEI typically receives half of its core funds from the U.S. Environmental Protection Agency and half from the worldwide motor vehicle industry. Frequently, other public and private organizations in the United States and around the world also support major projects or research programs. HEI has funded more than 330 research projects in North America, Europe, Asia, and Latin America, the results of which have informed decisions regarding carbon monoxide, air toxics, nitrogen oxides, diesel exhaust, ozone, particulate matter, and other pollutants. These results have appeared in more than 260 comprehensive reports published by HEI, as well as in more than 1000 articles in the peer-reviewed literature.

HEI's independent Board of Directors consists of leaders in science and policy who are committed to fostering the public–private partnership that is central to the organization. The Health Research Committee solicits input from HEI sponsors and other stakeholders and works with scientific staff to develop a Five-Year Strategic Plan, select research projects for funding, and oversee their conduct. The Health Review Committee, which has no role in selecting or overseeing studies, works with staff to evaluate and interpret the results of funded studies and related research.

All project results and accompanying comments by the Health Review Committee are widely disseminated through HEI's Web site (*www.healtheffects.org*), printed reports, newsletters and other publications, annual conferences, and presentations to legislative bodies and public agencies.

ABOUT THIS REPORT

Research Report 180, *Characterizing Ultrafine Particles and Other Air Pollutants In and Around School Buses,* presents a research project funded by the Health Effects Institute and conducted by Drs. Yifang Zhu and Qunfang Zhang of the Department of Environmental Health Sciences, Jonathan and Karin Fielding School of Public Health, University of California–Los Angeles. This research was funded under HEI's Walter A. Rosenblith New Investigator Award Program, which provides support to promising scientists in the early stages of their careers. This report contains three main sections.

The HEI Statement, prepared by staff at HEI, is a brief, nontechnical summary of the study and its findings; it also briefly describes the Health Review Committee's comments on the study.

The Investigators' Report, prepared by Zhu and Zhang, describes the scientific background, aims, methods, results, and conclusions of the study.

The Critique is prepared by members of the Health Review Committee with the assistance of HEI staff; it places the study in a broader scientific context, points out its strengths and limitations, and discusses remaining uncertainties and implications of the study's findings for public health and future research.

This report has gone through HEI's rigorous review process. When an HEI-funded study is completed, the investigators submit a draft final report presenting the background and results of the study. This draft report is first examined by outside technical reviewers and a biostatistician. The report and the reviewers' comments are then evaluated by members of the Health Review Committee, an independent panel of distinguished scientists who have no involvement in selecting or overseeing HEI studies. During the review process, the investigators have an opportunity to exchange comments with the Review Committee and, as necessary, to revise their report. The Critique reflects the information provided in the final version of the report.

HEI STATEMENT

Synopsis of Research Report 180

Ultrafine Particles and Other Air Pollutants In and Around School Buses

BACKGROUND

Children are considered particularly susceptible to the effects of outdoor particulate matter (PM). Some evidence suggests that the smallest particles in the complex mixture of PM — ultrafine particles, defined by having a diameter of 0.1 µm or less — have properties that may make them particularly toxic. Accurately assessing exposures to ultrafine particles, of which a major source is vehicle emissions, is considered a key research need. Dr. Yifang Zhu, who was a recipient of HEI's Walter A. Rosenblith New Investigator Award, and her colleague Dr. Qunfang Zhang assessed levels of ultrafine particles and other pollutants in and around school buses powered by diesel engines and identified factors contributing to these levels, including an evaluation of two retrofit devices.

APPROACH

Zhu and Zhang measured levels of ultrafine particles and other pollutants, such as $PM_{2.5}$ (PM with an aerodynamic diameter $\leq$ 2.5 µm) and black carbon, in and around school buses in four sets of tests: (1) on-road; (2) during idling; (3) before and after retrofitting with a diesel oxidation catalyst, a crankcase filter system, or both; and (4) before and after operating a high efficiency particulate air (HEPA) filter air purifier inside the cabin. Each set of tests comprised several conditions: for example, windows open or closed, air conditioner on or off, and buses idling in different positions relative to each other. Air pollutants were measured simultaneously inside the cabin as well as directly outside the cabin or close to the tailpipe, except for the on-road tests in which only the in-cabin levels were measured. Measurements were made in small sets of buses (model years 1990–2006) in Texas and California.

MAIN RESULTS AND INTERPRETATION

In its independent review of the study, the HEI Health Review Committee considered that the retrofit and idling tests in particular provided useful information. Tailpipe concentrations of ultrafine particles were significantly reduced (by 20% to 94%) in idling school buses after retrofitting with a

What This Study Adds

- Zhu and Zhang's study adds to the small number of studies assessing air pollutants including ultrafine particles in and around school buses. In a small sample of U.S. diesel-powered school buses evaluated under different conditions, the investigators measured levels of ultrafine particles and other air pollutants inside the bus, directly outside the bus, or close to the tailpipe.
- Retrofitting buses with a diesel oxidation catalyst, a crankcase filtration system, or both substantially reduced tailpipe concentrations of ultrafine particles, black carbon, and fine particulate matter during idling. However, retrofitting did not reduce in-cabin levels of the measured pollutants, indicating that factors other than the vehicle's self-pollution — in particular, ambient levels including emissions from nearby vehicles — were more important in influencing in-cabin concentrations.
- Assessment of in-vehicle pollutant levels remains an important area of study. This study demonstrates the importance of including measurements of ambient air pollution concentrations in future in-vehicle studies.

This Statement, prepared by the Health Effects Institute, summarizes a research project funded by HEI and conducted by Drs. Yifang Zhu and Qunfang Zhang at the University of California–Los Angeles. Research Report 180 contains both the detailed Investigators' Report and a Critique of the study prepared by the Institute's Health Review Committee.

diesel oxidation catalyst and/or a crankcase filter system. Black carbon and $PM_{2.5}$ were also reduced close to the tailpipe, on average by 64% and 47%, respectively, but only the reduction in black carbon was statistically significant. However, while the buses were idling or driving, in-cabin concentrations of the measured pollutants were not reduced after the buses were retrofitted. Similar results were found in the idling test: close-to-tailpipe ultrafine particle concentrations were greatly influenced by the bus' own engine (they were 7- to 26-fold greater than with the engine off), whereas in-cabin concentrations were affected only when the wind blew from the back to the front of the bus, especially with nearby buses idling together (producing up to a 5.8-fold increase). Thus, factors other than the vehicle's self-pollution — in particular, ambient levels including emissions from nearby vehicles — were more important in influencing in-cabin concentrations. The use of a HEPA air purifier and the air conditioner also substantially decreased in-cabin levels of ultrafine particles and $PM_{2.5}$.

A strength of the study was that in many of the tests (idling, retrofit, and HEPA air purifier tests) air pollutants were measured simultaneously inside as well as directly outside the cabin and with identical equipment. In addition, the same buses were tested before and after retrofit devices were installed. However, because of certain design decisions, the Committee concluded that some study results were open to interpretation and thus some conclusions should be considered cautiously. One major reason was that, in the on-road tests, no measurements of ambient concentrations were conducted, and therefore this particular test had limited value overall. In addition, the Committee expressed concern about the absence of an adjustment for varying ambient levels in the retrofit analyses, because it prevented reliable conclusions from being drawn about the effectiveness of the two retrofit devices in reducing in-cabin levels of ultrafine particles and $PM_{2.5}$.

For future studies, the Committee would make the following additional recommendations: (1) collect detailed data on other pollutants — in particular nitrogen dioxide, black carbon, and carbon monoxide — as well as data on traffic intensities; (2) use a standard dilution system when measuring tailpipe concentrations; (3) use a more common alignment of buses (that is, not perpendicular), and account for local fluctuations in wind direction or the possible effects of wind speed when studying idling buses; (4) within available resources, test a larger number of buses, with engines of varying ages, and in different seasons.

CONCLUSIONS

The HEI Health Review Committee concluded that the study by Zhu and Zhang adds to the small number of studies assessing air pollutants including ultrafine particles in and around U.S. school buses. Since a substantial fraction of at least some children's daily exposure may come from bus transfer locations or waiting areas where multiple buses are idling, the reduction in tailpipe concentrations after retrofitting could reduce children's overall exposure to air pollutants and contribute to overall cleaner outdoor air. Further reductions in children's exposure could also be achieved by reducing idling time, increasing the distances between buses during driving and idling, increasing the distances between buses and other vehicles, and avoiding high-traffic roads.

This study holds important methodologic lessons for future in-vehicle studies, since it highlights the importance of including measurements of ambient air pollution concentrations. In-vehicle studies remain an important area of future research because in-vehicle exposure may contribute substantially to a person's average exposure to pollutants such as ultrafine particles, in spite of the fact that time spent in vehicles makes up only a relatively small amount of a person's day. Additional studies are needed to estimate the relative contributions of in-vehicle microenvironments to air pollutant exposures, but also the assessment of the contributions of other microenvironments in which children and adults spend most of their time (such as home, work, and school) is particularly important.

INVESTIGATORS' REPORT

Characterizing Ultrafine Particles and Other Air Pollutants In and Around School Buses

Yifang Zhu and Qunfang Zhang

Department of Environmental Health Sciences, Jonathan and Karin Fielding School of Public Health, University of California–Los Angeles

ABSTRACT

Increasing evidence has demonstrated toxic effects of ultrafine particles (UFP*, diameter < 100 nm). Children are particularly at risk because of their immature respiratory systems and higher breathing rates per body mass. This study aimed to characterize UFP, $PM_{2.5}$ (particulate matter ≤ 2.5 µm in aerodynamic diameter), and other vehicular-emitted pollutants in and around school buses. Four sub-studies were conducted, including:

1. On-road tests to measure in-cabin air pollutant levels while school buses were being driven;
2. Idling tests to determine the contributions of tailpipe emissions from idling school buses to air pollutant levels in and around school buses under different scenarios;
3. Retrofit tests to evaluate the performance of two retrofit systems, a diesel oxidation catalyst (DOC) muffler and a crankcase filtration system (CFS), on reducing tailpipe emissions and in-cabin air pollutant concentrations under idling and driving conditions; and
4. High efficiency particulate air (HEPA) filter air purifier tests to evaluate the effectiveness of in-cabin filtration.

In total, 24 school buses were employed to cover a wide range of school buses commonly used in the United States. Real-time air quality measurements included particle number concentration (PNC), fine and UFP size distribution in the size range 7.6–289 nm, $PM_{2.5}$ mass concentration, black carbon (BC) concentration, and carbon monoxide (CO) and carbon dioxide (CO_2) concentrations. For in-cabin measurements, instruments were placed on a platform secured to the rear seats inside the school buses. For all other tests, a second set of instruments was deployed to simultaneously measure the ambient air pollutant levels. For tailpipe emission measurements, the exhaust was diluted and then measured by instruments identical to those used for the in-cabin measurements.

The results show that when driving on roads, in-cabin PNC, fine and UFP size distribution, $PM_{2.5}$, BC, and CO varied by engine age, window position, driving speed, driving route, and operating conditions. Emissions from idling school buses increased the PNC close to the tailpipe by a factor of up to 26.0. Under some circumstances, tailpipe emissions of idling school buses increased the in-cabin PNC by factors ranging from 1.2 to 5.8 in the 10–30 nm particle size range. Retrofit systems significantly reduced the tailpipe emissions of idling school buses. With both DOC and CFS installed, PNC in tailpipe emissions dropped by 20%–94%. No unequivocal decrease was observed for in-cabin air pollutants after retrofitting. The operation of the air conditioning (AC) unit and the pollutant concentrations in the surrounding ambient air played more important roles than retrofit technologies in determining in-cabin air quality. The use of a HEPA air purifier removed up to 50% of in-cabin particles. Because each sub-study tested only a subset of the 24 school buses, the results should be seen as more exploratory than definitive.

This Investigators' Report is one part of Health Effects Institute Research Report 180, which also includes a Critique by the Health Review Committee and an HEI Statement about the research project. Correspondence concerning the Investigators' Report may be addressed to Dr. Yifang Zhu, Department of Environmental Health Sciences, University of California–Los Angeles, 650 Charles E. Young Drive, Los Angeles, CA 90095; email: *yifang@ucla.edu*.

Although this document was produced with partial funding by the United States Environmental Protection Agency under Assistance Award CR–83234701 to the Health Effects Institute, it has not been subjected to the Agency's peer and administrative review and therefore may not necessarily reflect the views of the Agency, and no official endorsement by it should be inferred. The contents of this document also have not been reviewed by private party institutions, including those that support the Health Effects Institute; therefore, it may not reflect the views or policies of these parties, and no endorsement by them should be inferred.

* A list of abbreviations and other terms appears at the end of the Investigators' Report.

INTRODUCTION

HEALTH EFFECTS OF TRAFFIC EMISSIONS

A large number of toxicological and epidemiological studies have linked exposure to vehicle-related air pollutants with premature mortality and the exacerbation of various respiratory and cardiovascular diseases (Boldo et al. 2006; Brunekreef et al. 1997; de Kok et al. 2006; Lin et al. 2002; McCreanor et al. 2007; Peters et al. 2004; Sandstrom and Brunekreef 2007; vanVliet et al. 1997; Vinzents et al. 2005). In vitro toxicological studies found that traffic-emitted particulate matter (PM) produces several types of adverse cellular effects, including mutagenicity, cytotoxicity, DNA damage, and stimulation of proinflammatory cytokine production (de Kok et al. 2006). Short-term exposure to traffic emissions can cause oxidative DNA damage (Vinzents et al. 2005), increase concentrations of inflammatory and cardiopulmonary biomarkers, and promote airway acidification (Hinds 2010; McCreanor et al. 2007). A 4- to 12-week exposure to traffic emissions increased the inflammatory/endothelial response of elderly men, especially among those with diabetes (Alexeeff et al. 2011). Long-term exposure to traffic emissions has been linked to an elevated risk of respiratory and cardiovascular mortality (Brunekreef et al. 2009) and to adverse pregnancy outcomes (Wu et al. 2011).

Among vehicle-related pollutants, the exposure to PM was estimated to have the greatest health costs, ranging between 17 and 266 billion U.S. dollars (McCubbin and Delucchi 1999). Currently the U.S. Environmental Protection Agency (U.S. EPA) regulates PM on a mass basis as PM_{10} (PM $\leq$ 10 μm in aerodynamic diameter) and $PM_{2.5}$. However, PM number concentration was suggested to be more closely related with adverse health effects than was PM mass concentration (Oberdörster 2001; Peters et al. 1997). In urban environments, 90% of ambient particles are UFP, which dominate the PM number concentration but contribute only a small portion to PM mass concentration. Previous studies in animals and humans showed that UFP might be more toxic per unit mass than fine particles or coarse particles (particles larger than $PM_{2.5}$) (Alessandrini et al. 2006; Delfino et al. 2005; Ferin et al. 1990; Frampton et al. 2006; Peters et al. 1997). Because of their small size, UFP can evade alveolar macrophage clearance from the lung, penetrate the epithelium, enter the circulatory system, and deposit in the brain (Elder et al. 2006; Samet et al. 2009).

Traffic emissions are usually the dominant source of UFP and other air pollutants in an urban environment (Cyrys et al. 2008). UFP concentrations measured near major freeways were as high as 2.0×10^5 particles/cm^3, over 10 times higher than the background level (Zhu et al. 2002; 2007). An on-road study in Los Angeles showed that even though commuting time counted for only 6% (90 minutes) of Californians' daily time, 36% of total daily UFP exposure resulted from on-road vehicle emissions (Fruin et al. 2008). Many studies have estimated commuters' exposure to UFP from vehicle emissions (Cheng et al. 2010; Fruin et al. 2008; Knibbs and de Dear 2010; Zhu et al. 2007), but knowledge of children's exposure remains relatively limited. Children may be more susceptible to the adverse health effects of UFP because their physiological and immunological systems are still developing. Compared with adults, children may receive a higher dose of PM to their lungs due to a greater fractional deposition with each breath and to a larger ventilation rate relative to lung size (Bennett and Zeman 1998). In addition, children have distinctly different time-activity patterns from adults. Therefore, children's exposure to UFP may differ from that of adults, and exposure assessments based on adults may not apply to children.

UFP IN SCHOOL BUSES

In the United States, about 25 million children ride school buses (U.S. EPA 2012). A typical child rides a school bus 180 days per year for a decade. The median age of school buses in the United States was reported to be 8 years and approximately 10% of them were older than 15 years (U.S. EPA 2010). About 90% of 450,000 public school buses are powered by diesel fuel (U.S. EPA 2012). Wehner and colleagues (2009) found that for particles larger than 25 nm, the emission factor per mile was about 100 times higher for diesel engines than for gasoline engines; therefore, children are likely to be exposed to high levels of air pollutants while riding diesel-powered school buses. Sabin and colleagues (2005) reported that concentrations of BC, particle-bound polycyclic aromatic hydrocarbon (PB-PAH), and nitrogen dioxide (NO_2) were significantly higher inside conventional diesel school buses than in compressed clean natural gas school buses. Behrentz and colleagues (2005) found that the mean concentrations of BC, PB-PAH, NO_2, $PM_{2.5}$, and fine particle counts during school bus commutes were higher than those at nearby urban background sites. Even though school bus commutes accounted for less than 10% of a child's day, they contributed 33% of a child's daily exposure to BC (Behrentz et al. 2005). The inhalation of school bus emissions per onboard child was reported to be on average 100,000–1,000,000 times greater than for a typical resident in the South Coast Air Basin of California (Marshall and Behrentz 2005).

To reduce children's exposure to diesel particles, the U.S. EPA has recommended retrofitting old diesel-powered school buses with certified retrofit technologies, such as a DOC muffler and a CFS (U.S. EPA 2011). DOC was verified by the U.S. EPA to produce reductions for air pollutants contained in tailpipe emissions: 20%–26% for PM mass concentration, 49%–66% for hydrocarbons, and 38%–41% for CO, whereas DOC and CFS together achieved a reduction of 28%–32% of PM mass concentration, 42% of hydrocarbons, and 31%–34% of CO (U.S. EPA 2011). According to the Donaldson Company (2006), a CFS can reduce PM mass concentration emissions discharged directly from the crankcase by 2%–80%. However, specific information concerning the reductions for in-cabin PM number concentration, a more relevant indicator of children's exposure during school bus commutes, was not given by the U.S. EPA or by the Donaldson Company.

Only a few studies have evaluated the effectiveness of retrofit technologies on reducing in-cabin air pollutants, and the results were contradictory. Hammond and colleagues (2007) measured in-cabin number concentrations of particles 20 nm to 1 μm in diameter, and reported a 15%–26% decrease between retrofitted and nonretrofitted buses. Trenbath and colleagues (2009) tested three school buses for in-cabin particles ranging from 10 nm to greater than 1 μm, and found a 33%–41% reduction in average PNC before and after retrofitting. Three school buses in Round Rock, Texas, were measured for in-cabin particles between 20 nm and 1 μm in diameter, and the change of PNC ranged from 7% higher to 60% lower after retrofitting (Rim et al. 2008). A multicity study conducted by Hill and colleagues (2005) found no significant change in PNC in similar particle size ranges with the use of retrofit devices. Such disagreement was presumably due to the lack of control of confounding factors such as bus characteristics, meteorological conditions, and on-road pollutant levels, making it difficult to draw statistically solid conclusions about the efficiency of retrofit devices.

SPECIFIC AIMS

From an exposure assessment perspective, school buses are important microenvironments where children congregate and spend a substantial proportion of their active time. However, this microenvironment is not often the subject of study. To accurately estimate children's total daily exposure to PM and other air pollutants, it is crucial to study this microenvironment. The overall objective of this study was to identify the conditions under which children are likely to be exposed to high levels of UFP and other air pollutants in and around school buses. Once the variability of children's exposure to UFP and other air pollutants is known, future epidemiological studies can be designed to better target children exposed to high levels of pollutants. To accomplish the objective of this study, the following specific aims were pursued:

1. Identify important factors affecting air pollutants inside school bus cabins while driving and idling.
2. Quantify the effectiveness of two U.S. EPA-verified retrofit technologies for diesel-powered school buses on reducing tailpipe emissions and in-cabin air pollutant levels.
3. Explore the potential of in-cabin filtration to reduce PM levels inside school bus cabins.

METHODS AND STUDY DESIGN

POLLUTANTS AND INSTRUMENTS

Pollutant Measurements

Real-time air quality measurements included PNC, fine and UFP size distribution, $PM_{2.5}$ mass concentration, BC concentration, and CO and CO_2 concentrations. Other measurements included temperature and relative humidity (RH); location, traffic density, and driving speed; and wind speed and direction. Table 1 summarizes the measured parameters and associated instruments used in this study.

PNC PNC was measured by a water-based condensation particle counter (CPC, model 3785, TSI, St. Paul, MN). Particles larger than 5 nm in diameter are detected and counted by a simple optical detector after supersaturated water vapor condenses onto the particles and makes them grow into detectable droplets. The sampling flow rate of the CPC is 1.0 L/min. The CPC can measure from 0 to 10^7 particles/cm^3 with ±10% accuracy at $< 2 \times 10^4$ particles/cm^3. The response time is less than two seconds for a 95% response to a concentration step change. For most parts of the study, two identical CPCs were employed to simultaneously measure the in-cabin air and the ambient air. In a sub-study, which will be described later, measurements were simultaneously made in the upwind air, in-cabin air, and the air close to the tailpipes. A CPC 3781 and a CPC 3007 (TSI, St. Paul, MN) were used to measure PNC for the upwind air and air close to the tailpipes, respectively. The CPC 3781 counts the total PNC for particles larger than 6 nm. The sampling flow rate is 0.6 L/min. It can measure 0 to 5×10^5 particles/cm^3 with ±10% accuracy at 5×10^5 particles/cm^3. The CPC 3007 measures PNC for particles larger than 10 nm with a sampling flow rate of 0.7 L/min. It can measure 0 to 5×10^5 particles/cm^3 with ± 20% accuracy. Because the lower size cut

Table 1. Instruments and Measured Environmental Parameters[a]

Instrument	Species/Parameter	Detection Limit	Flow Rate (L/Min)	Log Interval[b]
CPC 3785	Particle number (> 5 nm)	Single particle	1.0	1 sec
CPC 3871	Particle number (> 6 nm)	Single particle	0.6	1 sec
CPC 3007	Particle number (> 10 nm)	Single particle	0.7	1 sec
SMPS	Size distribution (7.6–289 nm)	Single particle	1.0	2 min
DustTrak 8520	$PM_{2.5}$ mass	1 µg/m^3	1.7	30 sec
Aethalometer AE-42	BC mass	1 µg/m^3	2–5	1 min
Q-trak 7565	CO, CO_2, Temperature, RH	0.1 ppm, 1.0 ppm 0.1°C, 1%	n/a	30 sec
GPSmap 76CSx	Location and speed	10 m	n/a	10 sec

[a] CPC indicates condensation particle counter; SMPS indicates scanning mobility particle sizer; n/a indicates not applicable.

[b] The log interval is the time between recorded measurements.

differed among CPC instruments, the measurement results were not directly comparable among instruments. Instead, the results from the same CPC instrument were compared across different testing conditions. In this study, PNC is a general term referring to particle number concentrations measured by any of the CPCs mentioned above. Specific size cuts are mentioned only when relevant.

Fine and UFP Size Distribution Fine and UFP size distribution in the size range 7.6–289 nm were measured by a scanning mobility particle sizer (SMPS) at 2-minute intervals. An SMPS consists of two components: (1) an electrostatic classifier (model 3080, TSI, St. Paul, MN) with a long differential mobility analyzer (DMA, model 3081, TSI) to select particles of a given size, and (2) a CPC 3785 to count the particles. In the electrostatic classifier, the aerosol is first neutralized to the Boltzmann equilibrium charge distribution by passing through a Kr-85 bipolar neutralizer. Particles are then separated according to their electrical mobility in the DMA. Particles within a narrow range of electrical mobility exit with the airflow through a small slit located at the bottom of the collector rod and are transferred to a CPC to determine the particle concentration. The SMPS system is automated with a personal computer that controls the individual components and performs data logging and reduction. SMPS data reduction and analysis were done by the Aerosol Instrument Manager software (version 4.0, TSI).

In addition to particle size distribution, the SMPS also reports the total PNC in the size range 7.6–289 nm, which is defined as the UFP number concentration in this study. On a number basis, over 90% of particles emitted by vehicle engines are in a size range < 100 nm, therefore UFP number concentration measured by an SMPS is a good approximation of PNC measured by the CPCs.

$PM_{2.5}$ Mass Concentration DustTrak photometers (model 8520, TSI, St. Paul, MN) with a $PM_{2.5}$ inlet impactor were used to monitor particle mass concentration continuously. Powered by an internal battery, the DustTrak samples air at a constant flow rate of 1.7 L/min using a built-in diaphragm pump. The sampled airstream passes through a light-scattering optical sensing zone. The detected signal is processed by lock-in circuitry followed by high-resolution digitization. A part of this filtered stream is continuously diverted through and over all optically sensitive areas to protect them from particle deposition. The DustTrak covers a concentration range of 1 µg/m^3 to 100 mg/m^3 with a resolution of ±0.1% or ±0.001 mg/m^3, whichever is greater.

BC A Magee Scientific Aethalometer (model AE-42-2, Magee Scientific, Berkeley, CA) was used to measure BC concentrations at 1-minute intervals. This device measures the attenuation of a beam of light transmitted through a filter while the filter is continuously collecting an aerosol sample. The instrument measures particulate BC in the near infrared at a wavelength of 880 nm using a solid-state source. The rate of accumulation of BC is proportional to both the BC concentration in the airstream and the flow rate. The instrument has a sensitivity of < 0.1 µg/m^3 and an accuracy of 5%. Data are continuously logged into an internal data logger.

CO and CO_2 A Q-trak indoor air quality monitor (model 7565, TSI, St. Paul, MN) was used to determine the concentrations of CO and CO_2 as well as temperature and RH at 30-second intervals. An electrochemical cell was used for detecting CO and CO_2. For CO_2, the detection range is 0–5000 ppm with an accuracy of ±3% or ±50 ppm, whichever is greater. For CO, the detection range is 0–500 ppm with an accuracy of ±3% or ±3 ppm, whichever is greater. Data are continuously logged into an internal data logger and are later downloaded to a personal computer. Data reduction and analysis of the Q-trak and DustTrak output were done by the TrakPro software (version 3.33, TSI).

Meteorological Data Two units of Q-trak were used to monitor temperature and RH inside and outside the test school buses. Wind speed and direction together with ambient temperature and RH were obtained from the local weather station nearest to the sampling site.

Location and Speed The speed, direction, and location of the school buses were monitored every 10 seconds with a GPS tracking device (GPSmap 76CSx, Garmin). City Select North American map data were loaded onto the laptop to provide a trace of the driving path during each run.

Quality Assurance and Quality Control

Before each test, all instruments and laptops were synchronized to a satellite-signaled clock. Flow and zero checks of all instruments were conducted before and after each sampling period. The sampling flow rates were measured by a DryCal DC-Lite primary flow meter (Bios International Corporations, Butler, NJ). A zero reading was checked by connecting with a HEPA filter for the CPC, DustTrak, and Aethalometer. The identical instruments used simultaneously for in-cabin and ambient air measurements were collocated to collect data side-by-side for at least 10 minutes before and after each test to determine the relationship of their readings. Appendix Figure A.1 shows scatter plots for the comparison of the readings from two sets of collocated instruments for PNC, $PM_{2.5}$, CO_2, temperature, and RH. (Appendix A is available on the HEI Web site.) The set 1 instrument was always used for in-cabin measurements. In the analysis, data from set 2 instruments were corrected to the readings of the set 1 instrument by the respective best-fit Pearson least-square linear regression formula.

The SMPS systems were calibrated in the laboratory before and after the study by measuring monodisperse polystyrene latex spheres (PSL, Polysciences, Warrington, PA). DustTrak data were calibrated against simultaneous filter-based mass measurements using a personal environmental monitor. The Q-trak monitor was calibrated by sampling standard CO and CO_2 twice per year. For the Aethalometer, the flow rate and the response to a factory-supplied test filter was checked before and after each test.

Flexible conductive tubing with a stainless steel monitoring probe was used for all particle-measuring instruments to avoid particle losses due to electrostatic forces. Sampling line losses were measured by sampling ambient air with and without the sampling probes in place. The mean difference between the paired measurements characterized particle loss in the sampling lines, which was usually less than 5%.

Most of the monitoring instruments were robust against vibration, heat, and humidity; however CPCs were sensitive to vibration, especially when placed in a running school bus. Occasionally a CPC reported an error due to intensive vibration and gave invalid readings. Research staff recorded these events on field logs, including the cause of a problem, the starting and ending time of such problem, and if the problem had been fixed on site. The readings made during these events were removed from the database. Real-time continuous time-series plots were also reviewed during the measurement and after each sampling day. Rapidly changing, anomalous, or otherwise suspicious data were examined. Such data were removed if a reason was identified (e.g., instrument malfunction, communication problem between the laptop and instruments).

STUDY DESIGN

This study consisted of four sub-studies, as summarized in Table 2. On-road tests were conducted in Beeville, Texas, where air pollutants inside school buses were monitored when driving on roads. Idling tests (IT) were conducted in Los Angeles, California, to study the impact of idling on UFP in and around school buses. Retrofit tests (RT) evaluated the performance of two retrofit systems in Corpus Christi, Texas. RTs were carried out to study the effectiveness of retrofit systems on tailpipe emissions and in-cabin pollutants while idling in an open garage and driving on regular pick-up/drop-off routes. The effectiveness of in-cabin filtration was evaluated by the HEPA air purifier test in Corpus Christi.

Meteorological conditions such as temperature and RH have been reported to affect traffic exhaust PNC (Jamriska et al. 2008; Ronkko et al. 2006). It is thus important to conduct the pre-retrofit and post-retrofit measurements under similar meteorological conditions. Table 2 shows that temperature and RH were fairly comparable between pre- and post-retrofit measurements in this study. The changes of temperature and RH observed in this study were similar to the meteorological conditions reported by Trenbath and colleagues (2009). Therefore during the study period, the

Table 2. Summary of School Bus Tests and Meteorological Parameters

Test	Date	Location	Temperature (°C)	Relative Humidity (%)
On-road	3/18/2008–5/28/2008	Beeville, TX	25–39	24–61
Idling	8/30/2010–9/10/2010	Los Angeles, CA	18–23	58–80
Retrofit				
In open garage	7/6/2009–7/10/2009	Corpus Christi, TX	29–40	33–69
On-road				
Before retrofit	4/13/2009–4/20/2009	Corpus Christi, TX	24–29	48–60
After retrofit	10/28/2009–11/11/2009	Corpus Christi, TX	21–27	40–66
HEPA air purifier	4/19/2010–4/22/2010	Corpus Christi, TX	25–28	44–51

effect of temperature and RH on particles emitted by school buses was expected to be insignificant.

School Bus Characteristics

Twenty-four school buses were studied, covering a wide range of model years (MY), bus maker, and engine model, to represent the school buses commonly in use in the United States. School bus characteristics are shown in Table 3. Apart from the buses in the retrofit and the HEPA air purifier tests, none of the buses was retrofitted.

Four school buses were selected from the school bus fleet that served the Beeville School District in South Texas. Two 1990 MY school buses and two 2006 MY school buses were selected to represent the old and new school buses in service in the on-road tests, which investigated the in-cabin PM levels while the buses were being driven. All tested school buses were manufactured by Blue Bird, and the engine model was D5.9L. As shown in Table 3, the capacity of these school buses ranged from 70 to 74 persons. The engines of the 1990 MY school buses were under the floor of the cabin to the right of the driver's seat, while the engines of the 2006 MY school buses were under the hood in front of the cabin. The older school buses had about four times more mileage than the newer school buses. There were no AC units in the 1990 MY school buses. Thus, the AC units in the newer ones were kept off and only window position was changed during this study.

Nine school buses were selected from Tumbleweed Transportation, California, which provides school bus transport services in the greater Los Angeles area. One bus was a 1999 MY and the other eight were 2005 MY. These buses were of three different sizes with maximum capacities of 36, 48, and 72 persons. All of the buses had engines located under the front hood; none of the buses had AC units. These school buses were employed in the idling tests to assess the impact of idling on UFP levels in and around school buses.

Eleven school buses were selected from a fleet of 197 buses in the Corpus Christi Independent School District (CCISD), Texas. The manufacturers of these buses included Bluebird, Freightliner, and Navistar, the most popular brands used in the United States. The bus engines, comprising four different models, were made by two engine makers, Cummins and International. The buses ranged from 10 to 20 years old, representing the relatively old school buses used in CCISD. These buses were employed in the retrofit tests and the HEPA air purifier tests to evaluate the performance of retrofit technologies and in-cabin filtration devices. For school buses C1 and C2, there was no AC unit installed and the engine compartment was located under the floor of the cabin to the right of the driver. The rest of the buses had one or two AC units, depending on the size of the bus. Except for buses C9 and C11, their engines were located under the front hood.

On-Road Tests

To assess children's exposure to air pollutants inside school buses during driving conditions, four school buses (B1–B4) were studied on the roads from March 18 through May 28, 2008 in Beeville, Texas. On each sampling day the school buses were tested in the morning run from 6:30 to 8:30 and in the afternoon run from 15:00 to 17:00. Buses were tested on two different days (only B4 was sampled on one day). The two routes were selected to represent different student transportation patterns. A 19-km town route, which included surface streets and freeways, was chosen to represent town roads. A 32-km rural route, which included a long section of rural road, was selected to represent rural roads. In the morning, the school buses

started on the rural route to pick up children and then went to a bus transfer station to unload them. After idling for approximately 15 minutes, the test buses left, following other buses to an elementary school parking lot, where the children were unloaded (usually in about 10 minutes). The buses then continued on the town route. It took approximately two hours to complete both routes. In the afternoon, the test buses picked up children from the elementary school, transported them to the bus transfer station, ran on the rural route to return the children to their homes, and then went back to the elementary school to resume the test on the town route. The passenger numbers varied between 7 and 45, including a driver and the research staff operating the equipment. Buses B2 and B4 were tested during the 2008 spring break without children on board. As summarized in Table 4, the parameters measured inside the cabins included PNC, fine and UFP size distribution, $PM_{2.5}$, CO_2, CO, and BC. Detailed information on the instruments was described earlier in this report. Passenger numbers inside the school buses and the operation of doors and windows were recorded manually. All instruments were fixed on a two-layer portable sampling platform secured on the rear seats inside the school buses. Vibration mounts were used between the two layers to protect instruments from on-road vibration. The instruments were powered by four deep-cycle marine batteries. Air pollutant concentrations outside the cabins were not monitored during the on-road test.

Table 3. School Bus Characteristics

City / Test[a]	Bus Number	Bus Maker	Engine Maker	Engine Model	Model Year	Mileage (miles)	AC	Engine Position	Maximum Capacity (person)
Beeville, TX									
ORT	B1	Bluebird	Cummins	D5.9L	1990	311,478	No	In-cabin	70
ORT	B2	Bluebird	Cummins	D5.9L	1990	300,104	No	In-cabin	70
ORT	B3	Bluebird	Cummins	D5.9L	2006	62,654	Yes	Front hood	74
ORT	B4	Bluebird	Cummins	D5.9L	2006	68,333	Yes	Front hood	74
Los Angeles, CA									
IT	L1	Intel/Mid	Navistar	T444E	1999	125,328	No	Front hood	36
IT	L2	Freightliner	Mercedes	Meb 900	2005	98,922	No	Front hood	48
IT	L3	Freightliner	Mercedes	Meb 900	2005	10,153	No	Front hood	48
IT	L4	Freightliner	Mercedes	Meb 900	2005	93,356	No	Front hood	48
IT	L5	Freightliner	Mercedes	Meb 900	2005	104,351	No	Front hood	48
IT	L6	Freightliner	Mercedes	Meb 900	2005	58,974	No	Front hood	72
IT	L7	Freightliner	Mercedes	Meb 900	2005	61,251	No	Front hood	72
IT	L8	Freightliner	Mercedes	Meb 900	2005	59,917	No	Front hood	72
IT	L9	Freightliner	Mercedes	Meb 900	2005	68,299	No	Front hood	72
Corpus Christi, TX									
RTRB	C1	Bluebird	Cummins	D5.9L	1990	79,674	No	In-cabin	77
RTRA	C2	Bluebird	Cummins	D5.9L	1990	86,595	No	In-cabin	77
RTR, AP	C3	Freightliner	Cummins	D5.9BTA	1999	127,051	Yes	Front hood	77
RTR, AP	C4	Bluebird	Cummins	D5.9BTA	1996	130,727	Yes	Front hood	77
RTR, RTG, AP	C5	Freightliner	Cummins	D5.9ISB	1999	78,364	Yes	Front hood	38
RTR	C6	Freightliner	Cummins	D5.9L	1999	146,395	Yes	Front hood	38
RTR, RTG, AP	C7	Navistar	Navistar	D7.3L	1992	201,381	Yes	Front hood	35
RTG	C8	Freightliner	Cummins	D5.9BTA	1999	110,778	Yes	Front hood	77
RTG	C9	Freightliner	Cummins	D5.9L	1997	107,308	Yes	In-cabin	34
RTG	C10	Freightliner	Cummins	D5.9L	1999	79,366	Yes	Front hood	38
RTG	C11	Bluebird	Cummins	D5.9L	1992	12,123	Yes	In-cabin	72

[a] Abbreviations: AP = HEPA air purifier test; IT = idling test; ORT = on-road test; RTG = retrofit test in open garage; RTR = retrofit test on road (including both before and after retrofit); RTRA = retrofit test on road after retrofit; RTRB = retrofit test on road before retrofit.

Table 4. Instruments and Measured Parameters

Tests/ Species/Parameter	Instrument[a]		
	In-Cabin	Ambient	Tailpipe
On-Road			
PNC	CPC 3785		
Fine and UFP size distribution	SMPS		
$PM_{2.5}$	DustTrak 8520		
BC	Aethalometer AE-42		
CO, CO_2, temperature, RH	Q-trak 7565		
Location and speed	GPSmap 76CSx		
Idling			
PNC		CPC 3781	CPC 3007
Fine and UFP size distribution	SMPS		
$PM_{2.5}$	DustTrak 8520	DustTrak 8520	
CO_2, temperature, RH	Q-trak 7565	Q-trak 7565	
Retrofit			
In garage			
PNC	CPC 3785		CPC 3785
Fine and UFP size distribution	SMPS		SMPS
$PM_{2.5}$	DustTrak 8520		DustTrak 8520
BC			Aethalometer AE-42
CO_2, temperature, RH	Q-trak 7565		
On-road			
PNC	CPC 3785	CPC 3785	
Fine and UFP size distribution	SMPS	SMPS	
$PM_{2.5}$	DustTrak 8520	DustTrak 8520	
BC	Aethalometer AE-42		
CO_2, temperature, RH	Q-trak 7565	Q-trak 7565	
HEPA Air Purifier			
PNC	CPC 3785	CPC 3785	
Fine and UFP size distribution	SMPS	SMPS	
$PM_{2.5}$	DustTrak 8520	DustTrak 8520	
BC	Aethalometer AE-42		
CO_2, temperature, RH	Q-trak 7565	Q-trak 7565	

[a] In-cabin measurements were taken at the rear seats of the bus. Ambient measurements were made either directly outside the (driving) bus, or 2 m upwind of the bus (idling test). Tailpipe measurements were taken 0.5 m from the tailpipe (idling test), or directly sampled at the tailpipe with the use of a dilution system (open garage retrofit test).

Because the 1990 MY school buses had no AC units, the AC units in the newer ones were kept off and only window position was changed during this study. Two window positions tested in the on-road tests included: (1) window-closed condition (buses B2 and B4): all windows were kept closed, except two windows in bus B4 could not be closed tightly; (2) window-open condition (buses B1 and B3): three front windows on each side of the buses were open by about 20 cm.

Idling Tests

Measurements In and Around Idling School Buses

To investigate the impact of idling on UFP levels in and around school buses, nine school buses (L1–L9) were studied from August 30 through September 10, 2010, in Los Angeles, California. To control the variability of ambient air pollutant concentrations, this study was conducted in an open green space without traffic emissions on

the upwind side. The study location is shown in Appendix Figure A.2. Meteorological data, obtained from a weather monitoring station 5 km east of the study site, are summarized in Appendix Table A.1. During the study period, the dominant wind direction was from southwest to northeast, eliminating the influence of emissions from interstate highway 405 on the eastern side of the study area. Average daily wind direction ranged from 179° to 205°, and wind speed varied between 2.5 and 3.6 m/sec. Ambient temperature was between 18.2 and 22.6 °C, while RH varied from 58% to 80%. Constant wind direction during the study period allowed the buses to be positioned either parallel or perpendicular to the wind. Stable meteorology conditions also reduced the day-to-day variability of background pollutant concentrations, leading to a better signal-to-noise ratio during the measurements.

Five scenarios were set up by varying the emission source, wind direction, and window position. The layout of school buses for each scenario is shown in Appendix Figure A.3. Arrows indicate wind directions. Open boxes refer to school bus bodies, gray areas indicate tailpipe emissions, and open circles with numbers represent the equipment for air pollutant measurements. Constant wind direction during the study period allowed positioning the school buses either parallel to the wind, so that the bus cabin was downwind of engine emissions; or perpendicular to the wind, so that the bus cabin was upwind of the tailpipe emissions. Diesel emissions were either from the tested school buses' own tailpipes or from the tailpipes of the nearby school buses. Two window positions were tested for each scenario: (1) all windows were closed, although some of the windows could not be closed tightly; or (2) eight rear windows, four on each side, were open by 20 cm.

Scenarios A and B used one school bus. In scenario A, the school bus was parked perpendicular to the wind direction. Air quality data were collected with the engine off and then on for approximately 30 minutes each. In scenario B, the bus was turned 90° to simulate a parallel wind, so that the wind blew from the bus' tailpipe toward its hood. Scenarios C–E employed two school buses parked at a 90° angle to each other with the tailpipe of one pointing to the side of the other near the back at a distance of about two meters. The upwind bus was perpendicular to the wind and the other bus was parallel to the wind along the downwind side. Air quality data were first collected with both bus engines off for 30 minutes. In scenario C, only the engine of the downwind bus was turned on for 30 minutes. Air quality data in and around the upwind bus were collected. In scenario D, only the engine of the upwind bus was turned on and the air quality data in and around the downwind bus were collected for 30 minutes. In scenario E the engines of both buses were turned on and air quality data in and around both buses were collected for 30 minutes.

Doors remained closed and the bus was unoccupied during the measurements. Other than the buses being tested, no school buses were allowed to operate nearby. In the absence of emission sources on the upwind side, a difference in air pollutant concentrations between engine-off and engine-on conditions was primarily attributed to emissions from the idling buses. Scenarios A and B investigated the contribution of self-pollution with a perpendicular and parallel wind, respectively. Scenarios C and D investigated the impact of the emissions from idling buses on nearby buses. Scenario E investigated a scenario in which at least two buses were idling close together at a bus transfer station or a school parking lot.

For each scenario, the air quality of the upwind air, in-cabin air, and air close to the tailpipes was measured simultaneously, as summarized in Table 4. One set of instruments, including a CPC 3781 and a DustTrak, measured the PNC and $PM_{2.5}$ mass concentration in the upwind air approximately 2 meters from the bus at a height of 1 meter. The second set, a CPC 3007, measured PNC close to the tailpipes at the height of the tailpipe and 0.5 meters away from the point of tailpipe exhaust in the direction of the exhaust. The third set, including an SMPS, a DustTrak, and a Q-Trak, measured in-cabin air quality at the rear seat of each bus. The total PNC in the size range of 7.6 to 289 nm measured by the SMPS, or the UFP number concentration, was used as a proxy of the in-cabin PNC, as discussed earlier. For scenario E, because the air quality of both the upwind and downwind buses was measured simultaneously, two identical sets of SMPS were used for in-cabin UFP number concentrations. It should be noted that different models of particle counting instruments were used for the measurements of the upwind air, in-cabin air, and air close to the tailpipes. Each model has a different lower size cut, as shown in Table 1. Therefore the measurement results from these three locations were not compared directly. Instead, the comparisons were made between engine-off and engine-on conditions to assess the impact of tailpipe emissions from idling school buses on air quality at each of the three locations.

Air Exchange Rate and Deposition Rate For each school bus in the idling test, air exchange rates were estimated by the CO_2 decay method described by Zhu and colleagues (2005). From a compressed CO_2 cylinder, CO_2 was released into the cabin until the concentration was higher than 2000 ppm. The decay of CO_2 concentration was measured by

a Q-trak indoor air quality monitor. Then the air exchange rate was calculated by

$$\lambda = \frac{1}{t - t_0} \ln\left(\frac{C_{in}(t) - \overline{C_{out}}}{C_{in}(t_0) - \overline{C_{out}}}\right) \quad (1)$$

where λ is the air exchange rate (per hour), and t_0 and t are the beginning and end of the sampling interval (in hours); the averaged ambient CO_2 concentrations (ppm) for the period from t_0 to t is $\overline{C_{out}}$; the indoor CO_2 concentrations (ppm) measured at time t_0 and t are indicated as $C_{in}(t_0)$ and $C_{in}(t)$, respectively.

In the idling test, particle deposition rates were also determined for each bus based on data collected when the bus was parked parallel with the wind direction and with eight rear windows open by 20 cm. The engine was turned on for about 15 minutes. The particle sources for deposition were diesel particles emitted by the tailpipes and carried by the wind through open windows into the cabin. After the particle concentrations became stable, the engine was turned off and all the windows and doors were closed. The fan and recirculation system in the school bus were kept off. No passenger was present in the bus cabin. UFP deposition rates were determined based on size-resolved particle concentration decay that was sampled and recorded by an SMPS. Each test continued until the in-cabin UFP concentrations approached ambient levels (about 20–30 minutes). This process was repeated two to three times for each bus. Deposition rate was calculated as:

$$\ln\left(\frac{\mathrm{PNC}(t)}{\mathrm{PNC}(0)}\right) = (\lambda + k)t \quad (2)$$

where $PNC(t)$ and $PNC(0)$ are the PNCs (particles/cm^3) measured at time t and 0, respectively (Gong et al. 2009). The air exchange rate and the deposition rate (per hour) are indicated by λ and k, respectively. Data were calculated for each particle size to estimate size-resolved UFP deposition rates. The sum of total PNC in the size range 7.6–289 nm measured by the SMPS was also used to calculate the overall particle deposition rate.

The in-cabin volume and the interior surface area were determined for each bus by measuring the dimensions of each component present inside the cabins. Interior surface materials were the same for all buses, including rubber, plastic, artificial leather, metal, and glass. Additional areas of in-cabin geometric structure such as seats, dashboards, and monitoring instruments were included as interior surface area.

Retrofit Tests

The performances of two retrofit devices, a DOC muffler (Series 6100, Donaldson Company, Minneapolis, MN) and a Spiracle CFS (Donaldson Company) were tested from April 13 through November 11, 2009 in Corpus Christi, Texas (Table 2). The DOC was mounted on the exhaust tubing to reduce tailpipe emissions from in-use diesel engines. The CFS was connected to the crankcase vent and the engine intake to reduce emissions from the open crankcase vents of diesel engines. The effectiveness of DOC and CFS were evaluated under two school bus operating conditions: (1) when the school buses were idling in an open garage, and (2) when the school buses were being driven on roads.

We did plan to test the performance of a third retrofit device, a diesel particle filter. However, we were unable to conduct those tests because of an unexpected administrative delay at the bus company. This limited the comparisons of the performances of different retrofit technologies.

Retrofit Tests in an Open Garage Retrofit tests in an open garage were conducted on six school buses between July 6 and July 10, 2009. Windows and doors were closed and AC units were off during testing. Before retrofitting, the tailpipe emissions and the in-cabin air quality of each bus were measured simultaneously for 1.5 hours while the bus was idling. Then either the DOC or the CFS was installed and the tests were repeated. Finally, the second retrofit device (DOC or CFS) was installed and the tests were repeated with both devices installed. The DOC was installed first in three of the buses (C5, C7, C8) and the CFS was installed first in the other three buses (C9, C10, C11) (see Table 3). In-cabin air measurements were made with instruments fixed on the sampling platform at the back of the school buses. Measurements were taken of PNC, fine and UFP size distributions, $PM_{2.5}$ mass concentrations, and CO_2, as well as temperature and RH. The measurements and instruments are summarized in Table 4.

For tailpipe emission monitoring, a flexible aluminum duct with a diameter of 10 cm was attached to the end of the tailpipe and several holes were drilled through the duct about 20 cm away from the point of tailpipe exhaust. Flexible conductive tubing with a 1-cm diameter stainless steel monitoring probe was used for all particle-measuring instruments. A dilution system was used to dilute tailpipe emissions before they entered the instruments. The exhaust in the conductive tubing was first cooled down to ambient temperature and then split into two streams. One was filtered by a HEPA filter to produce particle-free air and then was mixed with the other stream to dilute the exhaust. The

flow of particle-free air was controlled by an orifice in the HEPA branch; a flow meter was installed in the HEPA branch to allow determination of the dilution ratio. Before each test, the dilution ratio was determined by the simultaneous measurements of two identical CPCs, one with the dilution system and the other without. In this study, the dilution ratios ranged from 100 to 300. PNC, fine and UFP size distributions, $PM_{2.5}$, and BC concentrations were measured after dilution. The actual concentrations in tailpipe emissions were derived by multiplying the readings of each instrument by the dilution ratios measured prior to the tests.

Retrofit Tests on Roads Retrofit tests on roads were conducted from April 13 to April 20, 2009, for pre-retrofit conditions and from October 28 to November 11, 2009, for post-retrofit conditions. Six school buses (C1, C3–C7) were employed for pre-retrofit measurements. Bus C1 was removed from the bus fleet before the post-retrofit on-road tests, so it was replaced by a bus with a similar configuration, bus C2, for post-retrofit measurements.

School buses were driven on two in-use routes in sequence every morning and afternoon. In the morning, school buses started from the first route to transport children from their homes to an elementary school. Then the buses continued on the second route to pick up a different group of students and drop them off at a high school. In the afternoon, school buses were driven on the same routes but in a reverse direction. The total length of two routes was 62 kilometers, and it took approximately two hours to complete both routes. These routes covered a range of student transportation patterns and road types: 58%–63% on surface streets, 13%–19% on freeways, 7%–8% stopped at traffic lights, 7%–8% stopped for pick-up/drop-off, and 5%–15% idled at schools.

Pre-retrofit in-cabin pollutant concentrations were monitored as the school buses traveled on the rural and city routes. The post-retrofit in-cabin air pollutants were measured by the same instruments. At the same time, another set of instruments measured the surrounding air pollutant concentrations outside of the tested school buses. As summarized in Table 4, the measurements and instruments were similar to those in the open-garage tests. For in-cabin air, the real-time measurements included PNC, fine and UFP size distributions, $PM_{2.5}$, BC, CO, and CO_2 concentrations, as well as temperature and RH. For the ambient air, a set of identical instruments measured the same pollutants, except for ambient BC, since only one device was available for measurements. The location and driving speed were determined by a GPS unit. Both sets of instruments were fixed on the sampling platform at the rear of the bus cabin. To sample ambient air, flexible conductive tubing with a stainless steel monitoring probe went through a small opening in the window on the right side of the bus above the instrument platform. Tape secured the probes and sealed the opening in the window. A similar probe was used for in-cabin air sampling to compensate for any diffusion loss in the sampling lines.

For buses C1 and C2, two different ventilation conditions were tested: (1) in the morning, all windows were closed; and (2) in the afternoon, six rear windows were open by 20 cm. The effect of AC on pollutant concentrations was tested in the rest of the buses (C3–C7). All windows were kept closed with the AC unit either on or off. The number of passengers in the buses varied from 3–45 over the entire run, including one bus driver and two research staff.

HEPA Air Purifier Tests

Four school buses (C3–C6) were employed to study the effectiveness of stand-alone HEPA air purifiers in terms of removing in-cabin particles (between April 19 and 22, 2010, in Corpus Christi, TX). Each bus was tested for two runs, one in the morning, and one in the afternoon. The AC unit was kept off in the morning and was turned on in the afternoon. All the windows and doors were closed during the measurements. One or two stand-alone HEPA air purifiers (HAP 8650, Sunbeam Products, Boca Raton, FL) were located in the rear of the school buses. The air purifier was designed for large rooms (up to 40 m^2). It had a built-in fan with four speeds to draw air through a carbon odor filter and then through HEPA filters. The fan speed was set to maximum during the measurements. The in-cabin and ambient air pollutant concentrations were monitored simultaneously as the buses were driven on the same routes used in the retrofit tests described earlier. Two sets of identical instruments were placed side by side for concurrent in-cabin and ambient air measurements for PNC, $PM_{2.5}$, and CO_2, as well as temperature and RH, as summarized in Table 4. To sample ambient air, flexible conductive tubing with a stainless steel monitoring probe went through a small opening in the window on the right side of the bus above the instrument platform. Tubing of the same length was also used for in-cabin air sampling at the height of the passengers' breathing zone.

STATISTICAL ANALYSIS

Descriptive Analysis

All data from the monitoring instruments were transformed to 1-minute averages. SAS (Statistical Analysis Software version 9.2; SAS Institute, Cary, NC) codes were

developed to process and synchronize the data from environmental monitoring instruments and from field logs. The time series were evaluated visually for outliers and instrument malfunction. Statistical summaries were produced for each pollutant, including mean, standard deviation, median, minimum, and maximum concentrations in cabin air, ambient air, and tailpipe emissions under different conditions. Pollutant concentration in-cabin/outdoor (I/O) ratios corresponding to each condition were calculated. The distributions of air pollutant concentrations in different modes were compared, and scatter plots of different air pollutants were completed to give qualitative insight into the influence of various factors.

Effect of Tailpipe Emissions on In-Cabin Pollutant Levels When Idling

Using data collected in idling tests, we modeled the effect of engine operation on in-cabin particle levels over time under different scenarios using longitudinal regression analysis (Weiss 2005). In-cabin particle size distribution, UFP number concentration, and $PM_{2.5}$ mass concentration were analyzed using separate longitudinal regressions. Longitudinal models were fit with the SAS mixed procedure.

All outcomes were analyzed after log transformation to stabilize variance and linearize the relationship with time. We used a first-order autoregressive moving average model to account for the correlation of observations on a single bus over time that fit best over the many correlation models considered. With the engine off, we assumed that the background in-cabin particle size distributions were constant across different scenarios. With the engine on, the size distribution data were modeled as linear with time with slopes differing by scenario. Data collected under scenarios A–D were used to estimate the effects of window position (closed/open), wind direction (parallel/perpendicular), emission source (bus' own tailpipe or other buses' emissions), and all two-way interactions on the log of in-cabin concentrations. This analysis was performed for both PNC and for particles in different size ranges separately. The mean of $\exp(X)$, where X is a normal random variable with mean μ and variance σ^2, is $M = \exp(\mu + 0.5\sigma^2)$. When μ and σ^2 are estimated by $\hat{\mu}$ and $\hat{\sigma}^2$, we can estimate the particle count on the unlogged scale as $\hat{M} = \exp(\hat{\mu} + 0.5\hat{\sigma}^2)$, although this may underestimate the count due to ignoring uncertainty in $\hat{\mu}$ and $\hat{\sigma}^2$. We get $\hat{\mu}$, $\hat{\sigma}^2$, and their standard errors from the SAS mixed output procedure where $\hat{\mu}$ represents any mean estimate on the logged scale. To get a 95% confidence interval for $\hat{M}$, we simulate $\hat{M}_k = \exp(\hat{\mu}_k + 0.5\hat{\sigma}_k^2)$, where $\hat{\mu}_k$ and $\hat{\sigma}_k^2$ are generated from independent normal distributions with means $\hat{\mu}$ and $\hat{\sigma}^2$, and standard errors $SE(\hat{\mu})$ and $SE(\hat{\sigma}^2)$, respectively. If $\hat{\sigma}_k$ is negative, we set it equal to 0. We repeat for $k = 1, \ldots, K$; we set K equal to 1000. This assumes that the estimates of μ and σ^2 are independent and normal, which indeed they are asymptotically. We estimated M by the mean of the M_k's and constructed a 95% confidence interval by taking the 2.5% and 97.5% quantiles of the M_k's.

Performance of Retrofit Systems on Tailpipe Emissions and In-Cabin Pollutant Concentrations

To evaluate the performance of retrofit systems, a nonparametric statistical method, the Wilcoxon signed-rank test, was used to determine if there was a significant change in the air pollutant concentrations of tailpipe emissions and in-cabin air after each DOC or CFS retrofit. The concentrations in tailpipe emissions were averaged for each bus before retrofit, with an individual retrofit unit, and with the combination. For each school bus, the average concentrations of tailpipe-emitted air pollutants before retrofit were paired with those of the same school bus with an individual retrofit unit. Then a Wilcoxon signed-rank test was run to test the difference between the pre-retrofit and post-retrofit average concentrations at a significance level of $P < 0.05$. The same procedure was applied to test the difference between the average concentrations before and after both units had been installed. The same methods were also used to test the difference of in-cabin air pollutant levels between the pre-retrofit and post-retrofit average concentrations when idling in an open garage or when driving on roads.

Effect of AC Systems and Ambient Air on In-Cabin Pollutant Concentrations

The Wilcoxon signed-rank test was also used to determine the effect of AC systems on in-cabin pollutant levels in the retrofit on-road test. The in-cabin air pollutant concentrations were averaged for each bus with the AC off and again with the AC on. For each school bus, the average concentrations of in-cabin air pollutants with the AC on and off were paired for each bus. The Wilcoxon signed-rank test tested the difference between these two conditions at a significance level of $P < 0.05$. In addition, Spearman correlation coefficients between run-averaged concentrations of in-cabin and the surrounding air pollutants were calculated. Each of the six buses was tested twice, so there were 12 data points for calculating Spearman correlation coefficients between in-cabin pollutant concentrations and pollutant concentrations in the surrounding air.

RESULTS AND DISCUSSION

This study measured air pollutant concentrations inside school buses when they were driven on regular pick-up/drop-off routes and when they were idling in open air parking spaces. The factors affecting in-cabin pollutant concentrations were analyzed for these two conditions. The performance of two retrofit systems for diesel-powered school buses, a DOC muffler and a CFS, was evaluated regarding PNC and $PM_{2.5}$ concentrations during driving and idling, both for bus tailpipe emissions and for the air in bus cabins. The effectiveness of in-cabin filtration for reducing in-cabin PNC and $PM_{2.5}$ under driving conditions was also tested.

ON-ROAD TESTS

In-cabin air quality was measured inside four school buses in Beeville, Texas, to quantify UFP and other air pollutant concentrations in school buses that were being driven on roads. The descriptive statistics of air pollutant concentrations and environmental parameters are summarized in Table 5.

Most of the measurements were conducted under fairly consistent temperature and RH conditions, except for the test conducted on one of the 2006 MY school buses under the window-closed condition (bus B4). Average PNCs ranged from 7.4–34.0 $\times 10^3$ particles/cm^3, which were consistent with previous school bus studies in suburban areas, for example, 6.1–32.0 $\times 10^3$ particles/cm^3 in Austin (Rim et al. 2008), 9.5–53.0 $\times 10^3$ particles/cm^3 in Ann Arbor, and 7.0–50.0 $\times 10^3$ particles/cm^3 in Atlanta (Hill et al. 2005). However, our measurements were lower than the 28.0–72.0 $\times 10^3$ particles/cm^3 in Birmingham (Hammond et al. 2007) and 30.0–75.0 $\times 10^3$ particles/cm^3 in Chicago (Hill et al. 2005). For $PM_{2.5}$, the mean concentrations were 6.5–19.7 μg/m^3, which were similar to 7–20 μg/m^3 in Austin (Rim et al. 2008) but much lower than 21–62 μg/m^3 in Los Angeles (Sabin et al. 2005), 21–76 μg/m^3 in Ann Arbor, and 40–163 μg/m^3 in Chicago (Hill et al. 2005). The BC mass concentrations of 0.4–2.9 μg/m^3 in this study were also lower than those previously reported in Los Angeles (Sabin et al. 2005; Solomon et al. 2001) and in Connecticut (Wargo and Brown 2002).

The Pearson correlation coefficients among all measured pollutants are shown in Table 6. In general, correlations were weak among all measured pollutants except for the correlation between PNC and BC, which is consistent

Table 6. Pearson Correlation Coefficients for Air Pollutants Measured Inside School Bus Cabins

	PNC	$PM_{2.5}$	BC	CO	CO_2
PNC[a]	1.00				
$PM_{2.5}$	0.13	1.00			
BC	0.50	0.30	1.00		
CO	0.19	−0.38	−0.16	1.00	
CO_2	0.16	−0.07	0.09	0.37	1.00

[a] PNC (>5 nm) was measured by a CPC 3785.

Table 5. Summary of Environmental Parameters Collected in On-Road Tests

	1990 Model Year				2006 Model Year			
	Bus B1 Windows Open		Bus B2 Windows Closed		Bus B3 Windows Open		Bus B4 Windows Closed	
Environmental Parameters	Mean	(SD)[a]	Mean	(SD)[a]	Mean	(SD)[a]	Mean	(SD)[a]
PNC[b] (particles/cm^3)	10,900	(7300)	33,900	(4200)	11,000	(7800)	7400	(4300)
$PM_{2.5}$ (μg/m^3)	19.0	(7.9)	10.5	(2.0)	19.7	(14.9)	6.5	(1.3)
BC (μg/m^3)	2.8	(3.4)	2.3	(0.8)	2.9	(2.3)	0.4	(0.3)
CO (ppm)	0.6	(0.9)	0.3	(0.4)	0.4	(0.4)	0.1	(0.1)
Temperature (°C)	31.3	(4.7)	31.7	(6.1)	32.1	(4.8)	25.5	(3.3)
Relative humidity (%)	61.1	(18.9)	43.6	(28.4)	59.7	(16.2)	24.8	(4.0)
Speed (km/hr)	33.3	(26.1)	33.9	(27.4)	40.4	(29.6)	39.5	(30.4)

[a] Arithmetic means and standard deviations.

[b] PNC (>5 nm) was measured by a CPC 3785.

with the results from a previous study (Zhu et al. 2007). Both PNC and BC indicate emissions from combustion sources, which may explain the stronger correlation. However the correlation between PNC and BC was different in terms of engine age and window positions, as shown in Appendix Figure A.4. The correlations were very poor for the buses with windows closed regardless of engine age. But when the windows were open, the correlation coefficients differed with engine ages. The R^2 for the 1990 MY school bus was 0.26, while for the newer bus the R^2 was 0.53. This suggests that the source of air pollutants differed by window position.

Window Position

Figure 1 presents the overall effect of window position on measured air pollutant concentrations. The measurements for the open and closed window positions were integrated across all of the buses. The gray bars represent the window-closed condition and the black bars represent the window-open condition, respectively. The *x*-axis is the measured pollutant concentration on a linear scale and the *y*-axis is the observation frequency as a percentage. In general, all pollutant concentrations show a primary mode with right-skewness. When the windows were closed, the PNC had a primary mode of 8.0×10^3 particles/cm^3, similar to that of the window-open condition. A secondary mode, around $30.0–40.0 \times 10^3$ particles/cm^3 was observed for the window-closed condition, suggesting that closed windows can elevate UFP levels. For the $PM_{2.5}$ concentration profiles, with the windows closed, the primary mode was around 10 µg/m^3 and a secondary mode was around 20 µg/m^3. The primary mode of $PM_{2.5}$ for the window-open condition was at a higher concentration than that for the window-closed condition, suggesting that opening windows might increase the $PM_{2.5}$ levels inside the buses. However, there was a wider spread in the distribution of $PM_{2.5}$ concentrations for the window-closed condition than for the window-open condition. With the windows

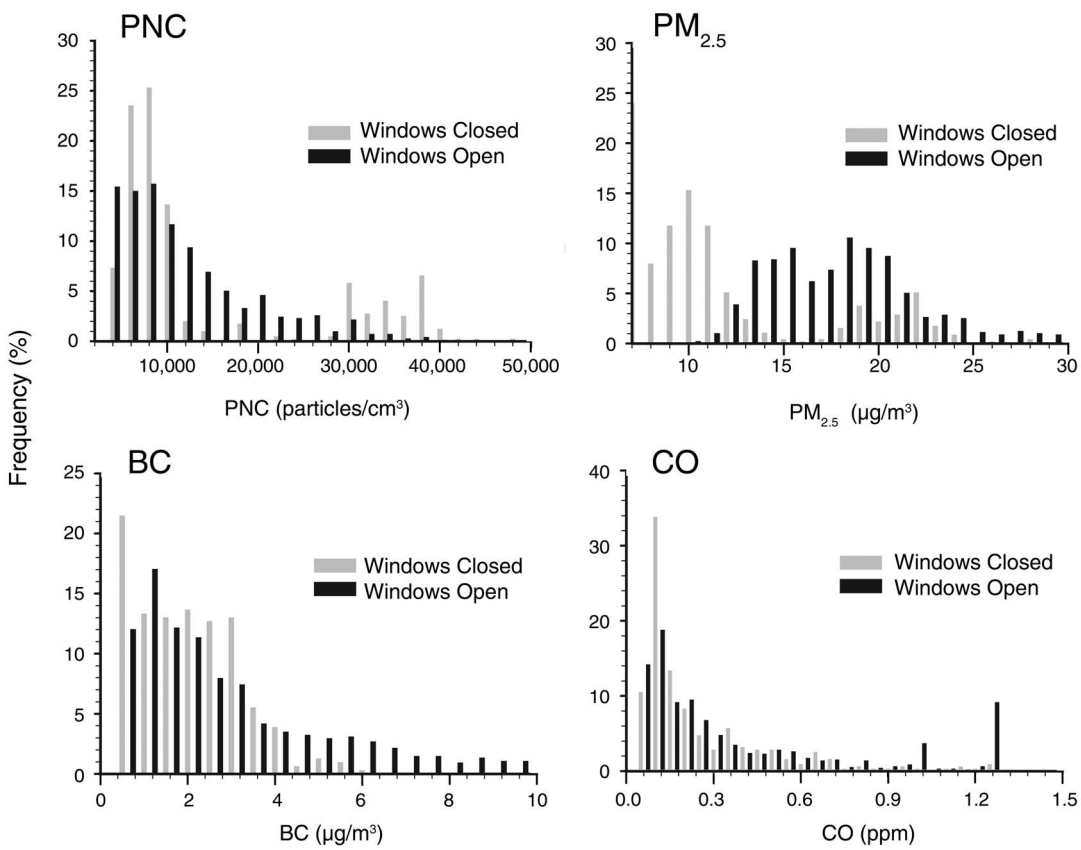

Figure 1. Effect of window position on pollutant concentrations. Histogram of PNC, $PM_{2.5}$, BC, and CO concentrations measured inside school buses.

closed, BC concentrations were < 6 μg/m^3 as opposed to the case with open windows where a small fraction of the BC measurements were > 6 μg/m^3. Window positions had little effect on CO concentrations, as indicated by their concentration profiles.

As shown in Table 5, window position affected in-cabin air pollutant levels differently in the older and newer buses. In the 1990 MY school buses, the PNC concentration was two times higher with the windows closed than with the windows open. In contrast, for 2006 MY buses, the PNC concentration was 33% lower with the windows closed. For other air pollutants such as $PM_{2.5}$, BC, and CO, the concentrations were always higher when the windows were open, regardless of bus age. The in-cabin air composition of school buses was a mixture of self-emissions and ambient air (although no ambient air pollution measurements were performed in this test). When the windows were open, the effects of the ambient air increased presumably because of a higher air exchange between the cabin interior and the surrounding environment (Gong et al. 2009; Ott et al. 2008). Table 5 shows that for the two older buses, the in-cabin PNC was lower in the bus with open windows. However, for the two newer buses, the in-cabin PNC was higher in the bus with open windows, as were other pollutants.

The effects of window position on the UFP size distributions inside buses of different ages are shown in Figure 2. The *x*-axis presents the time when the data were collected, and the *y*-axis is the particle size on a logarithmic scale. The color intensity indicates normalized PNC (dN/dLogDp) for a given size at a given time. The same concentration scale was used for all runs. Figure 2A is the size distribution for an afternoon test in a 2006 MY school bus with closed windows. Figure 2B shows the contour for an afternoon test inside a 1990 MY school bus with closed windows. Figures 2C and D represent a morning and an afternoon run, respectively, in a 1990 MY school bus with open windows.

Figures 2B and D show that the primary mode of particle diameter measured in the 1990 MY school was around 10–20 nm during driving conditions on all routes for both window positions. However, the concentration was higher when the windows were closed. High UFP concentrations were observed during the last 15 minutes of driving when the school buses were on the town route with its high-density traffic. Traffic on surface streets during the rush hours might significantly increase the in-cabin UFP concentrations. In Figure 2D, a notably higher UFP concentration was observed in the first half hour when the school buses picked up the students at the school parking lot and transferred them at the bus transfer station. This was not found in Figure 2B. A possible explanation was that the school

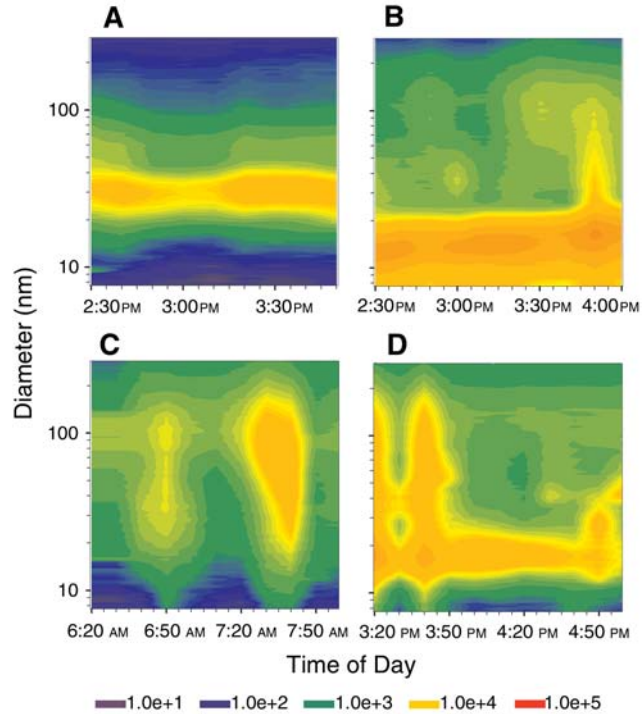

Figure 2. Size distribution of UFP inside school buses under different operating conditions. A: 2006 MY, closed window, afternoon test; **B:** 1990 MY, closed window, afternoon test; **C:** 1990 MY, open window, morning test; and **D:** 1990 MY, open window, afternoon test. The color intensity indicates normalized PNC (dN/dLogDp) for a given size at a given time.

bus with closed windows (Figure 2B) was tested during the 2008 spring break with few surrounding school buses, while the test in Figure 2D was conducted on a normal school day when many school buses were operating nearby at the bus transfer station and at the school parking lot. Thus, in Figure 2D the in-cabin air pollutants were affected by the emissions of surrounding buses. Another difference between the scenarios in Figures 2B and D is that even though the primary modes for both window positions were similar; the concentrations at the primary mode under window-closed conditions were much higher than those under window-open conditions. For the 1990 MY buses, opening the windows helped to dilute the nuclei mode particles (diameter < 30 nm), but increased the numbers of larger particles.

Bus Age

When the window-closed school bus tests were conducted during the spring break, there were few surrounding

vehicles, so the in-cabin air pollutants under window-closed condition mainly came from the bus' self-emissions. As shown in Table 5, under window-closed conditions higher pollutant concentrations were always observed in the older school buses. The average PNC concentrations under window-closed conditions in the old bus were 4.5 times greater than those in the newer bus. The highest 1-minute average PNC, 5.1×10^4 particles/cm^3, was measured in the older school bus under window-closed conditions. In the 1990 MY school bus under window-closed conditions, the average concentrations for $PM_{2.5}$ and BC were 61% and 475% greater, respectively, than those in the newer bus. The different engine positions may contribute to the observations. In the older bus, the engine compartment was located to the right of the driver, inside the bus, increasing the chance for air pollutants emitted from the engine to leak into the cabin. In the newer bus, the engine was under the front hood, so more of the exhaust could escape into the atmosphere. In addition, the airtightness of bus cabins may have an effect on where the pollutants go. Newer buses usually had fewer cracks or leaks in the crankcases and on the bus floors, reducing the chance that air pollutants would leak into the cabin.

School bus self-pollution originates from both the tailpipe and the engine crankcase. Particles from the crankcase were found to be larger than those emitted from tailpipes in previous studies. Hill and colleagues (2005) revealed that the crankcase was a strong source of $PM_{2.5}$ inside school buses. Compared to the newer bus, the older bus with closed windows had a higher concentration of air pollutants, indicating that self-pollution from both sources (crankcase and tailpipe) was greater in the older bus. As tailpipe emissions usually enter a bus cabin through improperly sealed doors and windows, upgrading the door and window seals might be an effective method to reduce the concentration of in-cabin air pollutants in older school buses.

As shown in Figures 2A and B, the UFP size distributions inside school buses with closed windows differed by the age of the bus. The primary mode was approximately 15 nm for the older bus, and about 30 nm for the newer one. A secondary minor mode larger than 30 nm was measured in the in-cabin air of the older bus, but this mode was less obvious inside the newer bus. Higher concentrations of particles < 20 nm inside the older bus suggested that children were exposed to a higher level of small UFP while riding in an older bus than in a newer one.

In addition to bus age, other factors such as engine size, the presence of a turbocharger, maintenance conditions, emission control systems, and ambient levels may affect in-cabin air pollutant concentrations. As shown in Table 3 for Beeville, Texas, the four buses used in the on-road tests were manufactured by the same company, powered by the same engine models, and had similar body sizes. The effect of emission control systems and ambient levels will be discussed later, in the retrofit tests section.

Driving Speed

Appendix Figure A.5 shows the in-cabin air pollutant concentrations at different driving speeds under window-open conditions for all school buses in the on-road tests. The driving speeds were classified into 12 categories and the in-cabin PNCs were averaged for each category. A linear relationship between the average PNC and the driving speed was found with an R^2 value of 0.85. Similar relationships between other pollutant concentrations and the driving speed were also observed, although the correlations were not as strong. The negative slopes suggest that the pollutant levels inside the buses decreased with increasing driving speed. This result may be due to a greater air exchange rate at a higher driving speed. Air exchange rates were estimated with a steady state mass balance for CO_2 with the following assumptions: the outdoor CO_2 was 400 ppm, the CO_2 emission rate of the occupants was about 1000 g/day, and the cabin volume of the school buses was about 40 m^3. Based on occupancy counts and real-time measurements of in-cabin CO_2 concentrations, air exchange rates were estimated to be 37/hr at 32 km/hr, 49/hr at 64 km/hr, and 72/hr at 96 km/hr. Due to the assumptions and simplifications made in this calculation, these values should not be treated as absolute, but should be used only for relative comparisons. These values were comparable to those in the previous studies on the effect of driving speed on average air exchange rate (Ott et al. 2008; Sabin et al. 2005). The larger air exchange rate at higher driving speed helped to dilute the in-cabin air pollutants with the cleaner ambient air. In addition, high speeds usually occurred on rural roads with low traffic density. It should be noted that the negative relationship between pollutant levels and driving speed may apply only to school buses with the windows open that are driving in rural areas.

Route and Operation

PNC varied with respect to different locations and routes, as shown in Appendix Figure A.6. For one bus, the in-cabin PNC ranged from 2.5 to 9.0×10^3 particles/cm^3 along the rural route. When the bus arrived at a bus transfer station, the PNC increased to 35.0×10^3 particles/cm^3, and then dropped slightly to 25.0×10^3 particles/cm^3. Approximately 10 minutes later, a PNC peak was observed when 27 school buses lined up and left the transfer station one-by-one. When the school bus arrived at a school parking lot behind other school buses, another

peak occurred around 30.0×10^3 particles/cm^3. After leaving the school parking lot, the PNC on the town route was around 15.0×10^3 particles/cm^3.

The PNC also varied with different operations, such as starting-up, idling, and driving. Starting-up referred to when the engine was just turned on. Idling occurred when the bus stopped with its engine running. The time between starting-up and idling was defined as driving. Appendix Figure A.6B shows a typical time-series plot of PNC for one starting-up/driving/idling period. A significant increase was observed when the bus was starting up. When driving at a steady speed, the PNC dropped to 25%–30% of the peak value observed earlier. When the school bus was idling, another peak occurred that was 15% lower than during starting-up.

The in-cabin air pollutant concentrations when the school buses were driven at different locations are plotted in Figure 3A. The *x*-axis represents operating conditions and the *y*-axis represents normalized pollutant concentrations, which are the quotients of dividing in-cabin concentration by the ambient concentrations measured in the rural area. To derive the latter, ambient concentrations were measured for about 15 minutes when the bus was parked in the parking lot, located in a rural area and away from major traffic. The data were averaged to get an estimate of the average ambient air concentrations in the rural area studied. These ambient concentrations in the studied rural area were: PNC 8.0×10^3 particles/cm^3, $PM_{2.5}$ 7.6 μg/m^3, BC 2.5 μg/m^3, and CO 0.1 ppm. Higher in-cabin air pollutant levels were observed when the buses were idling at the transfer station and at the school parking lot. Average PNCs were about 210% and 190% of the ambient concentrations measured in the rural area at the school parking lot and at the transfer station, respectively. In comparison, average PNCs on the rural and town routes were about 110% and 130%, respectively. Average $PM_{2.5}$ at the transfer station and at the school parking lot were also higher than the average $PM_{2.5}$ when driving on two routes, although the differences were not as large. CO was much higher at the school parking lot than in other locations. At these two locations, many school buses were idling, waiting to pick up and drop off children. The emissions from these buses could disperse and contaminate each other, deteriorating the air quality inside the tested buses. The PNC inside the tested buses on the town route was 17% higher than that on the rural route, which was likely due to the higher traffic density on the town route. For other air pollutants, the concentrations on these two routes were similar.

The air pollutant concentrations under differing operating conditions are plotted in Figure 3B. As with the single-bus data in Appendix Figure A.6B, the in-cabin air pollutants were higher when the school buses were starting up or idling. This was consistent with a previous study in

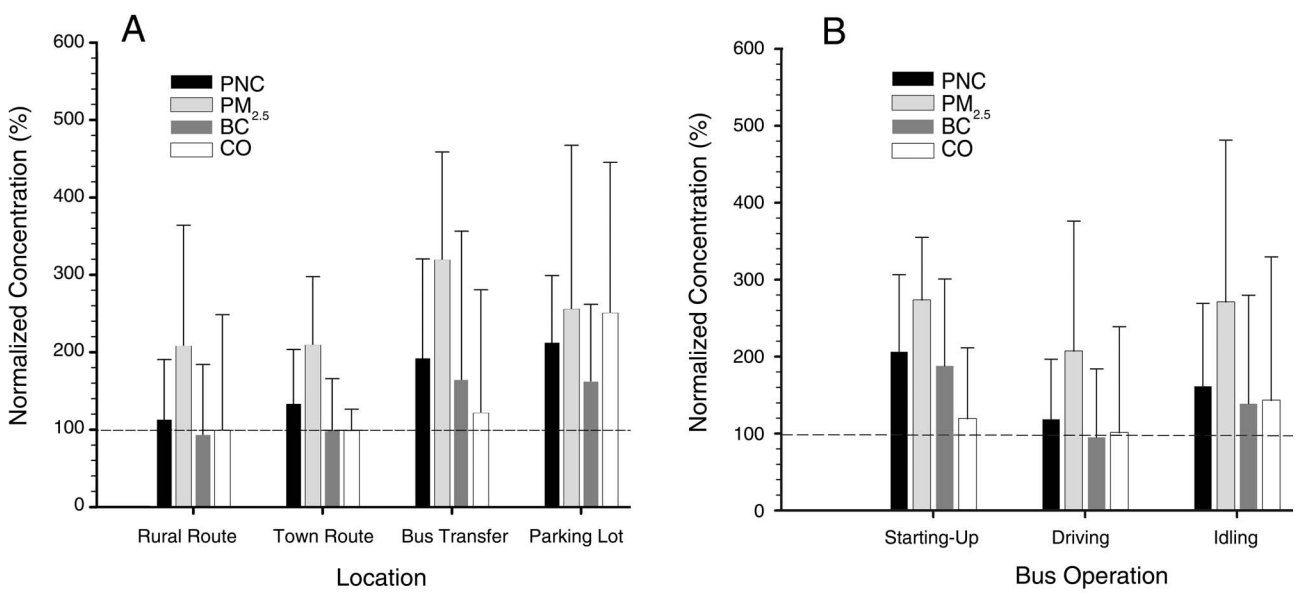

Figure 3. Normalized in-cabin pollutant concentrations on A: different bus routes, and B: under different bus operating conditions. Error bars indicate one standard deviation. Normalized concentrations are the quotients of dividing in-cabin pollutant concentrations by the background ambient concentrations in the rural area, which are as follows: (1) PNC: 8.0×10^3 particles/cm^3, (2) $PM_{2.5}$: 7.6 μg/m^3, (3) PM_{10}: 8.0 μg/m^3, (4) BC: 2.5 μg/m^3, and (5) CO: 0.1 ppm. Horizontal line indicates $y = 1$.

Austin, Texas, in which the mean UFP concentrations were higher during frequent stops than during the cruising mode (Rim et al. 2008). Compared with starting up, idling raised a greater concern about health risks because of its prolonged duration. The next section provides a more detailed discussion about the impacts of idling on PNC in and around school buses.

IDLING TESTS

Compared with regular cruising, air pollutant concentrations were higher at the transfer station and at the school parking lot where many school buses were idling simultaneously as they picked up and dropped off children. This implies that idling school buses may increase air pollutant levels in and around the buses. Diesel emissions from one school bus may penetrate into its own cabin through cracks, doors, and windows. Such exhaust could also enter the cabins of nearby school buses and deteriorate their in-cabin air. In addition, tailpipe emissions from idling school buses can increase air pollutant concentrations in the vicinity of the buses while many school children are waiting to board. To investigate the impact of tailpipe emissions from idling school buses on particle levels in and around school buses, five scenarios were tested in terms of wind direction and bus position (see Appendix Figure A.3). The results can facilitate the assessment of anti-idling practices in reducing children's exposure to particles from school bus emissions.

UFP In and Around Idling School Buses

Figure 4 summarizes the average PNCs under engine-off and engine-on conditions for the upwind air, in-cabin air, and air close to the tailpipes under different scenarios. For each bus and scenario, average concentrations were calculated under engine-off and engine-on conditions. The mean and standard deviation of these nine average concentrations were calculated to present the overall change of particle levels due to engine operation (see Figure 4). Simple sketches are provided to show bus position and wind direction. Shaded boxes indicate the buses in which the measurements were taken. Rays coming from the tailpipes indicate the buses in which the engines were turned on. Detailed time series of PNC under engine-off and engine-on conditions are shown in Appendix Figures A.7–A.9.

As indicated by the small variation of average PNC in the upwind air, the background particle levels were fairly stable, varying between 8.2 and 10.9×10^3 particles/cm^3. For the air close to the tailpipes, average PNCs were at about the same level as the background with the engine off. After the engines were turned on, the PNC close to the tailpipes increased sharply to $112.0–315.7 \times 10^3$ particles/cm^3. Such a sharp increase indicates potentially high exposure to diesel particles for children waiting to board near idling school buses.

The change in average in-cabin UFP number concentrations depended on wind direction and window position. With a perpendicular wind direction, no distinct change was measured after turning on the engines, regardless of window positions (Figure 4A). With a parallel wind direction (Figure 4B), the average in-cabin UFP increased from 11.1 to 16.0×10^3 particles/cm^3 with closed windows and increased from 8.3 to 23.4×10^3 particles/cm^3 with open windows. Such increases were significant at the significance level of 0.05, based on the results of longitudinal regression as described in the Statistical Analysis section. The penetration of tailpipe emissions from other nearby idling buses depended on bus position. The change in the in-cabin UFP of the upwind buses due to engine operation of the downwind buses was negligible (Figure 4C). The penetration of tailpipe emissions from the upwind bus to the downwind bus cabin was more evident. In Figure 4D, after turning on the upwind buses' engines, the average in-cabin UFP increased from 11.5 to 14.0×10^3 particles/cm^3 with closed windows and from 7.3 to 21.3×10^3 particles/cm^3 with open windows. The results of longitudinal regression showed that only the increase with open windows was significant at the significance level of 0.05. When two school buses were idling together (Figures 4E_1 and E_2), no increase of in-cabin UFP was observed in the upwind buses regardless of window positions or in the downwind buses when the windows were closed. However, when the windows were open, the downwind buses had the greatest increase of average in-cabin PNCs, going from 7.6×10^3 particles/cm^3 for the engine-off condition to 36.8×10^3 particles/cm^3 for the engine-on condition.

To indicate the relative impact of tailpipe emissions, the ratios of PNC between engine-on and engine-off conditions for the scenarios are presented in Figure 5. Ratios for the upwind air ranged from 1.0 to 1.1, indicating that the upwind air was not affected by the tailpipe emissions. The ratios for the tailpipe air were the highest, with a maximum around 26.0. For the in-cabin air, ratios varied with wind direction and window position. For closed windows the ratios were between 1.0 and 1.4, smaller than those measured for open windows, which ranged from 1.2 to 5.8. The greatest ratio, 5.8, was found for scenario E_2 in the downwind buses with open windows.

The change of in-cabin particle levels due to engine operation was not uniform across the size range measured.

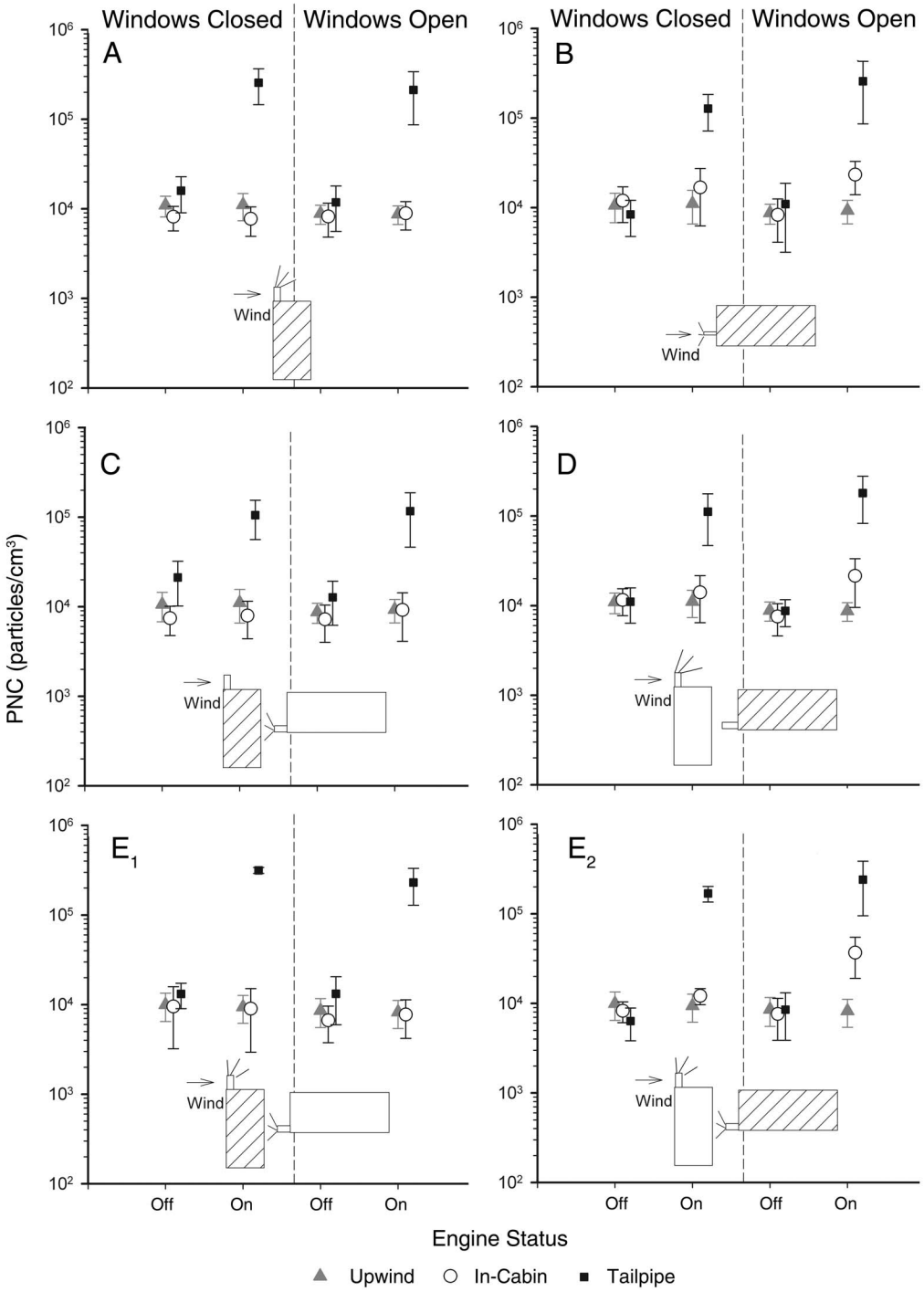

Figure 4. PNC in the upwind air, UFP in the in-cabin air, and PNC in the air close to the tailpipes under different scenarios with different window positions. Shaded boxes indicate the bus in which the measurements were taken. Rays coming from the tailpipes of the boxes (shaded or unshaded) indicate the buses in which the engines were turned on. Error bars indicate one standard deviation. PNC in the upwind air was measured by a CPC 3781 (> 6 nm), UFP of the in-cabin air by an SMPS (7.6–289 nm), and PNC in the air close to the tailpipes by a CPC 3007 (> 10 nm).

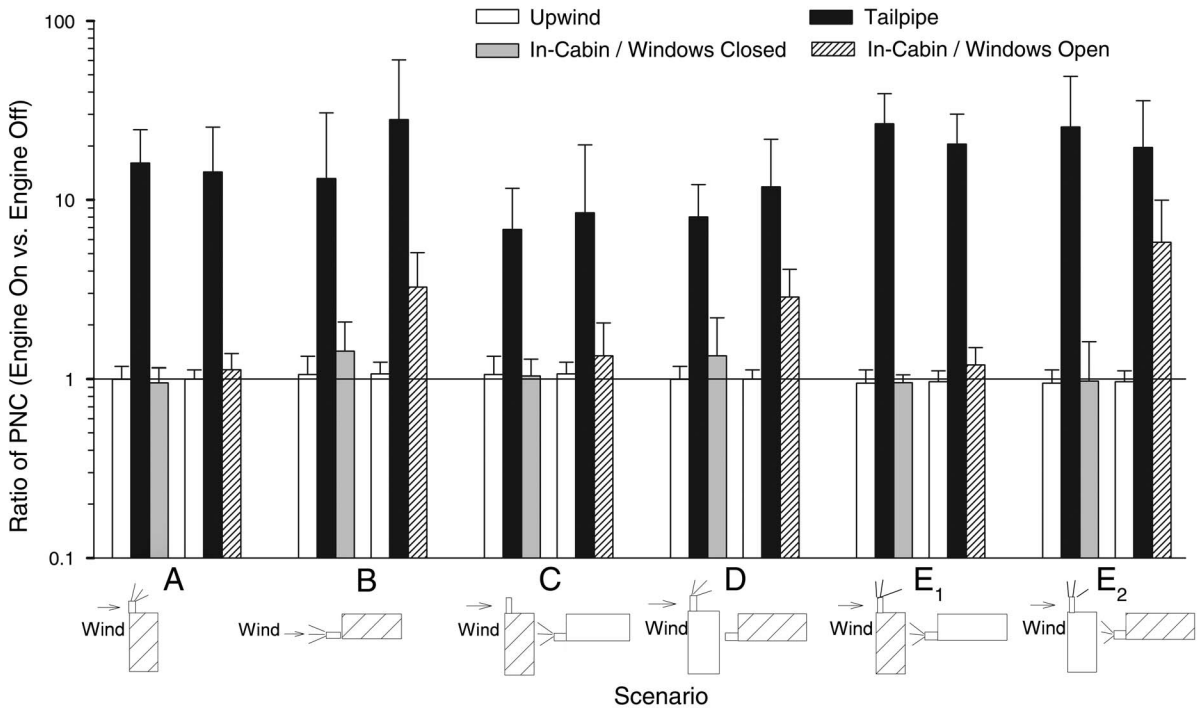

Figure 5. Ratios of PNC between engine-on vs. engine-off conditions under different scenarios. Error bars indicate one standard deviation. PNC in the upwind air was measured by a CPC 3781 (> 6 nm), UFP of the in-cabin air by an SMPS (7.6–289 nm), and PNC in the air close to the tailpipes by a CPC 3007 (> 10 nm).

Figure 6 shows some selected time series of size-segregated UFP inside bus cabins where significant increases of particle levels were observed after turning on the engines (Figures 4B, D, and E_2 under window-open conditions). After turning on the engines, the greatest increase occurred in the particle size range of 10–30 nm. The number concentration of particles in the 30–100 nm range also increased, but with much less magnitude than that of smaller particles. The number concentration of particles with diameters larger than 100 nm stayed the same. As most of the particles emitted by diesel engines are expected to be in the nuclei mode with diameters less than 30 nm (Kittelson 1998), it is not surprising that the greatest increase of in-cabin UFP was in this size range.

Since the penetration of particles differed greatly by particle size, we used longitudinal regression to analyze UFP for each particle size bin. Figure 7 shows the results of longitudinal regression for the particle size distributions inside school bus cabins under different scenarios. Lines represent the average concentration, and shades indicate 95% confidence intervals. Size distributions for the engine-off condition were based on all data collected under all simulated scenarios. Size distributions for the engine-on condition were the prediction of in-cabin UFP after the engines had been running for 15 minutes. Under the engine-off condition, the in-cabin particle-size distribution had a primary mode of 20 nm and a secondary mode of 60 nm. With a perpendicular wind direction, as shown in Figure 7A, the in-cabin UFP size distributions did not change with operation of the engine. With a parallel wind direction (Figure 7B), turning on the engines did not change the modes, but significantly increased the number concentrations of particles with diameters between 10 and 30 nm. The average particle concentration of the primary mode, 20 nm, increased by 110% with windows closed and by 170% with windows open. When the tailpipe emissions came from other buses, as shown in Figures 7C and D, only the downwind buses were affected significantly. Emissions from the upwind buses increased the mean concentrations of the primary mode inside the downwind buses by 50% with the windows closed and by 130% with the windows open, comparable to the increase in Figure 7B. When two buses were idling together, Figures 7E_1 and E_2 show that only the downwind buses had a significant increase of in-cabin UFP. The increase in concentrations of the primary mode was over 500% with the windows open, much higher than those in Figures 7B and D.

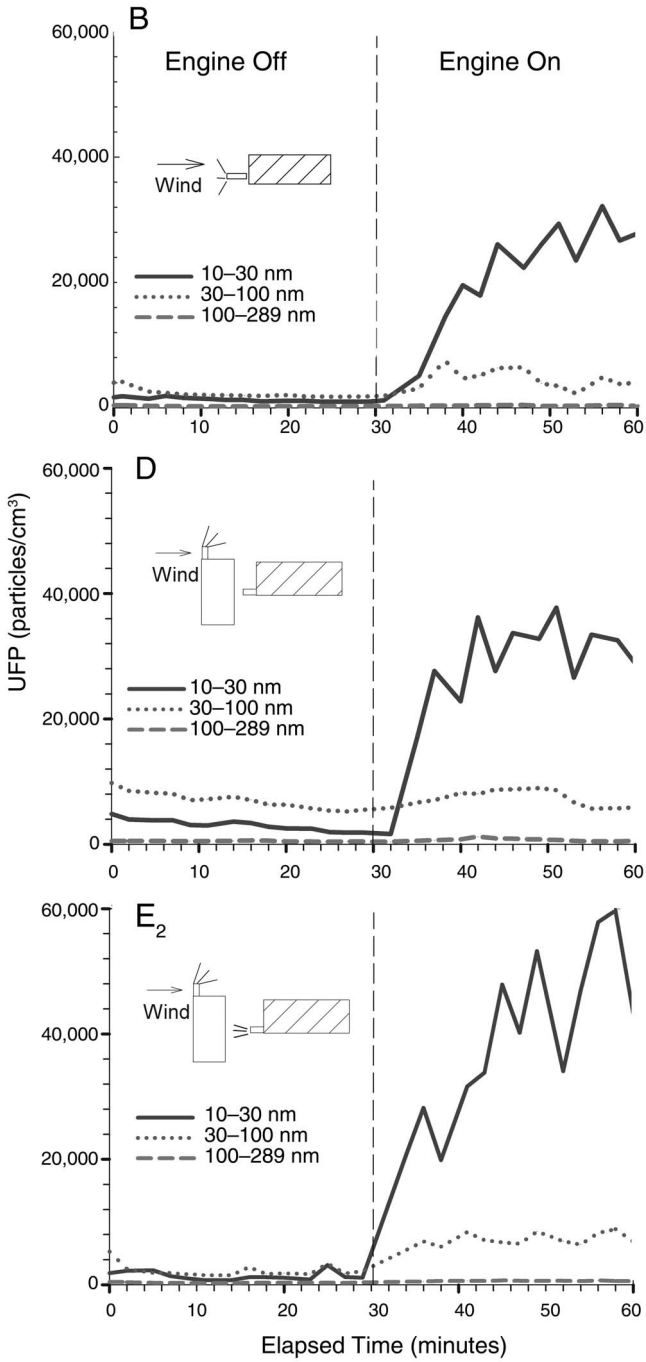

Figure 6. Selected time series of in-cabin UFP of different size ranges under the scenarios that had a significant increase in total PNC (B, D, and E2). UFP number concentration of the in-cabin air was measured by an SMPS (7.6–289 nm).

$PM_{2.5}$ in Idling School Buses

Using the same scenarios as in Figure 4, Figure 8 summarizes $PM_{2.5}$ mass concentrations for the upwind air and in-cabin air. Due to the limitation of available instruments, the air close to the tailpipe was not sampled. Compared with the upwind air, the average $PM_{2.5}$ concentrations in the cabins were lower by 3.0–5.8 μg/m^3 when the windows were closed and by 0.5–4.0 μg/m^3 when the windows were open. Except for the downwind buses with open windows in Figure 8F, of which the average $PM_{2.5}$ increased from 10.8 to 13.1 μg/m^3, turning on the engines did not significantly change in-cabin $PM_{2.5}$ mass concentrations. The results of longitudinal analysis found no significant difference in concentrations of in-cabin $PM_{2.5}$ between engine-off and engine-on conditions for all scenarios. Thus, $PM_{2.5}$ concentration, unlike UFP, was not affected by tailpipe emissions in this study. In-cabin $PM_{2.5}$ was more likely to be governed by regional air pollution than by local tailpipe emissions. Therefore, when assessing the impact of tailpipe emissions on particles in the vicinity of school buses, PNC is a more appropriate index than $PM_{2.5}$.

Previous studies have found that in-cabin $PM_{2.5}$ was dominated by the emissions from the crankcase (Hill et al. 2005; Ireson et al. 2011; Liu et al. 2010). Crankcase contributions were not observed in this study. The discrepancy might be explained by bus age. Older school buses have been observed to have higher crankcase $PM_{2.5}$ emission rates than those of newer buses (Adar et al. 2008; Liu et al. 2010; Zielinska et al. 2008). Newer buses had engines under the front hoods, but the older buses had their engines under the cabin floor. Therefore, the in-cabin air of older school buses is more likely to be contaminated by crankcase emissions. In 2000, the U.S. EPA announced the new PM emission standard for new heavy-duty engines and required the control of crankcase emissions (U.S. EPA 2001). Manufacturers of school buses may therefore have adopted new designs to control crankcase emissions, such as a sealed crankcase oil system or routing of crankcase emissions to the engine air intake system. In idling tests, the majority of tested school buses (L1–L9) were of the 2005 MY, with engines under the front hoods, so $PM_{2.5}$ from crankcase emissions was expected to be less important.

Factors Affecting UFP In and Around Idling School Buses

The PNC measured close to the tailpipes was consistently high for all simulated scenarios, but in-cabin PNC varied over the different scenarios. We used the longitudinal model to estimate the effects of wind direction, window position, emission source (whether or not the emissions were from the buses' own tailpipes or other

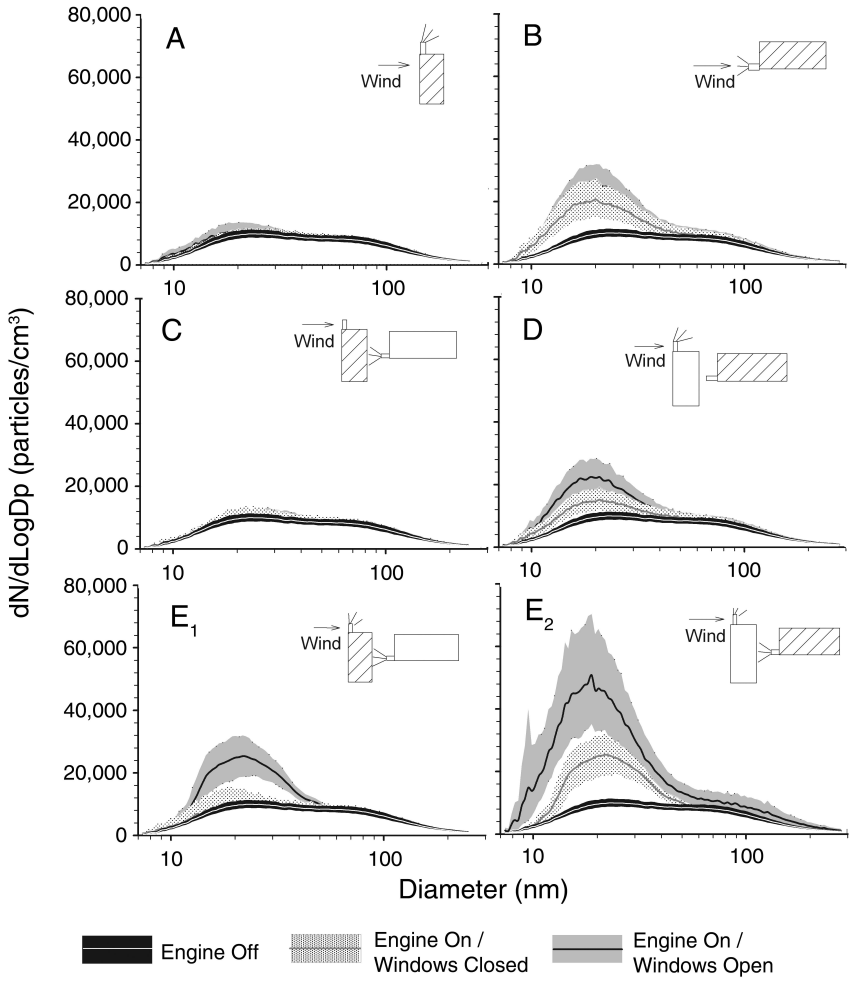

Figure 7. Simulated size distributions of UFP inside school buses under different scenarios. Solid lines indicate mean number concentrations and shaded areas indicate 95% confidence intervals. UFP number concentration of the in-cabin air was measured by an SMPS (7.6–289 nm).

buses' tailpipes), and their interactions on in-cabin particle size distribution. Emission source made no significant difference in particle counts of any size, regardless of wind direction or window position. This indicates that merely shutting down the engine of a school bus is not sufficient to protect children on that bus. Any school bus idling nearby may introduce levels of diesel particles similar to those of the bus' own emissions.

Wind direction and window position were found to affect in-cabin UFP concentrations. The effects of wind direction, window position, and their interaction on the concentration for all particle sizes measured by the SMPS were tested by a multivariable linear regression model. Figure 9 presents the *P* values of their effects. For particles in the size range of 10–30 nm, *P* values were usually less than 0.05, indicating that wind direction played a significant role in introducing nuclei mode particles freshly emitted from tailpipes into buses' cabins, regardless of window position or the emission source. No significant effect was found for window position itself, but the interaction of wind and window was significant. The effect of window position depends on wind direction. Specifically, only when the wind blew from the tailpipe to the hood did open windows result in higher particle concentrations.

Particle Deposition Rates Inside School Buses

After penetrating into bus cabins, particles may be removed by several mechanisms, including outdoor air exchange and surface deposition. Table 7 summarizes air exchange rates and deposition rates for each studied bus.

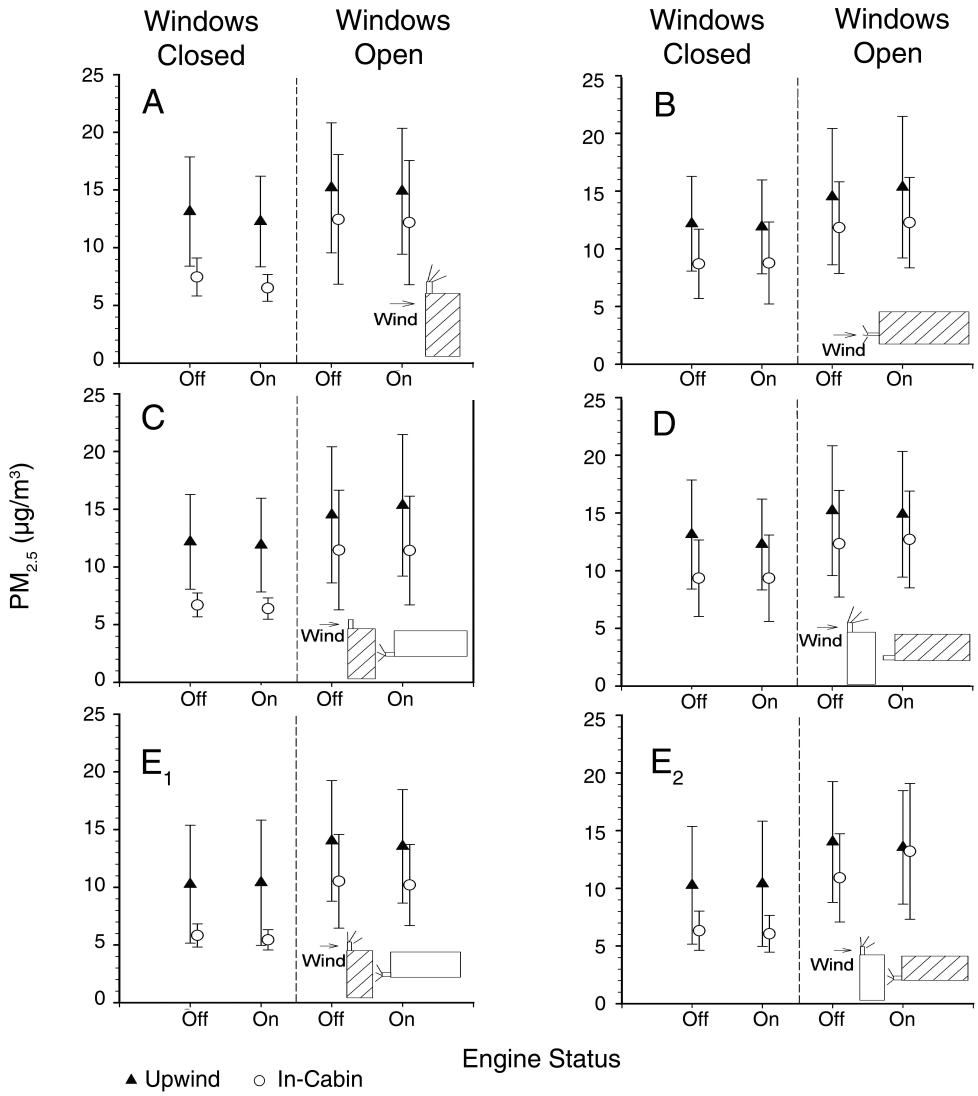

Figure 8. $PM_{2.5}$ mass concentrations in the upwind air and in-cabin air under different scenarios with different window positions. Shaded boxes indicate the bus in which the measurements were taken. Rays coming from the tailpipes of the boxes (shaded or unshaded) indicate the buses in which the engines were turned on. Error bars indicate one standard deviation.

When the windows were closed, air exchange rates in the stationary buses varied between 0.6 and 5.6 per hour. When the windows were open, the air exchange rates were greater, ranging from 11.1 to 34.4 per hour. These results were comparable to air exchange rates in school buses reported in a previous study (Sabin et al. 2005).

Using the method described earlier, deposition rates were calculated based on the PNC of each size (size-resolved deposition) and UFP number concentrations across all sizes (overall deposition). As shown in Table 7, the average deposition rates for UFP in the size range of 7.6–289 nm inside bus cabins were between 1.5 and 5.0 UFP/hr under natural convection conditions. Appendix Figure A.11 presents the size-resolved deposition rates inside school buses compared with the measurements in passenger vehicles and residential indoor environments from two earlier studies (Gong et al. 2009; Zhu et al. 2005). As in those environments, the size-resolved deposition rate inside school bus cabins was a strong function of particle size, with smaller particles having a higher deposition rate because of greater diffusion. The average deposition rate of 10 nm particles was 8.0 particles/hr, 5.5 times higher than that of 100 nm particles. Across the measured size ranges, the deposition rates

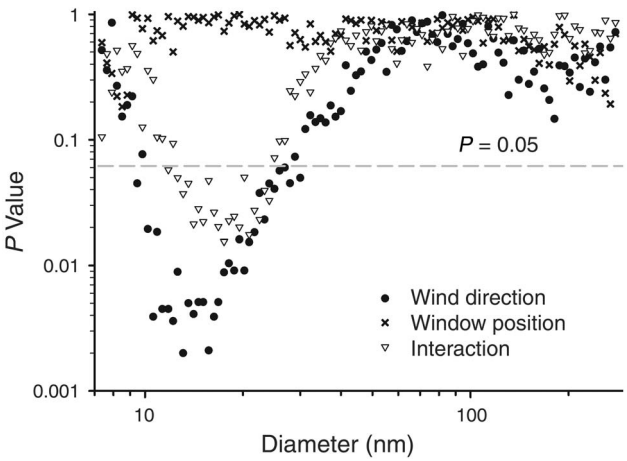

Figure 9. *P* values for the effect of wind, window position, and their interaction. Significance level is $P < 0.05$.

Table 7. Air Exchange Rate and Particle Deposition Rate Inside Stationary School Buses[a]

Bus ID	Air Exchange Rate[b] Windows Closed	Air Exchange Rate[b] Windows Open	Average Deposition Rate
L1	4.6	30.1	2.3
L2	0.6	34.0	3.7
L3	1.7	15.2	2.8
L4	3.9	22.2	4.2
L5	1.7	34.4	4.5
L6	1.1	11.1	1.5
L7	1.9	27.6	3.3
L8	3.5	14.7	5.0
L9	5.6	30.8	3.9

[a] Air exchange and deposition rates are per hour.

[b] Deposition rates were calculated based on UFP number concentration in a size range of 7.6–289 nm measured by an SMPS.

inside school buses were 1/5–1/3 of that inside passenger cars (Gong et al. 2009), and 1.6–2.7 times higher than those in a 26 m^3 residential apartment (Zhu et al. 2005). Gong and colleagues (2009) found higher surface area to volume (S/V) ratios favored higher UFP deposition rates. For the tested school buses, although the in-cabin volume and the interior surface areas varied widely, the S/V ratios of these buses were fairly consistent, ranging from 2.5–2.9/m. These S/V ratios were smaller than the 4.0–8.1/m for passenger vehicles measured by Gong and colleagues (2009), but larger than ~2.0/m for the residential apartment (Zhu et al. 2005), which may explain the difference in UFP deposition rates among these three environments.

RETROFIT TESTS

High concentrations of air pollutants measured in and around school buses suggest that the children riding those buses were exposed to high levels of diesel exhaust (Sabin et al. 2005). To protect children from diesel exhaust emitted by school buses, the U.S. EPA launched a series of programs, including retrofitting old diesel-powered school buses with certified retrofit technologies. In the present study, tests of two retrofit systems (a DOC and a CFS) for diesel-powered school buses evaluated the systems' performance in decreasing PNC and other air pollutants from tailpipe emissions and inside bus cabins.

Performance on Tailpipe Emissions

Tailpipe emissions were measured before and after the installation of retrofit systems while school buses were idling in an open garage. Figure 10 shows that retrofit systems significantly reduced air pollutant concentrations in tailpipe emissions of idling school buses. A DOC reduced PNC by 3%–55% and a CFS by 7%–74%. The combination of a DOC and a CFS performed better than either one alone, with reductions from 20%–94%, with an average of 61%. Reductions were also observed for UFP, $PM_{2.5}$, and BC. Either a DOC or a CFS reduced UFP number concentrations in the size range of 7.6–289 nm by 27%. The combination of both devices reduced PNC levels by 33%. The reduction in $PM_{2.5}$ was 33% for a DOC, 36% for a CFS, and the combination reduced levels to 47%. The individual reduction for BC was 20% and 47% for a DOC and a CFS, respectively. The combination reduced levels by 64%. However, this study did not collect compositional information of tailpipe exhaust. Therefore the physical and chemical transformations of tailpipe exhaust were unclear.

The Wilcoxon signed-rank test was run to test the difference between the pre-retrofit and post-retrofit average concentrations at a significance level of $P < 0.05$. The *P* values of Wilcoxon signed-rank tests are shown in Table 8. Either by using an individual retrofit unit or by combining them, the post-retrofit concentrations of PNC and BC were significantly different from the pre-retrofit concentrations. For UFP number concentrations in the size range of 7.6–289 nm, a significant effect was only observed for the combination of both retrofit systems rather than for either unit individually; although the latter was borderline significant ($P = 0.05$). For $PM_{2.5}$, there was no significant reduction from any retrofit system.

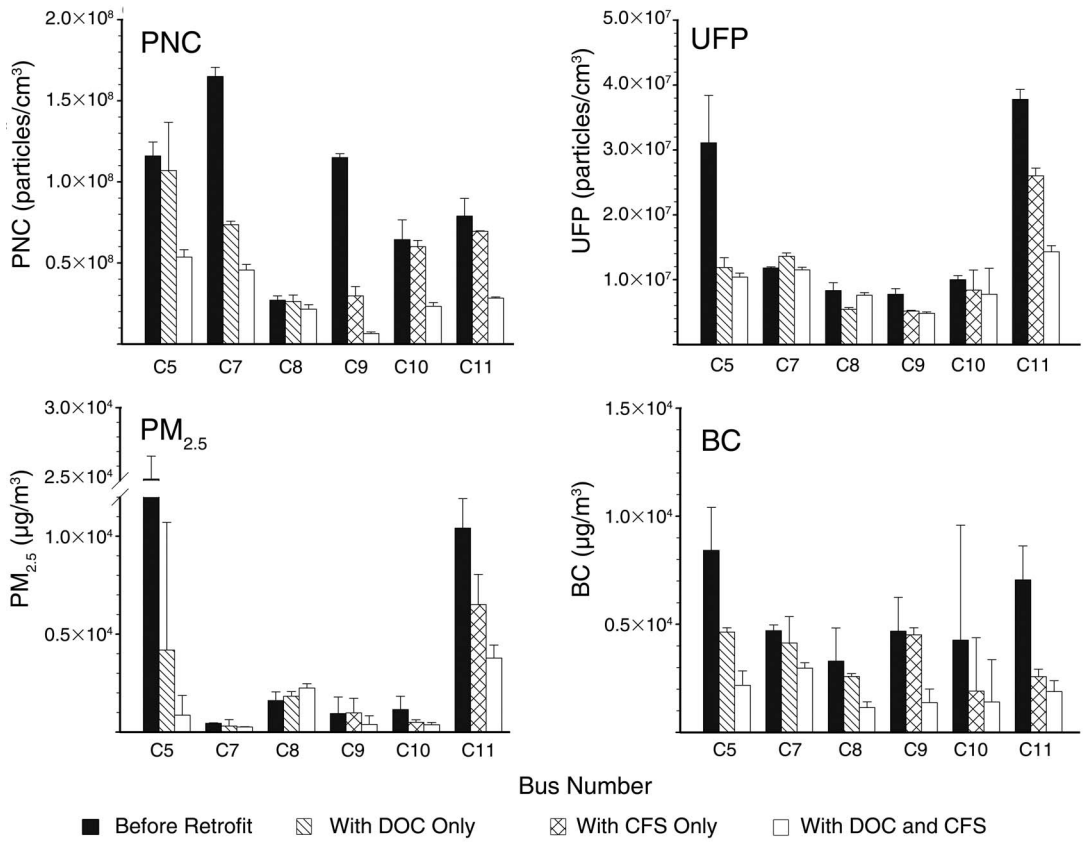

Figure 10. Air pollutant concentrations in school bus tailpipe emissions before and after retrofitting when idling in an open garage. Error bars indicate one standard deviation. PNC was measured by a CPC 3785 (> 5 nm). UFP was measured by an SMPS (7.6–289 nm).

Table 8. *P* Values for the Effect of Retrofit Systems by Wilcoxon Signed-Rank Test

	P Value	
Pollutant[a]	DOC or CFS	DOC + CFS
Tailpipe Air Pollutant Concentrations Before and After Retrofit in the Garage Test		
PNC	0.03	0.03
UFP	0.05	0.03
$PM_{2.5}$	0.17	0.11
BC	0.03	0.03
In-Cabin Air Pollutant Concentrations Before and After Retrofit in the Garage Test[b]		
PNC	0.45	0.91
UFP	0.46	0.25
$PM_{2.5}$	0.60	0.91

[a] PNC was measured by a CPC 3785 in a size range of > 5 nm. UFP of the tailpipe and in-cabin air by an SMPS in a size range of 7.6–289 nm.

[b] In-cabin BC was not measured because only one Aethalometer was available. It was used for tailpipe measurements.

Performance on In-Cabin Air When Idling

The reduction of in-cabin air pollutants by retrofit systems was not as consistent as was observed for tailpipe emissions. In Figure 11, bus C7 was the only one that showed remarkable decreases. PNC, UFP, and $PM_{2.5}$ decreased by 75%, 87%, and 23%, respectively, with a CFS installed, and by 45%, 65%, and 13%, respectively, when both retrofit units were installed. Similar reductions were not observed in the rest of the school buses.

The Wilcoxon signed-rank test was also applied to test the difference of bus-averaged pollutant levels under different retrofit conditions at a significance level of 0.05. The results in Table 7B showed that all *P* values were larger than 0.05. Thus the in-cabin concentrations of PNC, UFP, and $PM_{2.5}$ were not significantly different before and after retrofitting. In addition, as presented in Appendix Figure A.10, the correlation coefficients between in-cabin air and tailpipe emissions were poor (R^2 = 0.01 for PNC and R^2 = 0.08 for $PM_{2.5}$). Thus, the reduction of tailpipe emissions did not lead to pollutant reductions in the bus cabin air. These results are consistent with data presented earlier, in

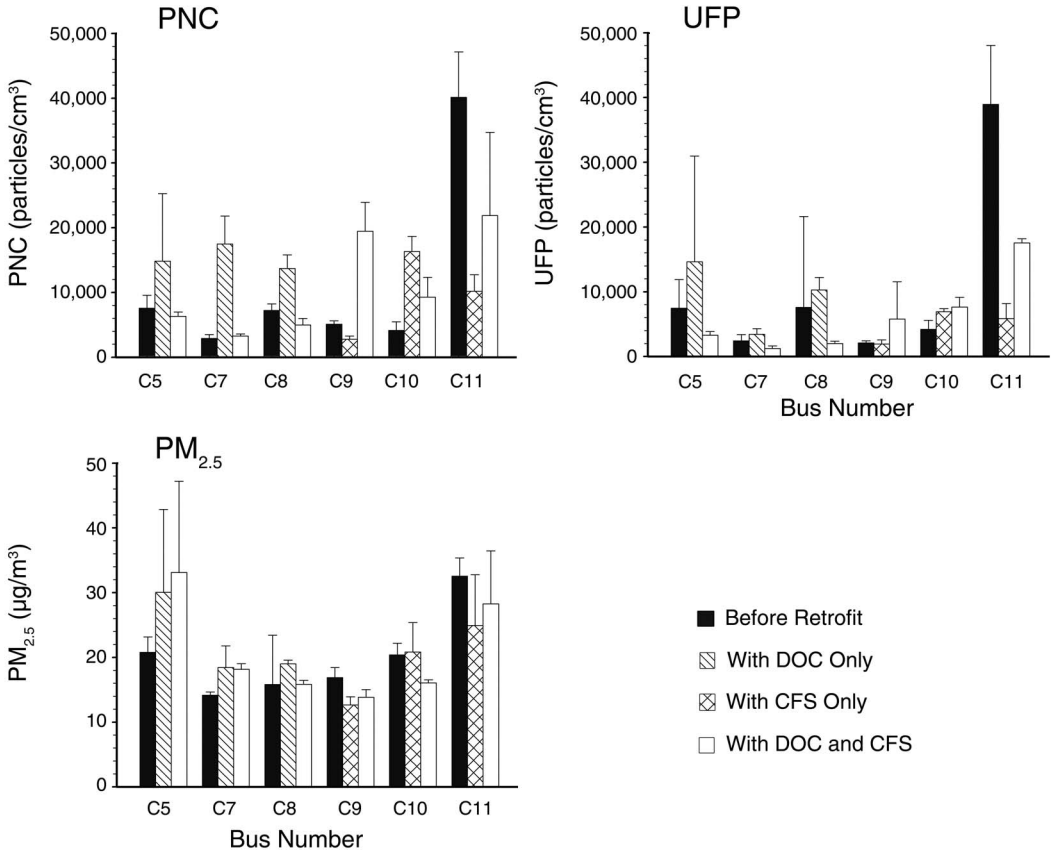

Figure 11. Air pollutant concentrations in the cabins of school buses before and after retrofitting when idling in an open garage. Error bars indicate one standard deviation. PNC was measured by a CPC 3785 (> 5 nm). UFP was measured by an SMPS (7.6–289 nm).

that only under certain circumstances (i.e., parallel wind direction and open windows) did school bus tailpipe emissions have a significant impact on in-cabin UFP levels. Wind directions relative to the buses were not parallel during most of the retrofit tests, therefore little tailpipe exhaust penetration was observed. Crankcase emissions may be more important for elevated in-cabin air pollutant concentrations, especially for $PM_{2.5}$ (Hill et al. 2005). The purpose of the CFS was mainly to control for crankcase emissions. However, contributions from crankcase emissions were not observed in this study because most of school buses tested had their crankcases under the front hoods, which reduced the chances of crankcase emissions leaking into the cabins.

Performance on In-Cabin Air When Driving

Pre- and post-retrofit concentrations of in-cabin air pollutants under different ventilation settings were averaged for each of the six buses and are presented in Figure 12. The differences between the pre- and post-retrofit concentrations were presented as the average efficiency of retrofit system on each air pollutant. Pre-retrofit bus C1 was replaced by bus C2 for post-retrofit testing. With the windows closed, all of the measured pollutants in bus C2 increased by 60%–98% compared with those of bus C1, except for PNC, which decreased by only 9%. With the windows open, PNC and UFP number concentrations in bus C2 decreased by 74% and 79% compared with bus C1, but $PM_{2.5}$ increased by 129%. For the other school buses, which had AC units, the effect of the retrofit systems was also inconsistent. When the AC was off, the post-retrofit concentrations of the measured air pollutants were higher than the pre-retrofit concentrations for buses C3 and C4; however, for the other three buses, lower concentrations of air pollutants were measured after retrofitting. When the AC unit was set to maximum, the increase of the in-cabin air pollutant concentrations ranged from −30% to 30% for PNC, from −13% to 87% for UFP, from −37% to 225% for $PM_{2.5}$, and from −21% to 437% for BC. Data presented in Figure 12 were not corrected for ambient air pollutant

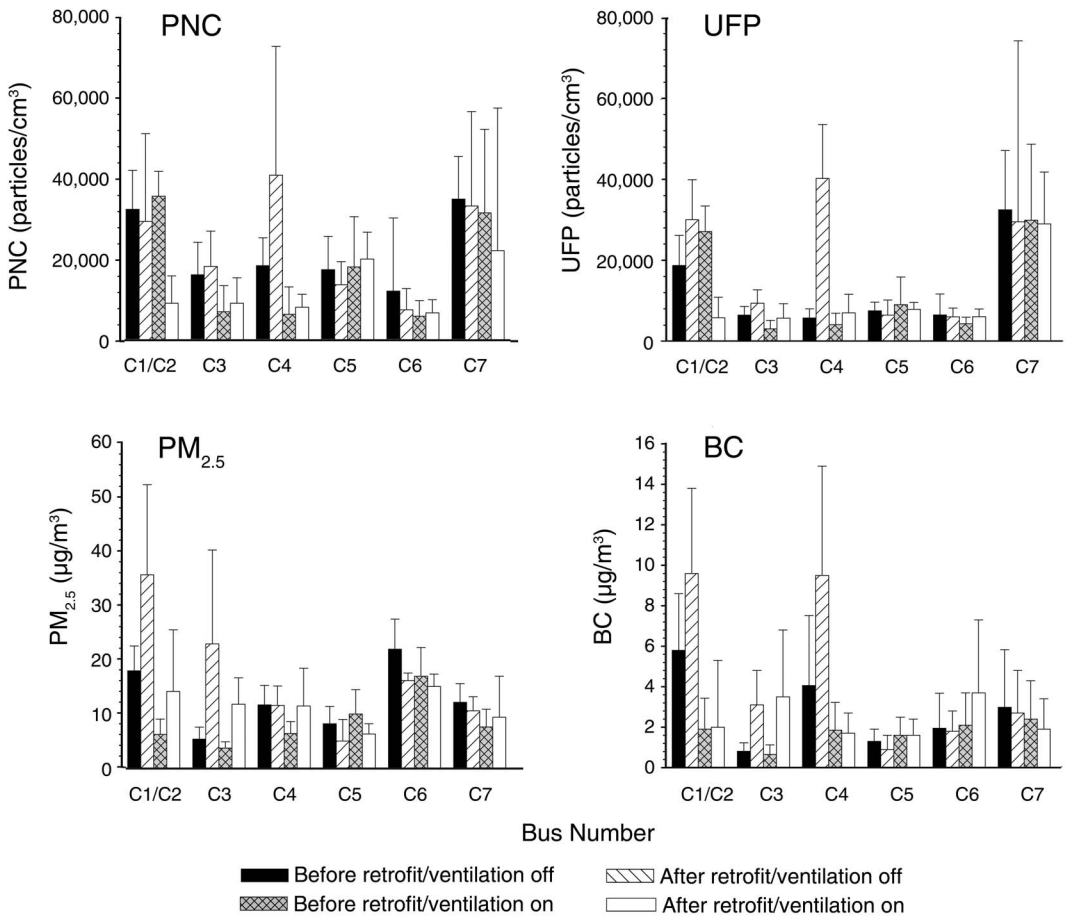

Figure 12. Air pollutant concentrations in the cabins of school buses before and after retrofitting when running. Error bars indicate one standard deviation. Bus C1 was replaced by Bus C2 for post-retrofit measurement. For Bus C1 and Bus C2, ventilation on/off refers to windows open/closed; for the rest, ventilation on/off indicates AC unit on/off. PNC was measured by a CPC 3785 (> 5 nm). UFP was measured by an SMPS (7.6–289 nm).

concentrations. When the impact of pollutant levels in ambient air was taken into account, retrofit systems could not be conclusively linked with a decrease or increase of in-cabin air pollutants (data not shown).

This finding is also consistent with those of previous studies. In the study of Hill and colleagues (2005), pre-retrofit concentrations (28.0–50.0 $\times$ 10^3 particles/cm^3) were not significantly different from post-retrofit concentrations (31.0–38.0 $\times$ 10^3 particles/cm^3). Rim and colleagues (2008) reported that the range of PM variation between repeated tests (54%–163%) was substantially higher than the reduction of UFP by retrofit (–7% to 64%), indicating that any reduction of PNC was probably due to the variability of ambient concentrations rather than to retrofitting. Even though a remarkable decrease of in-cabin PNC levels after retrofitting was reported by Hammond and colleagues (2007) and by Trenbath and colleagues (2009), the overall evidence from these studies and the current study is insufficient to conclude that retrofitting is correlated with lower in-cabin PNCs. In the study by Hammond and colleagues (2007), none of the school buses were tested both prior to and after retrofitting. In addition, the retrofitted buses were 2–6 years newer than the nonretrofitted buses. Thus, it might be bus characteristics rather than retrofit systems that caused the difference between the retrofitted and nonretrofitted buses. In the study of Trenbath and colleagues (2009), the variance of average PNCs among three runs for each bus was too large to draw a statistically solid conclusion. In addition, the meteorological conditions, especially ambient temperature, were different between pre- and post-retrofit runs, which might introduce bias.

Impact of AC Systems and Ambient Air

Before retrofitting, when the AC unit was on with closed windows, the AC unit removed 10%–65% of PNC, 8%–53% of UFP, 23%–45% of $PM_{2.5}$, and 19%–54% of BC, except for bus C5, which had a clogged filter in its AC unit (Figure 12, buses C3–C7). In addition, all of the buses experienced an increase in the ratio of UFP to PNC when the AC unit was on. This indicated that the school bus AC unit contributed to removing large particles from in-cabin air. Previous studies showed that different ventilation modes resulted in different in-cabin exposures to air pollutants (Chan and Chung 2003; Esber et al. 2007; Sabin et al. 2005; Zhu et al. 2007). Zhu and colleagues (2007) found that the maximum reduction of in-cabin UFP from both fan and recirculation was 85% in passenger vehicles. The reduction measured in the present study was slightly lower, which is probably due to the larger cabin of school buses and greater frequency of opening doors to let students on and off the buses.

A typical time series in Figure 13 shows that the in-cabin PNCs tracked the ambient level very well, suggesting that the in-cabin concentration was greatly affected by the ambient level. The results shown in Table 9 suggest that the ventilation conditions and the surrounding air quality rather than tailpipe emissions contributed significantly to the in-cabin air pollutant concentrations. The link between lower in-cabin PNCs with lower ambient air PNCs and the usage of the bus air conditioning systems was also reported by Rim and colleagues (2008).

The results from idling tests showed that UFP emitted by school bus tailpipes penetrated significantly into bus cabins only when the wind blew from the tailpipe of a bus toward its hood. Otherwise, there was no significant penetration of tailpipe emissions into school bus cabins. Based on the increase of PNCs for the in-cabin air and air close to the tailpipes between engine-on and engine-off conditions, as shown in Figure 4, a conservative estimate was made of the contribution from tailpipe emissions to in-cabin UFP concentrations. Only about 0.001%–0.069% of particles in tailpipe emissions entered into bus cabins. This result was somewhat less than that of Behrentz and colleagues (2004), who used SF_6 as a tracer gas and estimated that self-pollution ranged from 0.01% to 0.29%, with the majority between 0.01% and 0.04% when school buses were being driven on actual roads. Therefore, even though the retrofit technologies removed up to 50% of UFP from tailpipe emissions, the benefits of particle reduction were less obvious in the bus cabin. This may explain why no significant change was observed for in-cabin PNCs before and after retrofitting in this study. However, idling tests found tailpipe emissions from idling school buses increased PNCs by a factor of up to 26.0 for the air close to the tailpipes, and by a factor of 1.2–5.8 for the in-cabin air of nearby school buses under certain conditions. Therefore retrofit technologies may reduce children's exposure to particles while waiting at bus transfer stations and in school parking lots, and may contribute to lower levels of air pollutants inside other nearby school buses and vehicles that are on the downwind side of idling school buses.

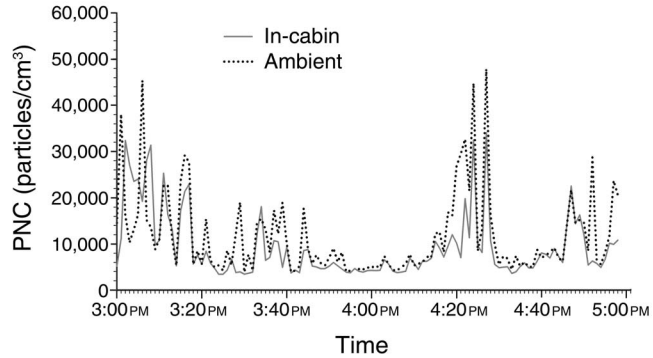

Figure 13. Typical time series of in-cabin and ambient PNC when driving on roads. PNC was measured by a CPC 3785 (> 5 nm).

Table 9. Effects of Retrofit, AC Unit Operation on In-Cabin Air Pollutant Concentrations and Correlation Coefficients with Ambient Concentrations When Driving on Roads

Pollutant	*P* Value[a]		Correlation Coefficient with Ambient Concentration[b] (*P* Value)
	Effect of Retrofit	Effect of AC	
PNC[c]	0.47	0.02	0.65 (<0.001)
UFP[c]	0.43	0.04	0.35 (0.009)
$PM_{2.5}$	0.31	0.02	0.63 (<0.001)
BC	0.47	0.32	n/a[d]

[a] Calculated by the signed rank test.

[b] Spearman correlation coefficients between in-cabin and ambient air pollutant concentrations.

[c] PNC (>5 nm) was measured by a CPC 3785; UFP was measured by an SMPS in a size range of 7.6–289 nm.

[d] BC ambient concentrations were not measured because only one aethalometer was available.

HEPA AIR PURIFIER TESTS

Because no unequivocal reduction in in-cabin air pollutants was observed after retrofitting, in-cabin filtration

that worked directly on in-cabin air was investigated to provide an alternative method to reduce in-cabin particle levels. Air purifiers have been widely used in the United States to remove indoor airborne particles (Shaughnessy and Sextro 2006). A HEPA filter type of air purifier was chosen for this study: it had been shown to be the most efficient air cleaner to reduce particle levels (Offermann et al. 1985), whereas ionizers had been found to generate ozone and organic compounds (Waring et al. 2008). Air purifiers with HEPA filters were employed in four of the school buses in the present study while they were being driven on their actual routes. A typical time series of in-cabin PNC (using one or two HEPA air purifiers in a school bus with the AC unit off) is presented in Figure 14. The average I/O ratios of PNCs after turning on one or two HEPA air purifiers are also indicated. One air purifier reduced the I/O ratios of PNC from 0.82 to 0.55. When two air purifiers were used, the I/O ratio was further reduced to 0.4. When the air purifiers were turned off again, the I/O ratio increased to 0.83.

Table 10 summarizes the PNC and $PM_{2.5}$ mass concentrations for the four school buses under different conditions for both in-cabin air and ambient air. Figure 15 summarizes the effects of HEPA air purifiers on the I/O

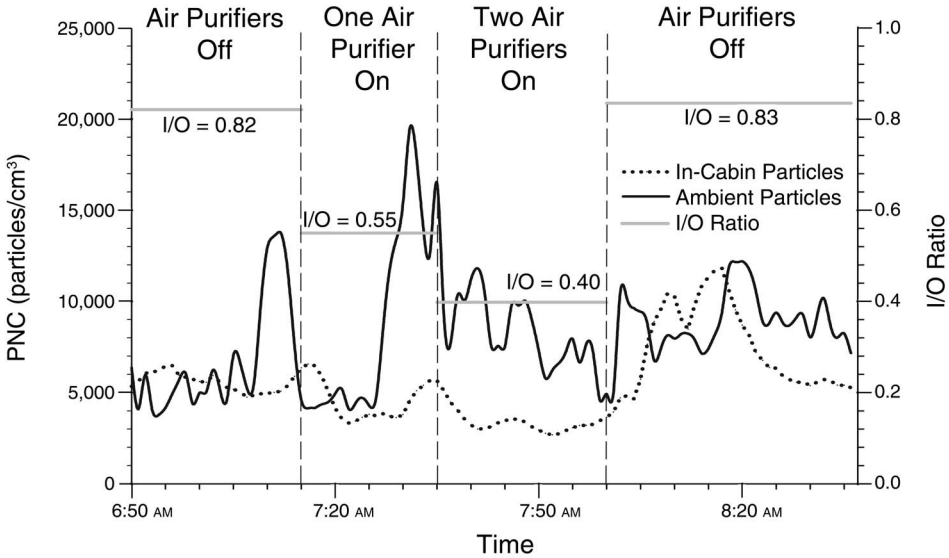

Figure 14. Time series of PNC inside a school bus using one to two HEPA air purifiers under the AC off condition. PNC was measured by a CPC 3785 (> 5 nm).

Table 10. Summary of In-Cabin and Ambient PNC and $PM_{2.5}$ in HEPA Air Purifier Tests

	No HEPA Air Purifier		One HEPA Air Purifier		Two HEPA Air Purifiers	
	In-Cabins Mean (SD)	Ambient Mean (SD)	In-Cabin Mean (SD)	Ambient Mean (SD)	In-Cabin Mean (SD)	Ambient Mean (SD)
PNC (particles/cm³)[a]						
AC Off	15,100 (5,700)	15,200 (8,000)	11,600 (10,900)	21,900 (19,300)	5,100 (4,200)	15,500 (12,300)
AC On	30,800 (4,500)	35,500 (8,000)	9,300 (7,900)	25,100 (20,200)	3,100 (600)	14,800 (3,200)
$PM_{2.5}$ (μg/m³)						
AC Off	14.3 (12.6)	19.2 (12.4)	9.7 (4.3)	21.5 (7.1)	7.1 (3.0)	17.5 (7.2)
AC On	8.7 (3.2)	13.9 (3.8)	6.6 (4.4)	13.2 (6.2)	5.5 (3.7)	12.4 (1.8)

[a] PNC (>5 nm) was measured by a CPC 3785.

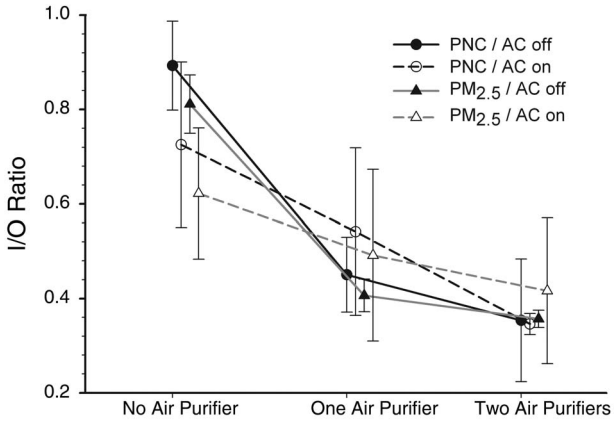

Figure 15. Effect of HEPA air purifier on I/O ratios of PNC and $PM_{2.5}$ under different ventilation settings. Error bars indicate one standard deviation. PNC was measured by a CPC 3785 (> 5 nm).

ratios of PNC and $PM_{2.5}$ under different AC unit settings. When the AC unit was off, an air purifier reduced I/O ratios from 89% to 45% for PNC and from 81% to 41% for $PM_{2.5}$. A second air purifier decreased I/O ratios further to 35% for PNC and to 36% for $PM_{2.5}$. When no air purifier was employed, the operation of the AC unit reduced the I/O ratio to 73% for PNC and to 62% for $PM_{2.5}$. However, when an air purifier was used, there were no significant differences in the I/O ratios of PNC and $PM_{2.5}$ between AC on and AC off settings. One air purifier removed about one half of in-cabin PNC and $PM_{2.5}$, which was over 1.5 times more efficient than the AC unit.

CONCLUSIONS

From an exposure assessment perspective, school buses are important but understudied microenvironments where children congregate and spend a substantial proportion of their time. The purpose of this study was to identify conditions under which children are likely to be exposed to high levels of UFP in and around school buses. To study these exposures, a total of 24 school buses were employed for measurements of air pollutant concentrations (1) inside school buses on their regular pick-up/drop-off routes and (2) in and around idling school buses. The performance of two retrofit systems was evaluated for their effectiveness in reducing levels of UFP and other air pollutants from tailpipe emissions and inside bus cabins. The effectiveness of HEPA air purifiers on reducing in-cabin particle levels was tested to provide an alternative strategy for reducing children's exposure to air pollutants.

When driving on roads, in-cabin PNCs ranged from 7.4 to 34.0×10^3 particles/cm^3, which was higher than typical ambient backgrounds levels. The average air pollutant concentrations inside school buses were dependent on window position, engine age, driving speed, route and location, and operating conditions. With closed windows, the average PNC was 4.5 times higher in the 1990 MY school buses than in the 2006 MY school buses. However, under window-open conditions, the in-cabin PNCs of both older and newer buses were approximately the same as those in the ambient air. For $PM_{2.5}$, BC, and CO, the in-cabin concentrations were lower when the windows were closed than when the windows were open. The concentrations of all air pollutants decreased with increasing driving speeds under window-open conditions in the rural area. While idling at bus transfer stations and in school parking lots, the in-cabin concentrations of all measured air pollutants were higher than those measured when the bus was driving on roads. Many school buses were idling together at these locations, which likely contributed to the higher pollutant concentrations. When driving on busy urban surface streets, the in-cabin PNC was 17% higher compared to driving on rural roads, which was probably due to higher traffic density on the busy surface streets. In terms of bus operating conditions, in-cabin PNC was higher during starting-up and idling than during driving.

Tailpipe emissions from the idling school buses significantly increased PNC close to the tailpipes under all simulated scenarios, by a factor of up to 26.0. After turning on the engines, the PNC near school buses increased sharply from the background level of about 11.8×10^3 particles/cm^3 to an average of 193.7×10^3 particles/cm^3. Tailpipe emissions from idling school buses also had important impacts on in-cabin PNC under certain conditions in terms of wind direction and window position. When the wind carried the emissions from the tailpipe toward the front of the bus, in-cabin UFP levels (7.6–289 nm) increased significantly by a factor of 1.2–5.8, with the greatest increase occurring in the 10–30 nm size range. No significant change of in-cabin $PM_{2.5}$ mass concentration was observed with and without tailpipe emissions, regardless of wind direction or window position, indicating that $PM_{2.5}$ may be insufficient for assessing the exposures to tailpipe emissions from idling school buses. The deposition rates for UFP (7.6–289 nm) varied between 1.5 and 5.0 per hour across the tested school buses under natural convection conditions, lower than those of passenger cars but higher than those of indoor environments.

Retrofit systems employing both a DOC and a CFS significantly reduced PNC tailpipe emissions from idling school buses by 20%–94%, with an average of 61%. Thus, retrofitting can reduce the contribution of school bus emissions to ambient air pollutant levels, and thus can decrease children's exposure to diesel exhaust when they are waiting at

bus transfer stations or school parking lots. However, no unequivocal decrease was observed for in-cabin air pollutants when idling and when being driven on roads. The dominant source of in-cabin UFP might not be the tailpipe emissions of the same bus, of which only less than 0.069% was found to enter the bus cabin. The operation of an AC unit and the air pollutant levels in the surrounding ambient air played more important roles in determining the in-cabin air quality of school buses than did retrofit technologies. An alternative method that worked directly on in-cabin air, the use of a HEPA air purifier, was found to remove in-cabin particles by up to 50%.

IMPLICATIONS OF FINDINGS

The findings of this study have wide-ranging policy implications on transportation planning and emission control strategy. In-cabin PNC was higher on the busy surface streets than on the rural roads; it was also higher for starting-up and idling than for driving. School bus routes with longer segments of busy roads and more frequent stops at traffic lights or stop signs would likely increase children's UFP exposure levels.

High UFP concentrations measured in and around idling school buses indicate an exposure hotspot for children. The U.S. EPA's National Idle-Reduction Campaign suggests that school bus drivers should turn off the engine while waiting at bus transfer stations and at school parking lots. Doing so can reduce children's exposure to UFP. Local meteorological conditions should be considered when designing any area where buses will wait to pick up or drop off children, so that for the majority of its operating time, no school bus is parked on the downwind side of tailpipe emissions from itself or any other buses. If such parking arrangements are not feasible, school bus windows should be closed while idling.

Retrofit systems greatly reduced tailpipe emissions, which helped reduce the exposure of children who were waiting to board or were riding in school buses downwind of an exhaust plume. However, retrofitting by itself does not satisfactorily protect children from in-cabin particle exposures. Technologies such as in-cabin filtration that work directly on in-cabin air offer promise in providing additional protection.

One limitation of this study was the lack of control over school bus characteristics. Although a great effort was made to select the school buses that best represented the available school bus fleet, the population of school buses that we had access to was small compared with the school buses currently in use across the country. This limits the generalization of the findings of this study. Second, in the first part of this study, the on-road test, we did not monitor the ambient air pollutant levels, which is an important contributor to the in-cabin air pollutants. Therefore, the factors that affected the in-cabin air pollutant concentrations may be confounded by the pollutant concentrations in the surrounding air and should be interpreted carefully.

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REFERENCES

Adar SD, Davey M, Sullivan JR, Compher M, Szpiro A, Liu LJS. 2008. Predicting airborne particle levels aboard Washington state school buses. Atmos Environ 42(33):7590–7599.

Alessandrini F, Schulz H, Takenaka S, Lentner B, Karg E, Behrendt H, et al. 2006. Effects of ultrafine carbon particle inhalation on allergic inflammation of the lung. J Allergy Clin Immunol 117(4):824–830.

Alexeeff SE, Coull BA, Gryparis A, Suh H, Sparrow D, Vokonas PS, et al. 2011. Medium-term exposure to traffic-related air pollution and markers of inflammation and endothelial function. Environ Health Perspect 119(4):481–486.

Behrentz E, Fitz DR, Pankratz DV, Sabin LD, Colome SD, Fruin SA, et al. 2004. Measuring self-pollution in school buses using a tracer gas technique. Atmos Environ 38(23):3735–3746.

Behrentz E, Sabin LD, Winer AM, Fitz DR, Pankratz DV, Colome SD, et al. 2005. Relative importance of school bus-related microenvironments to children's pollutant exposure. J Air Waste Manage Assoc 55(10):1418–1430.

Bennett WD, Zeman KL. 1998. Deposition of fine particles in children spontaneously breathing at rest. Inhal Toxicol 10(9):831–842.

Boldo E, Medina S, LeTertre A, Hurley F, Mucke HG, Ballester F, et al. 2006. Apheis: Health impact assessment of long-term exposure to $PM_{2.5}$ in 23 European cities. Eur J Epidemiol 21(6):449–458.

Brunekreef B, Beelen R, Hoek G, Schouten L, Bausch-Goldbohm S, Fischer P, Armstrong B, Hughes E, Jerrett M, van den Brandt P. 2009. Effects of Long-Term Exposure to Traffic-Related Air Pollution on Respiratory and Cardiovascular Mortality in the Netherlands: The NLCS-AIR Study. Research Report 139. Boston, MA:Health Effects Institute.

Brunekreef B, Janssen NAH, deHartog J, Harssema H, Knape M, vanVliet P. 1997. Air pollution from truck traffic and lung function in children living near motorways. Epidemiology 8(3):298–303.

Chan AT, Chung MW. 2003. Indoor-outdoor air quality relationships in vehicle: Effect of driving environment and ventilation modes. Atmos Environ 37(27):3795–3808.

Cheng Y-H, Liu Z-S, Chen C-C. 2010. On-road measurements of ultrafine particle concentration profiles and their size distributions inside the longest highway tunnel in Southeast Asia. Atmos Environ 44(6):763–772.

Cyrys J, Pitz M, Heinrich J, Wichmann HE, Peters A. 2008. Spatial and temporal variation of particle number concentration in Augsburg, Germany. Sci Total Environ 401(1–3):168–175.

de Kok T, Driece HAL, Hogervorst JGF, Briede JJ. 2006. Toxicological assessment of ambient and traffic-related particulate matter: A review of recent studies. Mutat Res 613(2–3):103–122.

Delfino RJ, Sioutas C, Malik S. 2005. Potential role of ultrafine particles in associations between airborne particle mass and cardiovascular health. Environ Health Perspect 113(8):934–946.

Donaldson Company. 2006. Eliminate crankcase emissions and improve in-cab air quality. Available: *www.donaldson.com/en/exhaust/support/datalibrary/002421.pdf* [accessed 1 May 2012].

Elder A, Gelein R, Silva V, Feikert T, Opanashuk L, Carter J, et al. 2006. Translocation of inhaled ultrafine manganese oxide particles to the central nervous system. Environ Health Perspect 114(8):1172–1178.

Esber LA, El-Fadel M, Nuwayhid I, Saliba N. 2007. The effect of different ventilation modes on in-vehicle carbon monoxide exposure. Atmos Environ 41(17):3644–3657.

Ferin J, Oberdorster G, Penney DP, Soderholm SC, Gelein R, Piper HC. 1990. Increased pulmonary toxicity of ultrafine particles? I. Particle clearance, translocation, morphology. J Aerosol Sci 21:384–387.

Frampton MW, Stewart JC, Oberdorster G, Morrow PE, Chalupa D, Pietropaoli AP, et al. 2006. Inhalation of ultrafine particles alters blood leukocyte expression of adhesion molecules in humans. Environ Health Perspect 114(1):51–58.

Fruin S, Westerdahl D, Sax T, Sioutas C, Fine PM. 2008. Measurements and predictors of on-road ultrafine particle concentrations and associated pollutants in Los Angeles. Atmos Environ 42(2):207–219.

Gong L, Xu B, Zhu Y. 2009. Ultrafine particles deposition inside passenger vehicles. Aerosol Sci Tech 43(6):544–553.

Hammond DM, Lalor MM, Jones SL. 2007. In-vehicle measurement of particle number concentrations on school buses equipped with diesel retrofits. Water Air Soil Poll 179:217–225.

Hill LB, Zimmerman NB, Gooch J. 2005. A Multi-City Investigation of the Effectiveness of Retrofit Emissions Controls in Reducing Exposures to Particulate Matter in School Buses. Boston, MA:Clean Air Task Force.

Hinds WC. 2010. Cardiovascular health effects of fine and ultrafine particles during freeway travel. Available: *www.arb.ca.gov/research/apr/past/04-324.pdf* [accessed 18 July 2011].

Ireson RG, Ondov JM, Zielinska B, Weaver CS, Easter MD, Lawson DR, et al. 2011. Measuring in-cabin school bus tailpipe and crankcase $PM_{2.5}$: A new dual tracer method. J Air Waste Manage Assoc 61(5):494–503.

Jamriska M, Morawska L, Mergersen K. 2008. The effect of temperature and humidity on size segregated traffic exhaust particle emissions. Atmos Environ 42(10):2369–2382.

Kittelson DB. 1998. Engines and nanoparticles: A review. J Aerosol Sci 29(5–6):575–588.

Knibbs LD, de Dear RJ. 2010. Exposure to ultrafine particles and $PM_{2.5}$ in four Sydney transport modes. Atmos Environ 44(26):3224–3227.

Lin S, Munsie JP, Hwang SA, Fitzgerald E, Cayo MR. 2002. Childhood asthma hospitalization and residential exposure to state route traffic. Environ Res 88(2):73–81.

Liu LJS, Phuleria HC, Webber W, Davey M, Lawson DR, Ireson RG, et al. 2010. Quantification of self pollution from two diesel school buses using three independent methods. Atmos Environ 44(28):3422–3431.

Marshall JD, Behrentz E. 2005. Vehicle self-pollution intake fraction: Children's exposure to school bus emissions. Environ Sci Tech 39(8):2559–2563.

McCreanor J, Cullinan P, Nieuwenhuijsen MJ, Stewart-Evans J, Malliarou E, Jarup L, et al. 2007. Respiratory effects of exposure to diesel traffic in persons with asthma. New Engl J Med 357(23):2348–2358.

McCubbin DR, Delucchi MA. 1999. The health costs of motor-vehicle-related air pollution. J Transp Econ Policy 33:253–286.

Oberdörster G. 2001. Pulmonary effects of inhaled ultrafine particles. Int Arch Occup Environ Health 74(1):1–8.

Offermann FJ, Sextro RG, Fisk WJ, Grimsrud DT, Nazaroff WW, Nero AV, et al. 1985. Control of respirable particles in indoor air with portable air cleaners. Atmos Environ (1967) 19(11):1761–1771.

Ott W, Klepeis N, Switzer P. 2008. Air change rates of motor vehicles and in-vehicle pollutant concentrations from secondhand smoke. J Expo Sci Env Epid 18(3):312–325.

Peters A, von Klot S, Heier M, Trentinaglia I, Hormann A, Wichmann HE, et al. 2004. Exposure to traffic and the onset of myocardial infarction. New Engl J Med 351(17): 1721–1730.

Peters A, Wichmann HE, Tuch T, Heinrich J, Heyder J. 1997. Respiratory effects are associated with the number of ultrafine particles. Am J Respir Crit Care Med 155(4):1376–1383.

Rim D, Siegel J, Spinhirne J, Webb A, McDonald-Buller E. 2008. Characteristics of cabin air quality in school buses in Central Texas. Atmos Environ 42(26):6453–6464.

Ronkko T, Virtanen A, Vaaraslahti K, Keskinen J, Pirjola L, Lappi M. 2006. Effect of dilution conditions and driving parameters on nucleation mode particles in diesel exhaust: Laboratory and on-road study. Atmos Environ 40(16): 2893–2901.

Sabin LD, Behrentz E, Winer AM, Jeong S, Fitz DR, Pankratz DV, et al. 2005. Characterizing the range of children's air pollutant exposure during school bus commutes. J Expo Anal Env Epid 15(5):377–387.

Samet JM, Rappold A, Graff D, Cascio WE, Berntsen JH, Huang YCT, et al. 2009. Concentrated ambient ultrafine particle exposure induces cardiac changes in young healthy volunteers. Am J Respir Crit Care Med 179(11):1034–1042.

Sandstrom T, Brunekreef B. 2007. Traffic-related pollution and lung development in children. Lancet 369(9561):535–537.

Shaughnessy RJ, Sextro RG. 2006. What is an effective portable air cleaning device? A review. J Occup Environ Hyg 3(4):169–181.

Solomon GM, Campbell TR, Feuer GR, Masters J, Samkian A, Paul KA. 2001. No breathing in the Aisles: Diesel exhaust inside school buses. Available: *www.nrdc.org/air/transportation/schoolbus/sbusinx.asp* [accessed 1 March 2012].

Trenbath K, Hannigan MP, Milford JB. 2009. Evaluation of retrofit crankcase ventilation controls and diesel oxidation catalysts for reducing air pollution in school buses. Atmos Environ 43(37):5916–5922.

U.S. Environmental Protection Agency. 2001. Control of air pollution from new motor vehicles: Heavy duty engine and vehicle standards and highway diesel fuel sulfur control requirements, final rule. 66 FR 5002. Available: *https://federalregister.gov/a/01-2.*

U.S. Environmental Protection Agency. 2010. MOVES2010 highway vehicle: Population and activity Data. Available: *www.epa.gov/otaq/models/moves/420r10026.pdf* [accessed 2 March 2013].

U.S. Environmental Protection Agency. 2011. Verified retrofit technologies. Available: *http://epa.gov/cleandiesel/verification/techlist-donaldson.htm#donaldson2* [accessed 1 June 2010].

U.S. Environmental Protection Agency. 2012. Clean school bus USA. Available: *www.epa.gov/cleanschoolbus/basicinfo.htm* [accessed 1 April 2012].

vanVliet P, Knape M, deHartog J, Janssen N, Harssema H, Brunekreef B. 1997. Motor vehicle exhaust and chronic respiratory symptoms in children living near freeways. Environ Res 74(2):122–132.

Vinzents PS, Moller P, Sorensen M, Knudsen LE, Hertel O, Jensen FP, et al. 2005. Personal exposure to ultrafine particles and oxidative DNA damage. Environ Health Perspect 113(11):1485–1490.

Wargo J, Brown D. 2002. Children's Exposure to Diesel Exhaust on School Buses. (Cullen M, Addiss S, Alderman N, eds). North Haven, CT:Environmental and Human Health.

Waring MS, Siegel JA, Corsi RL. 2008. Ultrafine particle removal and generation by portable air cleaners. Atmos Environ 42(20):5003–5014.

Wehner B, Uhrner U, von Lowis S, Zallinger M, Wiedensohler A. 2009. Aerosol number size distributions within the exhaust plume of a diesel and a gasoline passenger car under on-road conditions and determination of emission factors. Atmos Environ 43(6):1235–1245.

Weiss R. 2005. Modeling Longitudinal Data. New York, NY:Springer-Verlag.

Wu J, Wilhelm M, Chung J, Ritz B. 2011. Comparing exposure assessment methods for traffic-related air pollution in an adverse pregnancy outcome study. Environ Res 111(5): 685–692.

Zhu Y, Eiguren-Fernandez A, Hinds WC, Miguel AH. 2007. In-cabin commuter exposure to ultrafine particles on Los Angeles freeways. Environ Sci Tech 41(7):2138–2145.

Zhu YF, Hinds WC, Kim S, Shen S, Sioutas C. 2002. Study of ultrafine particles near a major highway with heavy-duty diesel traffic. Atmos Environ 36(27):4323–4335.

Zhu YF, Hinds WC, Krudysz M, Kuhn T, Froines J, Sioutas C. 2005. Penetration of freeway ultrafine particles into indoor environments. J Aerosol Sci 36(3):303–322.

Zielinska B, Campbell D, Lawson DR, Ireson RG, Weaver CS, Hesterberg TW, et al. 2008. Detailed characterization and profiles of crankcase and diesel particulate matter exhaust emissions using speciated organics. Environ Sci Tech 42(15):5661–5666.

APPENDIX AVAILABLE ON THE WEB

Appendix A. Supplemental Table and Figures

Appendix A contains supplemental material not included in the printed report. It is available on the HEI Web site *http://pubs.healtheffects.org*.

ABOUT THE AUTHORS

Yifang Zhu received her Ph.D. in environmental health sciences from the University of California–Los Angeles in 2003. Dr. Zhu was assistant professor in the Department of Environmental Science and Engineering at Texas A&M University–Kingsville between 2006 and 2010. She is currently associate professor in the Department of Environmental Health Sciences at the University of California–Los Angeles. Her research interest is primarily in the field of air pollution, environmental exposure assessment, and aerosol science and technology.

Qunfang Zhang received her Ph.D. in environmental health sciences from the University of California–Los Angeles in 2012. She is currently an air pollution specialist at the California Air Resources Board. Her major contributions in this study are data collection and data analysis.

OTHER PUBLICATIONS RESULTING FROM THIS RESEARCH

Zhang Q, Fischer H, Weiss RE, Zhu Y. 2013. Ultrafine particle concentrations in and around idling school buses. Atmos Environ 69:65–75.

Zhang Q, Zhu Y. 2012. Characterizing ultrafine particles and other air pollutants at five schools in South Texas. Indoor Air 22(1):33–42.

Zhang Q, Zhu Y. 2011. Performance of school bus retrofit systems: Ultrafine particles and other vehicular pollutants. Environ Sci Technol 45: 6475–6482.

Zhang Q, Zhu Y. 2010. Measurements of ultrafine particles and other vehicular pollutants inside school buses in rural South Texas roadways. Atmos Environ 44 (2) 253–261.

ABBREVIATIONS AND OTHER TERMS

AC	air conditioning
BC	black carbon
CO	carbon monoxide
CO_2	carbon dioxide
CCISD	Corpus Christi Independent School District
CFS	crankcase filtration system
CPC	condensation particle counter
DOC	diesel oxidation catalyst
DMA	long differential mobility analyzer
HEPA	high efficiency particulate air
I/O	in-cabin/outdoor pollutant concentration ratio
IT	idling test
MY	model year
NO_2	nitrogen dioxide
PB-PAH	particle-bound polycyclic aromatic hydrocarbon
PM	particulate matter
$PM_{2.5}$	particulate matter ≤ 2.5 μm in aerodynamic diameter
PM_{10}	particulate matter ≤ 10 μm in aerodynamic diameter
PNC	particle number concentration
RH	relative humidity
RT	retrofit test
RTG	retrofit test in open garage
RTRA	retrofit test on road after retrofitting
RTRB	retrofit test on road before retrofitting
SAS	Statistical Analysis Software
SMPS	scanning mobility particle sizer
S/V	surface area to volume ratio
UFP	ultrafine particles
U.S. EPA	United States Environmental Protection Agency

Research Report 180, *Characterizing Ultrafine Particles and Other Air Pollutants In and Around School Buses,* Y. Zhu and Q. Zhang

INTRODUCTION

Numerous studies have documented the adverse effects on mortality and morbidity of exposure to outdoor particulate matter (PM*) (recently reviewed in Rückerl et al. 2011). Children as a group are considered to be particularly susceptible to the effects of PM, which is a complex mixture of particles of different size ranges and composition. Current air quality standards in the United States are based on the mass of particles of aerodynamic diameter ≤ 2.5 µm ($PM_{2.5}$). Some evidence suggests that the smallest particles within this size range — ultrafine particles (UFP), defined as having a diameter of ≤ 0.1 µm — have properties that may make them particularly toxic (reviewed in HEI Review Panel 2013). Accurately assessing exposures to UFP is considered a key research need.

Dr. Yifang Zhu, then of Texas A&M University–Kingsville, submitted an application, "Assessing Children's Exposure to Ultrafine Particles from Vehicular Emissions," to HEI in 2007 under Request for Applications 06-3, the Walter A. Rosenblith New Investigator Award. She originally proposed to identify major factors that influence levels of UFP in two microenvironments in which school children would be likely to be exposed to high levels: during commutes on school buses powered by diesel engines, and inside and outside classrooms near major roads. The HEI Research Committee requested that Dr. Zhu include an evaluation of different retrofit devices, and in her revised proposal, she added an evaluation of two such devices, a diesel oxidation catalyst (DOC) and a crankcase filtration system (CFS). The Committee recommended the study for funding.

Dr. Yifang Zhu's 3-year study, "Assessing Children's Exposure to UFP from Vehicular Emissions," began in January 2008. Total expenditures were $300,000. The draft Investigators' Report from Zhu and Zhang was received for review in June 2012. A revised report, received in April 2013, was accepted for publication in June 2013. During the review process, the HEI Health Review Committee and the investigators had the opportunity to exchange comments and to clarify issues in both the Investigators' Report and the Review Committee's Critique.

* A list of abbreviations and other terms appears at the end of the Investigators' Report.

This Critique is intended to aid the sponsors of HEI and the public by highlighting both the strengths and limitations of the study and by placing the Investigators' Report into scientific and regulatory perspective.

BACKGROUND

Emissions from motor vehicles are a major source of ambient UFP (HEI Review Panel 2013). Central monitoring of UFP concentrations in ambient air is generally lacking, however, and at the start of this project, few studies had assessed levels of UFP in real-world environments. One of the first was conducted by Zhu and colleagues (2002): they measured concentrations of UFP close to a major highway and found that levels were highest close to the downwind direction of the highway, but dropped rapidly moving away from it. The steepest decline occurred within the first 50 meters from the road, and UFP levels were indistinguishable from background levels at 300 meters. These findings have been confirmed in other studies (e.g., Beckerman et al. 2008; Westerdahl et al. 2005).

Following up these findings of high concentrations of UFP close to major roads, several studies evaluated UFP levels inside vehicles. Most were conducted in passenger cars, but three had assessed UFP levels in school buses (Hammond et al. 2007; Hill et. al 2005; Rim et al. 2008). These studies showed that in-cabin UFP levels could be substantial, but were highly dependent on specific characteristics of the measurement scenario, such as engine age, driving conditions, ventilation behavior, local traffic, and ambient air pollution.

Exposures to pollutants while in traffic are typically much higher than those measured at fixed-site monitors because they are closer to the source (Kaur et al. 2007). Although time spent in traffic generally accounts only for a relatively short amount of time within a person's day — the average for the U.S. population is 52 commuting minutes per day (American Community Survey 2012) — exposures encountered during this period can make a substantial contribution to a person's daily average pollutant exposure. For example, for UFP this was estimated to be 33%–45% (Fruin et al. 2008) or 10%–50% (Zhu et al. 2007).

Making accurate assessments of pollutant emissions from traffic is important because epidemiologic and controlled-exposure studies have shown that exposure to traffic-related air pollution is associated with health effects (HEI Panel on the Health Effects of Traffic-Related Air Pollution 2010). In addition, an important component of traffic-related emissions, diesel exhaust, was recently classified by the International Agency for Research on Cancer as carcinogenic to humans (Benbrahim-Tallaa et al. 2012).

In light of such concerns, the U.S. Environmental Protection Agency (U.S. EPA) has established stringent new standards for diesel exhaust emissions and fuels for light- and heavy-duty diesel vehicles. Because initial exposure studies in diesel-powered school buses showed significant self-pollution — that is, the intrusion of a bus' own exhaust into its cabin (e.g., Solomon et al. 2001; Wargo and Brown 2002), there have been multiple efforts to reduce children's exposure to diesel emissions from school buses in particular (e.g., see the U.S. EPA's Clean School Bus Program [U.S. EPA 2003]). The U.S. EPA reported substantial reductions in PM emissions with certified retrofit technologies including a DOC and a CFS (U.S. EPA 2012). Only a few studies, however, have evaluated the effect of a retrofit device on in-cabin concentrations (see the Effects of Retrofitting section of this Critique). The current study was intended to add to the small number of studies assessing UFP concentrations in microenvironments in which children are likely to be exposed to high levels.

DESCRIPTION OF THE STUDY

OBJECTIVES AND SPECIFIC AIMS

The overall objective was to identify the conditions under which children are likely to be exposed to high levels of UFP and other air pollutants in and around school buses. The specific aims were to:

1. Identify important factors affecting air pollutants inside school buses while driving or idling.
2. Quantify the effectiveness of two U.S. EPA-verified retrofit technologies for diesel-powered school buses — a DOC and a CFS — in reducing tailpipe emissions and in-cabin air pollutant levels.
3. Explore the potential of in-cabin filtration to reduce PM levels inside school bus cabins.

Zhu and her colleague, Dr. Qunfang Zhang, also made a limited set of measurements of pollutant levels in and outside school buildings. After Dr. Zhu moved to California during the second year of the study, the Research Committee recommended focusing on the school bus study for the remainder of the study. During its evaluation, the HEI Health Review Committee concluded that these limited measurements in the school buildings, which have been published elsewhere (Zhang and Zhu 2012), did not make a significant contribution to the overall study. Dr. Zhu agreed with the Review Committee's recommendation to remove the section on measurements at school buildings from the report.

STUDY DESIGN

Critique Table 1 summarizes the key points of the study's design, which consisted of four sets of tests for UFP and other pollutants in and around school buses in different conditions: (1) on-road; (2) during idling; (3) before and after retrofitting; and (4) before and after operating a high efficiency particulate air (HEPA) filter air purifier inside the cabin. Among the points to note are that the four sets of tests included mainly different buses (from 4 to 9 buses in each set), in different locations, and in different years and seasons. Most of the buses were evaluated in sequence and each on a single day. Only the buses in the retrofit tests and the HEPA air purifier test were retrofitted.

For most tests, air pollutants were measured simultaneously inside the cabin as well as directly outside the cabin or close to the tailpipe with identical equipment. However, only in-cabin concentrations were measured for the on-road test.

Measurements inside the cabin were taken at the rear seats of the bus. Outside measurements were either made directly outside the (driving) bus, 2 m upwind of the bus (idling test), 0.5 m from the tailpipe (idling test), or directly sampled at the tailpipe with the use of a dilution system (open garage retrofit test).

The investigators measured air pollution concentrations in real time. In most tests, particle number concentrations (PNC) were measured using a condensation particle counter (CPC 3785), which counts particles in the size range 5 nm to > 1000 nm. Because number count is dominated by UFP, PNC can be interpreted as a measure of UFP. Particle size distribution was measured using a scanning mobility particle sizer (SMPS), which additionally counts particles in a specific size range (7.6–289 nm), and this provided an alternative estimate of UFP. Note that the investigators use "PNC" to refer to the CPC measurements and "UFP" to refer to the SMPS measurements. In this Critique, however, we use UFP to cover both types of measurements.

$PM_{2.5}$ and black carbon (BC) were measured with a DustTrak and an Aethalometer, respectively. Carbon monoxide (CO) and carbon dioxide (CO_2) were measured with a Q-track. The pollutants that were measured differed

Critique Table 1. Summary of Study Design

Test / *N* Buses / Model Year Range	Sampling Period (Retrofit)	Sampling Schedule per Bus	Important Features	Location of Pollutant Measurements		
				Inside	Outside	Tailpipe
1. On-roadway: Beeville, TX						
4 Buses 1990–2006	March–May 2008 (No retrofit)	Two 2-hr runs (morning, afternoon) on two days.	One run consisted of a town route and a rural route. No AC. Windows closed and no children on board in 2 buses. Windows open on 2 other buses; number of people on board (children + research staff) varied from 7–45.	Yes	No	No
2. Idling: Los Angeles, CA						
9 Buses 1999–2005	August–September 2010 (No retrofit)	5 hrs on one day (30 min before and after turning on the engine per scenario)	Different scenarios with one or two buses parked parallel or perpendicular to wind direction. Two buses parked at 90 degrees from each other. Window position varied. AC turned off. Door closed. Test conducted in an open green space under stable meteorologic conditions. No children on board.	Yes[a]	Yes[b]	Yes[c]
3. Retrofit: Corpus Christi, TX[d]						
Open Garage						
6 Buses 1992–1999	July 2009 (before and after retrofit)	1.5 hrs before and 3 hrs after retrofit on one day	Test was done while idling in an open garage. AC turned off, and windows closed. No children on board.	Yes[e]	No	Yes[f]
On-Road						
6 Buses 1990–1999	April 2009 (before retrofit) October–November 2009 (after retrofit)	Two 2-hr runs (morning, afternoon) on one day before and one day after retrofit	One run consisted of two routes. Windows were closed. AC turned on in afternoon. Number of children on board varied from 3–45.	Yes	Yes[e]	No
4. HEPA Air Purifier: Corpus Christi, TX						
4 Buses 1992–1999	April 2010 (retrofitted buses only)	Two 2-hr runs (morning, afternoon) on one day	The same run as in the on-road retrofit test was used. One or two HEPA air purifiers were installed at the rear of the bus. Windows were closed. AC turned on in afternoon. Number of children on board not reported.	Yes	Yes[e]	No

[a] Ultrafine particles (UFP) were measured with the use of a scanning mobility particle sizer. Black carbon (BC) was not measured.

[b] Particle size distribution and BC were not measured.

[c] Particle size distribution, $PM_{2.5}$, BC, carbon monoxide (CO), and carbon dioxide (CO_2) were not measured. (Only UFP was measured.)

[d] Two different devices were tested individually and in combination (only the combination in the on-road test). The DOC muffler (Series 6100) and the Spiracle CFS were both obtained from Donaldson Company, Minneapolis, MN. See for more information: *http://www.donaldson.com/en/exhaust/emission/index.html.*

[e] BC was not measured.

[f] CO and CO_2 were not measured.

between and within tests (see footnotes to Critique Table 1). Data on UFP and $PM_{2.5}$ were the most complete. BC measurements were often missing because only one device was available. CO and CO_2 were reported only in the on-road test, even though these measurements had been obtained in most of the tests.

Speed, direction, and location of the school buses were monitored by a global positioning system. Meteorologic data including wind direction and wind speed were obtained from local weather stations.

STATISTICAL ANALYSES

Descriptive Statistics

Means and standard deviations were calculated for each pollutant, inside and outside of the cabin, for the different tests. In addition, correlations were calculated between the different air pollutants inside the bus (in the on-road test), as well as between in-cabin and outside-the-cabin concentrations (in the retrofit test). Ratios were calculated of air pollutants between engine-on and engine-off conditions in the idling tests. Indoor/outdoor (I/O) concentration ratios were calculated in the HEPA air purifier test.

Additional Statistics

Longitudinal regression modeling was used to estimate the effects of wind, window position, and emission source over time, as well as their interactions for in-cabin air pollution levels in the idling test. This was done for UFP and $PM_{2.5}$ as well as for different particle size distributions. Correlation between measurements over time on a single bus was taken into account with the use of a first-order autoregressive moving average model. Means and 95% confidence intervals of measurements under each scenario were reported.

The statistical significance of the effect of retrofitting was tested with the use of Wilcoxon signed-ranked tests with pre- and post-data matched by bus. In addition, similar tests were used to determine the effects of air conditioning (AC) systems on in-cabin levels. A P value of less than 0.05 was considered statistically significant.

OVERVIEW OF KEY RESULTS

Retrofitting

- Tailpipe concentrations of UFP were significantly reduced (by 20% to 94%) in six idling school buses after they were retrofitted with a DOC and/or a CFS. In the same test, BC and $PM_{2.5}$ were also reduced close to the tailpipe, on average by 64% and 47%, respectively, but only the reduction in BC was statistically significant.
- However, none of the measured in-cabin pollutant concentrations were reduced after a DOC and/or CFS retrofit, either while the buses were idling or driving with children on board.
- Other factors — including ambient levels and the use of AC — were more important than the retrofit device in affecting in-cabin pollutant levels.

HEPA Air Purifier and the Use of AC

- In-cabin concentrations of UFP and $PM_{2.5}$ were substantially reduced (by about 50%) by using one or two HEPA air purifiers, and to a somewhat lesser extent (~25%) by using the AC while driving with children on board.

Idling

- In buses not retrofitted, close-to-tailpipe UFP concentrations during idling were greatly influenced by the bus' own engine; UFP increased 7- to 26-fold in the different scenarios.
- In-cabin UFP concentrations were affected only when the wind blew from the back to the front of the bus, especially with nearby buses idling together (up to a 5.8-fold increase). A very small fraction (< 0.1%) of tailpipe UFP concentrations entered the bus cabin when idling. Thus, the tests indicated that a vehicle's self-pollution generally had little impact on in-cabin UFP levels. In contrast, infiltration of emissions from nearby buses affected in-cabin levels considerably.
- In-cabin $PM_{2.5}$ concentrations were not affected by engine operation in the different idling scenarios.

HEALTH REVIEW COMMITTEE EVALUATION

In its independent review of the study by Zhu and Zhang, the HEI Health Review Committee noted that the study adds to the small number of studies assessing air pollutants including UFP in and around U.S. school buses (Hill et al. 2005; Liu et al. 2010; Rim et al. 2008; Trenbath et al. 2009). The Committee considered that the retrofit and idling tests in particular provided useful information with insights about pollutant levels to which children may be exposed and the factors influencing exposure.

A strength of the study was that in many of the tests (idling, retrofit, and HEPA air purifier tests) air pollutants were measured simultaneously inside as well as directly

outside the cabin and with identical equipment. Although many in-vehicle studies acknowledge the importance of ambient air pollution levels when characterizing in-vehicle levels, only a few have measured them (reviewed in Knibbs et al. 2011). Some previous studies evaluating the performance of retrofit technologies on in-cabin levels did not make concurrent measurements of ambient concentrations as well (Hammond et al. 2007; Trenbath et al. 2009).

Because of certain design decisions, the Committee concluded that some study results were open to interpretation and thus some conclusions should be considered cautiously. One major reason was that the influence of ambient pollution levels on in-cabin levels was not readily determined: ambient levels were not measured in the on-road tests, and in-cabin measurements in the retrofit tests were not adjusted for the influence of ambient levels.

EFFECTS OF RETROFITTING

The Committee liked the fact that the same buses were tested before and after retrofitting. This is especially important since some previous retrofit studies were unable to distinguish the effects of a retrofit device from other attributes of the newer buses on which they were installed (Hammond et al. 2007; Adar et al. 2008).

The Committee agreed that the study provided evidence that retrofitting a bus with a DOC, a CFS, or both substantially reduced tailpipe concentrations of UFP, BC, and $PM_{2.5}$ during idling; only the tailpipe reductions in UFP and BC were statistically significant, however. The investigators found that in-cabin concentrations of all measured pollutants were not reduced in idling and on-road tests of in-use buses after retrofitting, which suggests that factors other than vehicle self-pollution were more important determinants of in-cabin concentrations. The Committee agreed with the investigators that one of the key factors in determining in-cabin levels was the ambient levels of pollutants, as had been shown in previous studies (Asmi et al. 2009; Kaur et al. 2007; Knibbs et al. 2011; Zuurbier et al. 2010).

However, the Committee expressed concern about the absence of an adjustment for varying ambient levels in the retrofit analyses, because it prevented reliable conclusions from being drawn about the effectiveness of the two retrofit devices in reducing in-cabin levels of UFP and $PM_{2.5}$. The Committee felt that this correction was especially important since the before- and after-retrofit measurements were made in different seasons, which were likely to have different ambient air pollution levels.

The Committee also noted that the tailpipe concentration measurements were performed with a non-standard dilution system that was unique to the study. This limited the ability to extrapolate and compare results with those of other studies. In addition, the investigators' evaluation of the effect of adding the CFS — which is designed to reduce crankcase rather than tailpipe emissions — would have been strengthened by making measurements at the crankcase, as well as at the tailpipe.

The current study's data on the effect of retrofitting can be compared only with a small number of previous retrofit studies that used similar technologies. The results have not been consistent among these studies; some found a decrease in in-cabin UFP after retrofitting (Hammond et al. 2007; Trenbath et al. 2009), another found no substantial effect on UFP (Rim et al. 2008), and one found a reduction in UFP after retrofitting with a CFS, but not with a DOC (Hill et al. 2005). Some of these studies have issues regarding simultaneous ambient measurements that are similar to those in the current study: Hammond and colleagues (2007) and Trenbath and colleagues (2009) did not include measurements of ambient air pollution, whereas Rim and colleagues (2008) did not formally correct for ambient levels in the in-cabin evaluation, although measurements were obtained and compared. Hill and colleagues (2005) subtracted ambient levels from in-vehicle concentrations in order to estimate properly the in-cabin effect of a retrofit device.

Further limitations of the study by Zhu and Zhang were the small number of buses (due in part to limited resources), which may have reduced the study power, and the lack of information on other important pollutants associated with traffic that were either not measured in all tests in all locations (BC and CO), or not measured at all (e.g., nitrogen dioxide [NO_2]). However, it should be noted that the investigators ran into feasibility issues regarding the number of instruments that can be installed inside in-use buses. It would have been very useful to obtain NO_2 concentrations in particular, because particle traps, when used without a scrubbing device for oxides of nitrogen, have been shown to increase the ratio of NO_2 to NO markedly (HEI 2011). Also, it was unfortunate that the evaluation of a third retrofit device — the widely used diesel particulate filter, which was intended to be part of the idling test — did not take place in time to be included in the study.

IDLING TEST

The Committee noted that close-to-tailpipe UFP concentrations increased greatly after the engine was turned on during idling and that in-cabin concentrations were affected only when the wind blew from the back to the front of the bus, especially with nearby buses idling together. Thus, the Committee agreed with the investigators that self-pollution was a minor factor for UFP, suggesting that other factors — in particular, ambient levels

including emissions from nearby vehicles — were more important in influencing in-cabin UFP levels.

However, the Committee considered that some decisions regarding the study design prevented extrapolation of the main findings from the bus idling tests to other, more general situations, for several reasons.

First, the investigators used a unique alignment of buses — at right angles to each other — that is unlikely to be encountered in real-world settings because buses at U.S. schools usually line up closely behind each other in one, or sometimes more, rows (e.g., Behrentz et al. 2005).

Second, the investigators used wind direction information from a weather site five kilometers away from the school bus testing site. Although they reported stable conditions at the monitor, they did not establish whether there were local, smaller scale fluctuations in wind direction or possible effects of wind speed onsite during the idling tests that may have affected the results.

Third, although the Committee generally agreed with the investigators' point that crankcase emissions were unlikely to be important to in-cabin pollution levels, which have been shown in some other studies (e.g., Hill et al. 2005) to be an important source of $PM_{2.5}$ in particular. However, the Committee noted that no specific test was conducted to eliminate this source of self-pollution.

EFFECTS OF THE USE OF AC AND HEPA AIR PURIFIERS

The Committee agreed with the investigators that turning on the AC reduced in-cabin concentrations of all pollutants, including substantial reductions in UFP and $PM_{2.5}$ concentrations, as has been demonstrated in previous studies (e.g., Sabin et al. 2005; Zhu et al. 2007). To date, one other study investigated effects of an air purifier and reported large reductions of UFP in a small test with three passenger cars (Tartakovsky et al. 2013). However, the Committee considered the HEPA air purifier test to be limited because only a small number of samples were taken and no formal tests were performed to determine statistical significance. In addition, because the HEPA air purifier was placed at the rear of the bus, the study could not provide information about whether an alternative placement of the purifier would have resulted in larger or smaller decreases in pollutant levels. Hence, the reported reductions of UFP and $PM_{2.5}$ associated with HEPA air purifier use needs further study.

ON-ROAD TEST

The Committee concluded that the on-road tests provided some information about factors — such as window position, engine age, driving speed, route and location, and operating conditions — that may affect in-cabin concentrations, but overall the tests had limited value. The main reason for the Committee's concern was that ambient pollutant concentrations, which can affect in-cabin concentrations considerably, were not measured during these tests. Although the investigators discuss the possibility that the results of their on-road tests may have been confounded by ambient levels, the Committee thought this point was not given enough prominence in the report. In addition, it would have been useful to collect data on traffic intensities on any of the routes, including whether there were other vehicles in close proximity to the bus, because previous studies (e.g., Sabin et al. 2005; Westerdahl et al. 2005) have shown that the pollutants emitted by passing and preceding vehicles can substantially increase in-vehicle concentrations.

IMPLICATIONS OF EXPOSURE OF SCHOOL CHILDREN TO UFP IN AND AROUND SCHOOL BUSES

The current study was intended to provide information about the factors influencing levels of UFP and other pollutants in and around school buses. It was not designed to provide information on the relative contribution of the school bus commute to children's overall UFP exposures. A few studies have estimated that a short commute can account for up to 50% of daily UFP exposures (Fruin et al. 2008; Zhu et al. 2007). However, the measurement campaigns in those studies were relatively limited and the studies were done only in Los Angeles, California, and thus the results may not be applicable to other settings. Additional studies are needed to estimate the relative contributions of in-vehicle microenvironments to air pollutant exposures, but also the assessment of the contributions of other microenvironments in which children and adults spend most of their time (such as home, work, and school) is particularly important.

Since a substantial fraction of at least some children's daily exposure may come from bus transfer locations or waiting areas where multiple buses are idling, the Committee thought that the reduction in tailpipe concentrations after retrofitting could reduce children's overall exposure to air pollutants and contribute to overall cleaner outdoor air. Further reductions in children's exposure could also be achieved by reducing idling time, increasing the distances between buses during driving and idling, increasing the distances between buses and other vehicles, and by avoiding high-traffic roads.

The current study measured air pollution concentrations in and around school buses and did not assess possible health effects related to school bus commutes on the

school children. Children were actually on board the buses in only a few of the tests, and the effect of children's being on board on exposure levels was not further investigated. Moreover, in-cabin concentrations were measured only at the rear of the bus. Some studies that used a tracer gas to explore the gradient of pollutants inside a bus found higher concentrations at the rear than at the front (e.g., Behrentz et al. 2004). If there is a similar gradient of UFP levels, it is likely that the current study overestimated average in-cabin exposure of children.

To date, no studies have been performed that have linked in-cabin exposures to health effects in school children. Epidemiologic and controlled-exposure studies have shown that exposure to traffic-related air pollution is associated with health effects and that children are particularly susceptible (HEI Panel on the Health Effects of Traffic-Related Air Pollution 2010; Rückerl et al. 2011). Whether these associations are also found in situations in which relatively high exposures are experienced for only a limited time (such as while in traffic) has not yet been fully explored — although the few epidemiologic and controlled-exposure studies to date suggest that being in traffic can elicit both acute respiratory and cardiovascular effects (reviewed in Knibbs et al. 2011). Therefore, reducing UFP as well as other traffic-related air pollutants would be likely to lead to improvements in the health effects associated with traffic exposures.

SUMMARY AND CONCLUSIONS

The goal of Zhu and Zhang's study was to identify factors likely to influence children's exposure to UFP and other pollutants, such as $PM_{2.5}$ and BC, in and around school buses powered by diesel engines. The study consisted of four sets of tests: (1) on-road; (2) during idling; (3) before and after retrofitting; and (4) with a HEPA air purifier in the cabin. Air pollutants were measured simultaneously inside as well as directly outside the cabin or close to the tailpipe, except during the on-road test in which measurements were made only in the cabin. Measurements were made in small sets of buses (model years 1990–2006) in Texas and in California.

In its independent review of the study, the HEI Review Committee concluded that the study by Zhu and Zhang adds to the small number of studies assessing air pollutants including UFP in and around U.S. school buses. The Committee considered that the retrofit and idling tests in particular provided useful information and agreed with the investigators that idling substantially increased levels of UFP close to the tailpipe. Retrofitting buses with a DOC, a CFS, or both substantially reduced the tailpipe concentrations of UFP, BC, and $PM_{2.5}$ (although the $PM_{2.5}$ reduction was not statistically significant) during idling. Retrofitting did not reduce in-cabin levels of any measured pollutant, suggesting that factors other than self-pollution by the vehicle itself were more important determinants of in-cabin concentrations. In particular, both the Committee and the investigators concluded that ambient levels of pollutants that included emissions from nearby vehicles were more important than self-pollution in influencing in-cabin concentrations. The use of a HEPA air purifier and the AC also substantially decreased in-cabin levels of UFP and $PM_{2.5}$.

Since a substantial fraction of at least some children's daily exposure may come from bus transfer locations or waiting areas where multiple buses are idling, the Committee considered that the reduction in tailpipe concentrations after retrofitting could reduce children's overall exposure to air pollutants and contribute to overall cleaner outdoor air. Further reductions in children's exposure could also be achieved by reducing idling time, increasing the distances between buses during driving and idling, increasing the distances between buses and other vehicles, and by avoiding high-traffic roads.

Because of certain design decisions by the investigators, the Committee concluded that some study results were open to interpretation and thus some conclusions should be considered cautiously. One major factor was that the influence of ambient levels on in-cabin levels could not readily be determined, since ambient levels were not measured in the on-road tests, and in-cabin measurements in the retrofit tests were not adjusted for the influence of ambient levels. For future studies, the Committee would make the following additional recommendations: (1) collect detailed data on other pollutants — in particular NO_2, BC, and CO — as well as data on traffic intensities; (2) use a standard dilution system when measuring tailpipe concentrations; (3) use a more common alignment of buses (i.e., not perpendicular), and account for local fluctuations in wind direction or the possible effects of wind speed when studying idling buses; (4) within available resources, test a larger number of buses, with engines of varying ages, and in different seasons.

The HEI Health Review Committee concluded that the study by Zhu and Zhang holds important methodologic lessons for future in-vehicle studies, since it highlights the importance of including measurements of ambient air pollution concentrations. In-vehicle studies remain an important area of future research, because in-vehicle exposure may contribute substantially to a person's average exposure to pollutants such as UFP, in spite of the fact that time spent in vehicles makes up only a relatively small amount of a person's day. Additional studies are needed to estimate

the relative contributions of in-vehicle microenvironments to air pollutant exposures, but also the assessment of the contributions of other microenvironments in which children and adults spend most of their time (such as home, work, and school) is particularly important.

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REFERENCES

Adar SD, Davey M, Sullivan JR, Compher M, Szpiro A, Liu SLJ. 2008. Predicting airborne particle levels aboard Washington state school buses. Atmos Environ 42:7590–7599.

American Community Survey. 2012. Available: *http://factfinder2.census.gov/faces/tableservices/jsf/pages/productview.xhtml?pid=ACS_12_1YR_S0801&prodType=table.*

Asmi E, Antola M, Yli-Tuomi T, Jantunen M, Aarnio P, Mäkelä T, et al. 2009. Driver and passenger exposure to aerosol particles in buses and trams in Helsinki, Finland. Sci Total Environ 407:2860–2867.

Beckerman B, Jerrett M, Brook JR, Verma DK, Arain MA, Finkelstein MM. 2008. Correlation of nitrogen dioxide with other traffic pollutants near a major expressway. Atmos Environ 42:275–290.

Behrentz E, Fitz DR, Pankratz DV, Sabin LD, Colome SD, Fruin SA, et al. 2004. Measuring self-pollution in school buses using a tracer gas technique. Atmos Environ 38:3735–3746.

Behrentz E, Sabin LD, Winer AM, Fitz DR, Pankratz DV, Colome SD, et al. 2005. Relative importance of school bus-related microenvironments to children's pollutant exposure. J Air Waste Manag Assoc 55:1418–1430.

Benbrahim-Tallaa L, Baan RA, Grosse Y, Lauby-Secretan B, El Ghissassi F, Bouvard V, et al. 2012. Carcinogenicity of diesel-engine and gasoline-engine exhausts and some nitroarenes. Lancet Oncology 13:663–664.

Fruin S, Westerdahl D, Sax T, Sioutas C, Fine PM. 2008. Measurements and predictors of on-road ultrafine particle concentrations and associated pollutants in Los Angeles. Atmos Environ 42:207–219.

Hammond D, Jones S, Lalor M. 2007. In-vehicle measurement of ultrafine particles on compressed natural gas, conventional diesel, and oxidation-catalyst diesel heavy-duty transit buses. Environ Monit Assess 125:239–246.

Health Effects Institute. 2011. The Future of Vehicle Fuels and Technologies: Anticipating Health Benefits and Challenges. Communication 16. Boston, MA:Health Effects Institute.

HEI Panel on the Health Effects of Traffic-Related Air Pollution. 2010. Traffic-Related Air Pollution: A Critical Review of the Literature on Emissions, Exposure, and Health Effects. HEI Special Report 17. Boston, MA:Health Effects Institute.

HEI Review Panel on Ultrafine Particles. 2013. Understanding the Health Effects of Ambient Ultrafine Particles. HEI Perspectives 3. Boston, MA:Health Effects Institute.

Hill L, Zimmerman N, Gooch J. 2005. A multi-city investigation of the effectiveness of retrofit emissions controls in reducing exposures to particulate matter in school buses. Boston, MA:Clean Air Task Force.

Kaur S, Nieuwenhuijsen MJ, Colvile RN. 2007. Fine particulate matter and carbon monoxide exposure concentrations in urban street transport microenvironments. Atmos Environ 41:4781–4810.

Knibbs LD, Cole-Hunter T, Morawska L. 2011. A review of commuter exposure to ultrafine particles and its health effects. Atmos Environ 45:2611–2622.

Liu LJ, Phuleria HC, Webber W, Davey M, Lawson DR, Ireson RG, et al. 2010. Quantification of self pollution from two diesel school buses using three independent methods. Atmos Environ 44:3422–3431.

Rim D, Siegel J, Spinhirne J, Webb A, McDonald-Buller E. 2008. Characteristics of cabin air quality in school buses in Central Texas. Atmos Environ 42:6453–6464.

Rückerl R, Schneider A, Breitner S, Cyrys J, Peters A. 2011. Health effects of particulate air pollution: A review of epidemiological evidence. Inhal Toxicol 23:555–592.

Sabin LD, Behrentz E, Winer AM, Jeong S, Fitz DR, Pankratz DV, et al. 2005. Characterizing the range of children's air pollutant exposure during school bus commutes. J Expo Anal Environ Epidemiol 15:377–387.

Solomon G, Campbell T, Ruderman Feuer G, Masters J, Samkian A, Paul K. 2001. No breathing in the aisles. Diesel exhaust inside school buses. Available: *www.nrdc.org/air/transportation/schoolbus/schoolbus.pdf*. Natural Resources Defense Council, Coalition for Clean Air.

Tartakovsky L, Baibikov V, Czerwinski J, Gutman M, Kasper M, Popescu D, et al. 2013. In-vehicle particle air pollution and its mitigation. Atmos Environ 64:320–328.

Trenbath K, Hannigan MP, Milford JB. 2009. Evaluation of retrofit crankcase ventilation controls and diesel oxidation catalysts for reducing air pollution in school buses. Atmos Environ 43:5916–5922.

U.S. Environmental Protection Agency. 2003. Clean school bus program. Available: *www.epa.gov/cleanschoolbus/csb-overview.htm*.

U.S. Environmental Protection Agency. 2012. U.S. EPA verified retrofit technologies. Available: *www.epa.gov/cleandiesel/verification/verif-list.htm.*

Wargo J, Brown D. 2002. Children's exposure to diesel exhaust on school buses. (Cullen M, Addis S, Alderman N eds). Available: *www.ehhi.org/reports/diesel/diesel.pdf*. North Haven, CT:Environment and Human Health.

Westerdahl D, Fruin S, Sax T, Fine PM, Sioutas C. 2005. Mobile platform measurements of ultrafine particles and associated pollutant concentrations on freeways and residential streets in Los Angeles. Atmos Environ 39:3597–3610.

Zhang Q, Zhu Y. 2012. Characterizing ultrafine particles and other air pollutants at five schools in South Texas. Indoor Air 22:33–42.

Zhu Y, Eiguren-Fernandez A, Hinds WC, Miguel AH. 2007. In-cabin commuter exposure to ultrafine particles on Los Angeles freeways. Environ Sci Technol 41:2138–2145.

Zhu Y, Hinds WC, Kim S, Shen S, Sioutas C. 2002. Study of ultrafine particles near a major highway with heavy-duty diesel traffic. Atmos Environ 36:4323–4335.

Zuurbier M, Hoek G, Oldenwening M, Lenters V, Meliefste K, van den Hazel P, et al. 2010. Commuters' exposure to particulate matter air pollution is affected by mode of transport, fuel type, and route. Environ Health Perspect 118:783–789.

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HEALTH
EFFECTS
INSTITUTE

101 Federal Street, Suite 500
Boston, MA 02110, USA
+1-617-488-2300
www.healtheffects.org

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