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Abstract Traffic emissions have been associated with a wide range of adverse health effects. Many schools are situated close to major roads, and as children spend much of their day in school, methods to reduce traffic-related air pollutant concentrations in the school environment are warranted. One promising method to reduce pollutant concentrations in schools is to alter the timing of the ventilation so that high ventilation time periods do not correspond to rush hour traffic. Health Canada, in collaboration with the Ottawa-Carleton District School Board, tested the effect of this action by collecting traffic-related air pollution data from four schools in Ottawa, Canada, during October and November 2013. A baseline and intervention period was assessed in each school. There were statistically significant ($P < 0.05$) reductions in concentrations of most of the pollutants measured at the two late-start (9 AM start) schools, after adjusting for outdoor concentrations and the absolute indoor–outdoor temperature difference. The intervention at the early-start (8 AM start) schools did not have significant reductions in pollutant concentrations. Based on these findings, changing the timing of the ventilation may be a cost-effective mechanism of reducing traffic-related pollutants in late-start schools located near major roads.

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Practical Implications

This study found that adjusting the ventilation system to operate earlier in the morning, prior to high traffic commute periods, significantly reduced traffic-related air pollutants within late-start schools (9 AM) located near major roads. Given the ease of implementation and cost-effectiveness, this intervention may be a feasible approach for improving indoor air quality in schools and other public buildings with heating, ventilation, and air conditioning (HVAC) systems.

Introduction

In Canada, approximately 36% of urban public elementary schools are located within 200 m of a major road (Amram et al., 2011). This is of concern, given that traffic-related air pollutants are elevated within approximately 100–500 m of major roadways (Zhou and Levy, 2007) and elevated concentrations of traffic-related pollutants have been found in schools located near highways or major roads (Janssen et al., 2001; Patel et al., 2009). Chronic exposure to traffic-related pollution has been linked with several adverse health outcomes in children. These include increased respiratory symptoms, asthma exacerbations, and adverse effects on lung and cognitive development (Brauer et al., 2002; Dales et al., 2008; Gauderman et al., 2002, 2004; Janssen et al., 2003; Kim et al., 2004; McConnell et al., 2010; Sunyer et al., 2015; Venn et al., 2001). Given that children spend approximately 6 h a day at school, for a minimum of 194 days a year (Ontario Ministry of Education, 1990), pollutants in this environment are an important consideration for children's health. In their recent review, Choo et al. (2014) identified that most of the indoor air pollutants in schools can be attributed to vehicle emissions and industrial sources.

A number of strategies have been proposed to reduce children's exposure to traffic-related pollutants in the school environment. These include mandatory setbacks of schools from roadways; increased filtration/filtration maintenance; the use of sound walls, trees or other vegetation as barriers; and modified building/HVAC design (Brauer et al., 2012; White et al., 2005). However, these interventions are often not feasible or are cost prohibitive for many school boards with limited resources. In addition, many of these strategies have limited applicability for gaseous pollutants.

One promising strategy for reducing exposure to traffic-related pollutants is to alter the timing of the heating, ventilation, and air conditioning (HVAC) system operations to correspond with diurnal reductions in ambient pollutant concentrations (Sonoma Technology Inc., 2010). Although the diurnal variability of pollutant concentrations depends on many factors such as photochemistry, meteorology, and emission sources, traffic-related pollutants typically peak between 7 and 9 AM as a result of rush hour traffic (Touma et al., 2006). It has been suggested that starting HVAC systems earlier in the morning when outdoor pollutant concentrations are low, and then turning the systems off from the beginning of the morning rush hour until the systems must be on for building occupancy could result in reductions of more than 50% in pollutant concentrations (Sonoma Technology Inc., 2010). Jamriska et al. (2003) used a single-zone mathematical model to determine the effects of air filtration and ventilation on indoor particles. They found that indoor particle concentrations could

be reduced by minimizing the intake of outdoor air during periods when outdoor air concentrations were higher than indoor concentrations.

Existing air quality management programs related to motor vehicles generally focus on reducing emissions from individual vehicles. These reductions have been achieved through changes to emission standards, cleaner fuels, and vehicle inspection programs ([HEI] Health Effects Institute, 2010; Environment Canada 2014). There has been less attention placed on reducing population exposures to traffic sources through land-use policies, or building/ventilation practices (Ministry of Environment, 2006). This study examines one mitigation strategy to potentially reduce exposures to traffic-related air pollution for susceptible subpopulations. This strategy could be readily implemented for buildings with a programmable HVAC system.

The two primary objectives of this study were as follows: (i) to determine the impact of modifying the timing of the HVAC operations on indoor air pollution concentrations to ensure that high ventilation time periods do not correspond to rush hour periods; and (ii) to estimate the infiltration efficiencies (F_{inf}) and the contributions of ambient- and indoor-generated pollutants under typical and modified ventilation conditions.

Methods

Study design

In October and November of 2013, traffic-related air pollutant concentrations were measured at four Ottawa public elementary schools during normal HVAC operations (baseline) and during a period when the timing of the fresh air intake was changed (intervention). Two schools were early-start schools (8 AM–2:30 PM), and two schools were late-start schools (9 AM–3:30 PM). A total of 16 consecutive school days of baseline monitoring and 16 consecutive school days of intervention monitoring were completed at each school. Data were not collected on weekends, professional development days, or other holidays. To minimize the influence of temporal patterns in outdoor pollutant concentrations and temperature, two schools (one early start and one late start) were assigned to have the intervention first, and two schools (one early start and one late start) were assigned to have the intervention second.

Measurements were collected from three locations at each school: (i) in one classroom; (ii) outdoors near the HVAC air intake; and (iii) outdoors near the front door. Classrooms were chosen based on the age of the children and the teachers' willingness to participate. Classrooms with older children (grades three to six) were preferentially chosen to reduce the potential for damage to the instruments. Teachers were instructed to carry out daily activities as usual, such as any window

Table 1 Overview of equipment used in each location

Parameter	Unit of measurement	Device	Time period	Location		
				Classroom	HVAC inlet	Front entrance
Fine Particles (FP)	DustTrak reading	DustTrak	Continuous	X	X	X
Black Carbon (BC)	ng/m ³	MicroAeth	Continuous	X	X	X
Ultrafine Particles (UFPs)	(#/cm ³)	CPC 3007	Continuous	X	X	X
Volatile Organic Compounds (VOCs)	(µg/m ³)	Summa Canister	6.5 h	X	X	X
Carbon Dioxide (CO ₂), Temperature, Relative Humidity	(ppm, °C, %)	Vaisala GMW21D	Continuous	X		X
Wind Speed		Vantage Pro2 Weather Station	Continuous		X	

opening, science experiments, and arts/crafts. Monitors were placed on a desk/table in the classroom to be representative of student’s breathing height. Outdoor monitors were located away from busy areas including idling zones and remained in the same location throughout the study. At each location, some or all of the following traffic-related air pollutants were measured: fine particles (FPs; particles of aerodynamic diameter <2.5 µm), black carbon (BC), ultrafine particles (UFPs; particles of aerodynamic diameter 20–100 nm), and volatile organic compounds (VOCs). Some pollutants were site specific, although in most cases pollutants were measured at all three locations (Table 1). Other air parameters [such as temperature, carbon dioxide (CO₂), and relative humidity] were also measured to ensure that occupant comfort was not affected by the intervention.

Information regarding school characteristics and daily activities influencing indoor air quality was collected via four questionnaires. A walk-through log was completed prior to the start of the study to ensure that all HVAC equipment was functioning properly. A baseline questionnaire was used to collect information about the school, such as the number of students, the size of the school, school facilities, and indoor sources of pollutants. A daily questionnaire was used to collect information from school staff regarding activities that took place in the school that could affect daily air quality (e.g., window opening, science experiments, cooking, etc.). Finally, a HVAC questionnaire was completed by the Ottawa-Carleton District School Board (OCDSB), which collected information on the specific characteristics of each HVAC system in place at each school.

Approval was obtained from Health Canada’s Research Ethics Board and the Ottawa-Carleton Research Advisory Committee.

School selection and recruitment

Four schools were chosen from Ottawa, ON to take part in this study from a previously developed dataset containing the distance to roadways of all public elementary schools in 10 Canadian urban areas (Amram et al., 2011). To be included in this study, schools were required to meet the following inclusion criteria; governed by the OCDSB; located within 200 m of a

major road; have a functioning HVAC system that is controlled by a centrally located computer; have one central HVAC system operating in the area of the school being monitored; not conducting renovations in the school at the time of sampling; and finally, have live monitoring of electricity and gas consumption in place to track energy costs associated with the intervention.

The final four schools were chosen from the list of eligible schools based on their willingness to participate and the traffic volume/density at the school. Recruitment for this study was mediated by the OCDSB.

Implementation of the intervention

Typical HVAC operations in public buildings such as schools entail turning off the ventilation system in the evening once the building is no longer occupied, and then restarting the normal operation of the system in the morning approximately 1 h prior to planned occupancy to purge the stale air and bring in fresh air. For most schools, this occurs during morning rush hour. In each of the four schools, the intervention comprised of the HVAC systems being started prior to rush hour traffic. For Ottawa, the timing of rush hour was confirmed by examining daily trends in nitrogen dioxide levels collected by the National Air Pollution Surveillance Network. Based on this timing, the HVAC systems in the Ottawa schools were re-started 1 h prior to rush hour and then turned off from the beginning of morning rush hour until the systems were required to be operational for occupation of the building. The specific schedule was as follows: For all schools, the HVAC system was taken out of nighttime mode to flush the building air between 5:30 and 6:30 AM, followed by a period of recirculation until school began. As two of the schools were early (8:00 AM)-start schools and two others late (9:00 AM)-start schools, they resumed normal ventilation programming based on their start times; early-start schools (SCH-002 and SCH-003) resumed ventilation at 8:00 AM, while late-start schools (SCH-001 and SCH-004) resumed ventilation at 9:00 AM.

The filters for the HVAC systems were changed prior to both the baseline and intervention time periods to ensure that observed changes were not a result of different filtration efficiencies.

Indoor and outdoor air exposure measurements

Continuous monitoring equipment. DustTraks (model 8520, TSI, St Paul, MN, USA), Microaethalometers (MicroAeth AE51, AethLabs, San Francisco, CA, USA), and condensation particle counters (CPC 3007, TSI, ST Paul, MN, USA) were used to continuously monitor FPs, BC, and UFPs, respectively. Both DustTraks and Microaethalometers collected measurements at 1-min intervals. As measurement of UFPs could not occur continuously over a 24-h period due to limitations of the sampling equipment, two instruments of each type were used (each collecting 15 1-min samples every hour). This sampling schedule allowed us to accurately capture exposures occurring during occupied hours and also capture diurnal fluctuations in pollutant concentrations over a 24-h period. Note that DustTraks are an optically-based measurement of fine particulates, and thus not directly comparable to a gravimetric measurement. All values reported are direct DustTrak readings. The results of reduced major axis (RMA) regression with gravimetric measurements we collected suggest that a division of all DustTrak readings by 2 will give a reasonable estimate of the PM_{2.5} mass concentration in $\mu\text{g}/\text{m}^3$. See section S2.3 in the Supplemental Material for more details of this analysis and the gravimetric measurements.

Changes to the timing of the HVAC system operation were hypothesized to potentially alter the temperature, relative humidity, and CO₂ levels in the schools. Therefore, these parameters were measured throughout the study to better understand the effects of this intervention on thermal comfort. These parameters may also impact the infiltration of outdoor pollutants into the indoor environment. CO₂, temperature, and relative humidity were collected at 1-min intervals using Vaisala GMW21D monitors (Vaisala Instruments, Helsinki, Finland).

Local meteorological conditions were also measured continuously (every 10 min) from the school roof using Vantage Pro2 weather stations (Davis Instruments Corp, Hayward, CA, USA).

VOCs. Summa canisters (6.0 l) were used to monitor traffic-related VOC concentrations including benzene, toluene, ethylbenzene, and *m,p*-xylene. VOC sampling occurred only during the time when children were present (6.5 h), to accurately represent children's exposures while in the school environment. Summa canisters were analyzed by gas chromatography–mass spectrometry (GC-MS) according to US EPA method TO-15 (US EPA, 1999).

Quality assurance

Continuous measurement methods. During fieldwork, all instruments were assessed for drift and were zeroed

daily. After the data were collected, all data from continuous instruments underwent a rigorous visual inspection. All outdoor values were plotted concurrently to visually inspect for bias in the instruments, instrument malfunction, or abnormal peaks. Where necessary, measurements were invalidated. Illogical values were also flagged and recoded to missing. Flags were used to identify the reason for any changes between the original and corrected variables. The readings from the two co-located CPC monitors were also compared and where disagreements occurred, concentrations were invalidated.

All instruments used in the field were also compared before and after the study. The intercomparisons involved exposing all instruments to a well-mixed laboratory atmosphere at various pollutant concentrations for multiple hours. The bias of the individual instruments (relative to the mean of all instruments) was calculated, and the bias-corrected precision was determined using the equation:

$$1 - \text{minute precision} = \text{Abs}(A - M)/M \quad (1)$$

where A is the 1-min average for the instrument of interest (after correction for bias) and M is the mean of all instruments for that minute. The overall precision is then the mean of the 1-min precision measurements. The intercomparisons were carried out for four types of continuous instruments (DustTraks, CPC 3007s, Microaethalometers and AeroQual500).

Questionnaire data. Questionnaire data were reviewed daily by a technician, and any inconsistencies were clarified with the participating schools. Logic checks were also built into the online questionnaire. These checks improved error detection and permitted the research team to immediately correct any errors. Detailed information regarding the QAQC methods and results for all instruments can be found in the Supplemental Material.

Statistical analysis

Descriptive statistics for pollutant concentrations. School day averages for the continuously measured pollutants were calculated between the hours of 8:00 AM and 2:30 PM, or 9:00 AM and 3:30 PM depending on the school-specific hours. At least 75% data completeness was required to calculate a daily average. In the case of UFPs, this criterion resulted in large data losses when only one of the two instruments each measuring for a different 15-min period was functional during each hour. As an examination of the two instruments during all hours for which the 75% criterion was met revealed good agreement, we relaxed this criterion for UFPs; specifically, we required that at least one

monitor in a given hour met the 75% completeness criterion.

Efficacy of the intervention. A general linear model using an indicator variable for intervention status was used for each school to test the efficacy of the intervention. Estimates were considered statistically significant when the *P*-value was <0.05. Both unadjusted and adjusted models were examined. Adjusted models included outdoor pollutant concentrations, absolute indoor–outdoor temperature differences (Emmerich et al., 2004; Wallace et al., 2002), and window opening (where sufficient window opening was reported) as covariates. Given the similarities in pollutant concentrations between the HVAC and the front door site, the HVAC site was chosen to be representative of outdoor concentrations. In the rare situation where air pollution data from the HVAC site were missing, the front door concentration was used to impute the missing data. Tables S2–S8 for the detailed descriptive statistics by location.

A generic equation capturing the model used for these analyses is presented below.

$$\begin{aligned}
 (\ln) Y_i = & \beta_o + \beta_1(\text{intervention status}) \\
 & + \beta_2(\text{outdoor concentration}) \\
 & + \beta_3(\text{absolute indoor-outdoor temperature} \\
 & \text{difference}) + \beta_4(\text{open windows}) + \varepsilon_i
 \end{aligned}
 \tag{2}$$

Here Y_i represents the measured pollutant concentration of interest; β_o signifies the regression intercept; β represents the effect of X (intervention, outdoor concentration, absolute indoor–outdoor temperature difference, or open windows) on Y ; and ε_i represents the random error.

We had originally planned to also adjust for indoor sources and wind speed. However, there were virtually no indoor sources reported on the questionnaires, and wind speed was correlated with both the outdoor pollutant concentrations and the absolute indoor–outdoor temperature difference. Given that wind speed did not change the parameter estimates by more than 10%, it was not included in the final models. Cooking was expected to be an indoor source of particles at all schools. However, staff rooms were minimally equipped (microwave, kettle, toaster) and located away from the monitored classrooms. In addition, none of the selected schools had student cafeterias. One major indoor source (welding during maintenance) was reported on 1 day inside SCH-003. This event resulted in abnormally high FP concentrations (>900 $\mu\text{g}/\text{m}^3$) inside the school, and therefore, this day of data was excluded from the final models.

Normality and equal variance assumption was assessed based on the residuals of the model using the Anderson Darling test for normality (Stephens, 1974) and Levene’s test for homogeneity (Levene, 1960). Modeling was conducted on the log-transformed data for all pollutants, because the assumptions were not satisfied using the original scale of the data.

Multicollinearity within the data was assessed by examining the tolerance, variance inflation factors (VIF), and condition indices of the full models. The effects of influential points were examined in all models, as well as plots of the residuals.

Where possible, models were run on the entire school day (8:00 AM–2:30 PM or 9:00 AM–3:30 PM) and the morning hours only. However, as the models were very similar between the two time periods, only the entire day models are presented here. All analysis was carried out in SAS EG 4.2 (SAS Institute, Cary, NC, USA).

Diurnal trends. Figures representing the diurnal trends in pollutant concentrations were completed using Statistica v10 (Statsoft Inc, Tulsa, OK, USA). Averages were calculated for 30-min time periods using the data collected from all monitoring weekdays. Indoor and outdoor concentrations during both the intervention and the baseline time periods were overlaid to easily compare the results.

Estimating the infiltration efficiency (F_{inf}) and the outdoor/indoor components of particulate matter size fractions. The infiltration efficiency (F_{inf}) is defined as the fraction of the outdoor air particle concentration that penetrates the building envelope and remains suspended in indoor air. It is governed by the penetration efficiency (P), the air exchange rate (a), and the decay rate (k) by the following equation (Wilson et al., 2000):

$$F_{\text{inf}} = (Pa)/(a + k)
 \tag{3}$$

To calculate F_{inf} for UFPs and BC, we used the ratio of the ‘censored’ indoor concentration to the outdoor concentration (Censored I/O ratio). This method removes (censors) any indoor-generated peaks from the half-hourly time series. This method has been explained in detail in several peer-reviewed publications (Kearney et al., 2011, 2014; MacNeill et al., 2012, 2014). For FPs, the constant resuspension of settled particles due to the movement of children throughout the day invalidated the use of the censoring method. Therefore, the ambient (outdoor) and non-ambient (indoor-generated) components were not estimated for FPs.

For UFPs and BC, F_{inf} estimates were used to quantify the contribution of ambient and non-ambient particles to total indoor concentrations. The ambient c-

Table 2 School and HVAC Characteristics

Characteristic	SCH-001	SCH-002	SCH-003	SCH-004
School start time	9:00 AM	8:00 AM	8:00 AM	9:00 AM
Intervention randomization	First	Second	First	Second
Year school built	1910 (ren. 1997)	1951	1991	1974
Year HVAC equipment installed	1997	1951	1991	1974
Distance to major road or highway	7 m	126 m	196 m	90 m
Average Annual Daily Traffic Count (500 m) (n)	9 645	n/a ^a	n/a ^a	22 925
Density of major roads or highway (meter road length/m ² in a 200 m buffer)	0.0029	0.0007	0.0016	0.0029
Number of students	770	171	336	185
Age of students	4 (JK) – 14 (Grade 8)	4 (JK) – 11 (Grade 6)	4 (JK) – 11 (Grade 6)	4 (JK) – 11(Grade 6)
Number of students in monitored classroom	19	20	29	16
Location of monitored classroom	3 rd floor classroom	Main floor classroom	2 nd floor classroom	2 nd floor classroom
Number of above grade stories	4	2	3	2
Exterior material	Brick	Brick	Brick	Brick
HVAC intake location:	Roof	2 nd floor	2 nd floor	Ground level
Filters in use:	100 filters (MERV=8)	25 filters (MERV=11)	80 filters (MERV=11)	28 filters (MERV=11)
Outdoor air intake rate/exhaust rate:	5050 L/s	3000 L/s	1800 L/s	4575 L/s
Conditioned volume:	29 502 m ³	5644 m ³	10 444 m ³	7 558 m ³
Design air exchange rate (h-1):	0.62	1.91	0.62	2.18
Unique factors	Underground parking garage	Located near transitway	Exhaust through bathroom vents (not on timer)	Boys and girls club location

^an/a: not available. There was no traffic count monitoring location nearby, and the closest monitoring stations were not representative of traffic at the schools.

component was calculated by multiplying the daily F_{inf} estimate by the daily outdoor concentration. The non-ambient component was calculated by subtracting the daily ambient component estimate from the daily total indoor concentration. Where the ambient component exceeded the total indoor concentration, the ambient component was set to the total indoor concentration and the indoor-generated component was set to zero.

Results and discussion

Description of schools and HVAC systems

School and HVAC characteristics are presented in Table 2. The four schools included in this study varied in a number of important characteristics, including age, size, student population, and their proximity to a major roadway. The oldest HVAC system was installed in 1951 and the newest in 1997. These systems covered volumes ranging in size from 5 644 m³ to 29 502 m³ and had varying numbers and ratings of filters, and design air exchange rates as presented in Table 2. SCH-001 had the largest student population with 770 students ranging from Junior Kindergarten to Grade Eight as well as an underground parking garage. Road proximity ranged between 7 and 196 m. The location of the sampling equipment also varied depending on where an appropriate site could be located.

Pollutant concentrations

Pollutant concentrations are presented in Figure 1 (FPs, UFPs, and BC) and 2 (BTEX species—benzene, toluene, ethylbenzene, and xylenes). Overall, indoor

and outdoor particulate air pollutant concentrations were lower than what has been reported elsewhere (Kim et al., 2014; McCarthy et al., 2013; Mullen et al., 2011; Raysoni et al., 2013; Reche et al., 2014; Rivas et al., 2014; Weichenthal et al., 2008). UFP concentrations were lower indoors compared to outdoor concentrations. FP and BC concentrations were more similar inside and outside, but in most cases were higher outdoors (apart from the day when welding was taking place inside SCH-003; FP data from this event were removed from the analyses) (Figure 1). BTEX concentrations were higher indoors than outdoors (Figure 2). Indoor concentrations of BTEX species were similar to levels found in other school studies (Godwin and Batterman, 2007; Raysoni et al., 2013). Tables with the descriptive statistics for the air pollutants of interest can be found in the supplemental material (Tables S2–S8).

Efficacy of the intervention

A summary of the reductions in the adjusted and unadjusted geometric mean concentrations can be found in Table 3. Detailed model results can be found in the supplemental material (Tables S12–S18).

Statistically significant reductions between baseline and intervention periods were observed at both late-start schools during occupied periods (SCH-001 and SCH-004; *adjusted models*). Specifically, BTEX species were reduced by 34–42% in SCH-001 and 18–23% in SCH-004. FPs were reduced by 43% at SCH-001, and UFPs were reduced by 34% at SCH-004. Although measured FPs at SCH-004 were 40% lower during the intervention period, this was not statistically significant

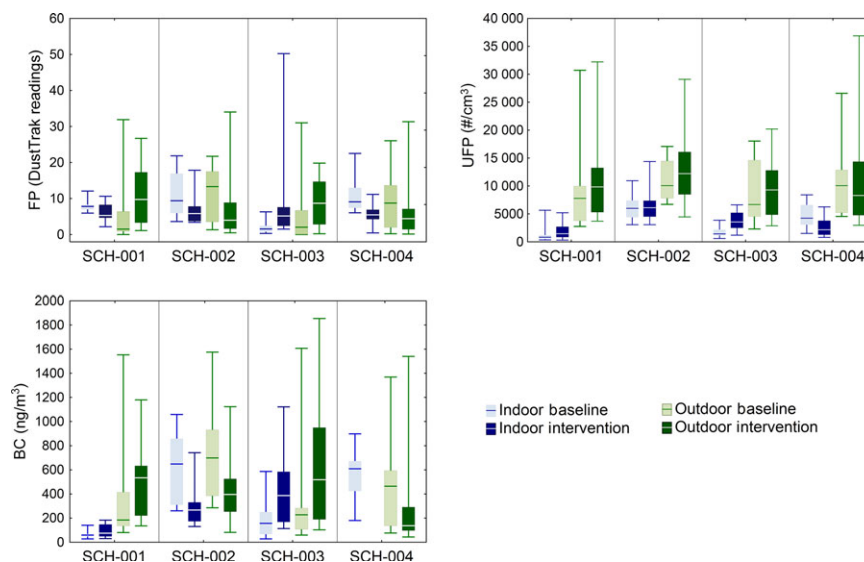


Fig. 1 Concentrations of fine particles (FPs), ultrafine particles (UFPs) and black carbon (BC) by school and intervention status. Boxplots show medians, 25–75 percentiles (box) and min-max (whiskers)

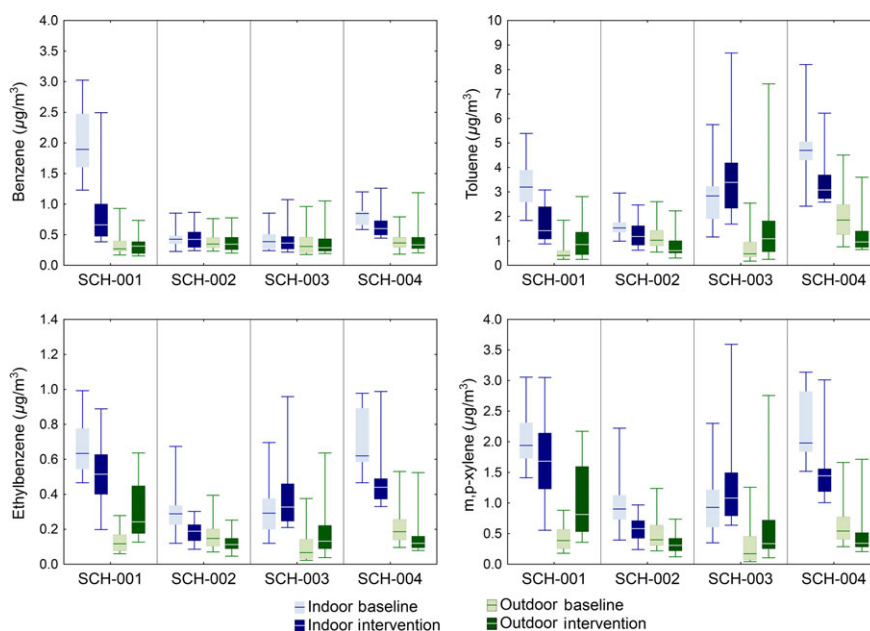


Fig. 2 Concentrations of BTEX species (benzene, toluene, ethylbenzene, and m,p-xylene) by school and intervention status. Boxplots show medians, 25–75 percentiles (box) and min-max (whiskers)

($P = 0.10$) and UFPs at SCH-001 were essentially unchanged (increase of 4%, $P = 0.91$). Neither of the early-start schools (SCH-002, SCH-003) had statistically significant changes in pollutant concentrations between baseline and intervention. Early-start schools resumed ventilation at 8 AM when outdoor pollution levels are more elevated due to rush hour as compared with late-start schools, which resumed ventilation at 9 AM when rush hour is over. However, marginally significant ($P < 0.10$) reductions in both BC (of 26%, $P = 0.06$) and ethylbenzene (of 27%, $P = 0.06$) were observed at SCH-002.

To our knowledge, no studies have examined the efficacy of changing the timing of the ventilation in schools or other public buildings. However, studies have examined the influence of other modifications to the HVAC system on indoor air quality in schools. A Nevada school intervention study tested the effectiveness of installing improved filtration systems into the existing HVAC systems. Before the improvements, the HVAC system reduced indoor BC levels by 31–66% relative to outdoor air. After the intervention, BC levels were reduced by 74–97% relative to outdoor air (McCarthy et al., 2013). Reductions of 87–96% were also found in BC, UFP, and $PM_{2.5}$ in a California

Table 3 Summary of reductions in geometric mean pollutant concentrations during baseline and intervention time periods.

SCHOOL	Pollutant	Entire School Day-Adjusted ^a				Entire School Day-Unadjusted			
		B ^b	I ^c	% Change ^d	P-value	B ^b	I ^c	% Change	P-value
SCH-001 (9 AM start)	FP (DustTrak reading)	8.7	5.0	-43	0.00	7.7	5.7	-26	0.02
	UFP (#/cm ³)	1101	1146	4	0.91	877	1481	69	0.06
	BC (ng/m ³)	68	64	-6	0.72	57	78	37	0.10
	Benzene (μg/m ³)	1.65	0.95	-42	0.01	1.93	0.75	-61	<0.001
	Toluene (μg/m ³)	2.97	1.72	-42	0.00	1.16	0.44	-62	<0.001
	Ethylbenzene (μg/m ³)	0.69	0.45	-35	0.00	0.65	0.48	-26	0.01
SCH-002 (8 AM start)	m,p-xylene (μg/m ³)	2.13	1.4	-34	0.01	1.97	1.56	-21	0.07
	FP (DustTrak reading)	8.1	6.9	-15	0.22	9.8	5.8	-41	0.01
	UFP (#/cm ³)	5275	6547	24	0.18	5848	5973	2	0.87
	BC (ng/m ³)	446	331	-26	0.06	594	268	-55	<0.001
	Benzene (μg/m ³)	0.43	0.4	-7	0.48	0.43	0.41	-5	0.69
	Toluene (μg/m ³)	1.44	1.26	-13	0.38	0.47	0.15	-68	0.03
SCH-003 (8 AM start)	Ethylbenzene (μg/m ³)	0.26	0.19	-27	0.06	0.29	0.18	-38	0.00
	m,p-xylene (μg/m ³)	0.79	0.63	-20	0.15	0.92	0.56	-39	0.00
	FP (DustTrak reading)	2.1	2.9	38	0.22	1.4	4.4	214	0.00
	UFP (#/cm ³)	1815	2741	51	0.12	1610	3212	100	0.00
	BC (ng/m ³)	223	256	15	0.69	144	358	149	0.00
	Benzene (μg/m ³)	0.37	0.4	8	0.45	0.39	0.4	3	0.91
SCH-004 (9 AM start)	Toluene (μg/m ³)	2.41	3.57	48	0.14	0.93	1.23	32	0.10
	Ethylbenzene (μg/m ³)	0.27	0.38	41	0.13	0.29	0.37	28	0.20
	m,p-xylene (μg/m ³)	0.85	1.24	46	0.15	0.91	1.2	32	0.16
	FP (DustTrak reading)	8.8	5.3	-40	0.10	10.3	5.0	-51	0.00
	UFP (#/cm ³)	3736	2471	-34	0.03	4034	2201	-45	0.01
	BC (ng/m ³)	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	Benzene (μg/m ³)	0.79	0.62	-22	0.00	0.81	0.61	-25	0.00
	Toluene (μg/m ³)	4.31	3.53	-18	0.02	1.55	1.2	-23	0.00
	Ethylbenzene (μg/m ³)	0.62	0.48	-23	0.05	0.68	0.45	-34	0.00
	m,p-xylene (μg/m ³)	1.98	1.55	-22	0.04	2.17	1.45	-33	0.16

Adjusted statistically significant results ($P < 0.05$) are bolded.

^aAdjusted for outdoor pollutant concentrations, absolute I-O temperature difference, and window opening (where sample size permitted).

^bB: Baseline.

^cI: Intervention.

^dPercent Change = ((Intervention-Baseline)/Baseline) * 100.

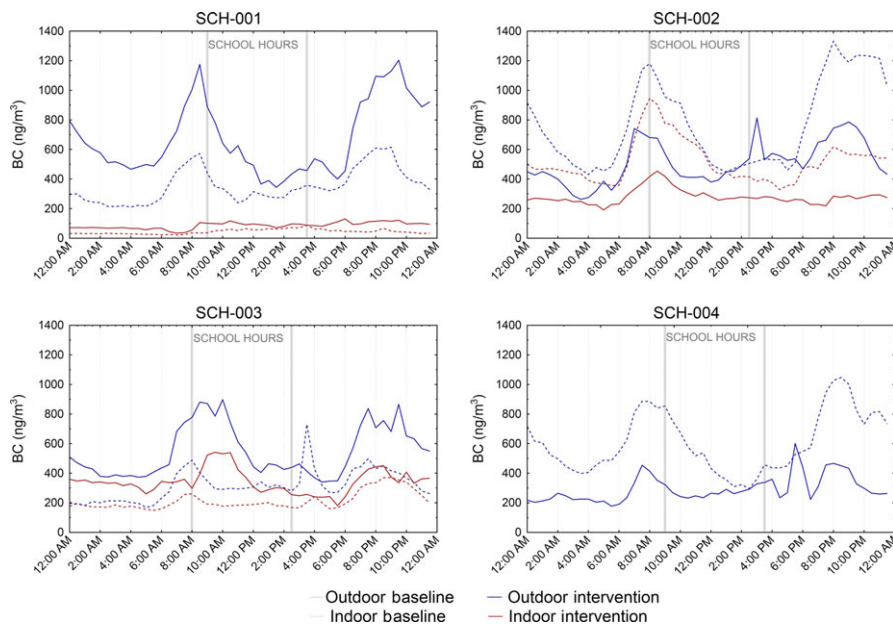


Fig. 3 Diurnal patterns of black carbon (BC) concentrations by school and intervention status

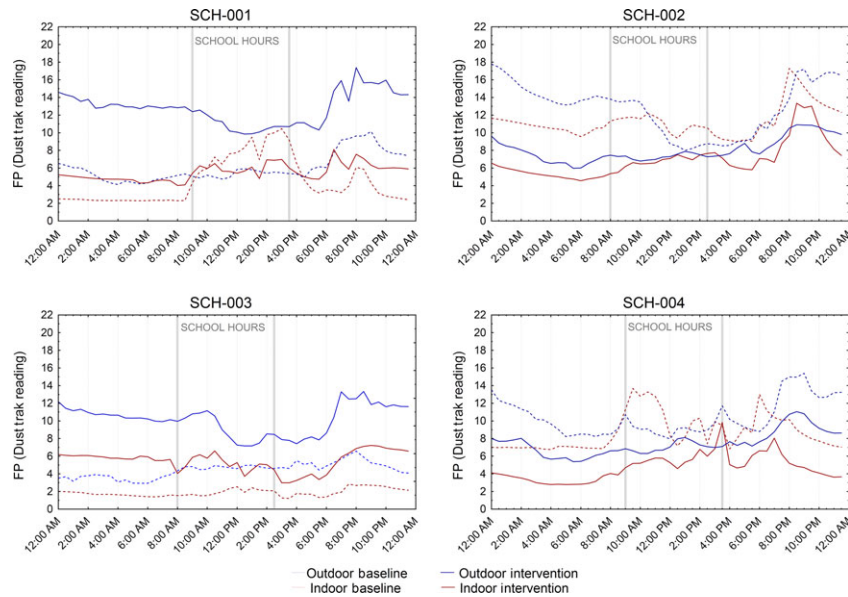


Fig. 4 Diurnal patterns of fine particle (FP) concentrations by school and intervention status

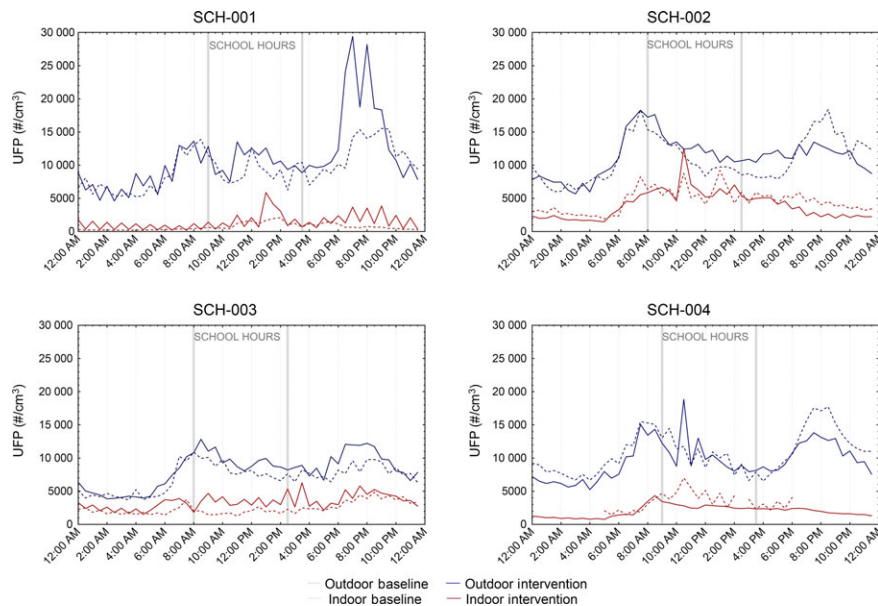


Fig. 5 Diurnal patterns of ultrafine particle (UFP) concentrations by school and intervention status

school study where register-based air purifiers were combined with HVAC systems (Polidori et al., 2013). These options may be additional steps to consider when the HVAC systems must operate during higher pollution periods (i.e., early-start schools).

Diurnal patterns

The diurnal patterns for BC, FPs, and UFPs for each school can be found in figures 3, 4, and 5, respectively. BC, FPs, and UFPs showed very different diurnal patterns, with BC showing the strongest associations with traffic periods. The diurnal trends observed for BC also follow the hypothesized results most closely. At

SCH-001, there was a dip of roughly 50 ng/m^3 in BC concentrations during the recirculation period, despite high ($>500 \text{ ng/m}^3$) outdoor concentrations during this time (Figure 3).

It is also clear that very little ambient BC was able to penetrate the building envelope and remain suspended at this school, as levels remained low (geometric means of 57 and 78 ng/m^3 during the baseline and intervention periods, respectively) and did not follow outdoor concentration fluctuations. At SCH-002, indoor BC concentrations followed outdoor concentrations more closely. During the baseline time period, the indoor levels increased steeply with the outdoor increase to a peak of approximately 925 ng/m^3 . However, during the inter-

vention, the indoor increase was attenuated and only rose to approximately 425 ng/m³. At SCH-003, BC increased only when the HVAC system started actively drawing in outdoor air. In all schools, the outdoor BC concentrations were higher (by about 125 ng/m³ overall) than the indoor concentrations. This is consistent with results reported by McCarthy et al. (2013).

For FPs, there was no clear morning rush hour peak, which is not unexpected given that only components of FPs, such as BC, are considered to be traffic-related. Therefore, although FP concentrations decreased at SCH-001 and SCH-004 (late-start schools) during the intervention, the mechanism is unclear.

For UFPs, outdoor concentrations began to increase earlier than anticipated. This may partially explain why the intervention was generally not effective at reducing this pollutant. Although validation is required, it is possible that restarting the HVAC systems earlier may be more effective in reducing indoor UFPs. However, given that the UFP concentrations increased throughout rush hour, the intervention still should have resulted in some reductions in indoor UFP concentrations. The diurnal trends in ambient particle concentrations observed in this study are consistent with what has been reported elsewhere (Buonocore et al., 2009; Mishra et al., 2012). Specifically, Buonocore et al., 2009 also found that BC featured a peak during morning rush hour, whereas UFPs demonstrated an earlier increase. They also reported that PM_{2.5} had less diurnal variability than either BC or UFPs.

Diurnal trends in pollutant concentrations have also been reported for several schools in the literature. Mullen et al. (2011) tracked indoor and outdoor UFP levels in six California classrooms and demonstrated that unless there were specific indoor sources, such as candle burning, the indoor levels were typically lower than outdoor. They surmised that higher levels of indoor UFP during occupancy were a result of increased ventilation that coincided with times of elevated outdoor levels. A Barcelona study of 39 schools conducted by Reche et al. (2014) included schools without air conditioning, in which ventilation was achieved via window opening. The traffic levels surrounding the schools significantly influenced the indoor UFP levels especially in the schools where windows opened onto the street rather than onto a courtyard or playground. The authors noted that secondary aerosol formation contributed to increases in ambient UFP levels around midday. This phenomenon was not identified in our study probably because the study was carried out in the fall at a time when windows were mostly closed, but this could be an important source to consider during summer months.

Influence on CO₂ and temperature

Daily CO₂ concentrations and temperature data for occupied periods during both baseline and intervention

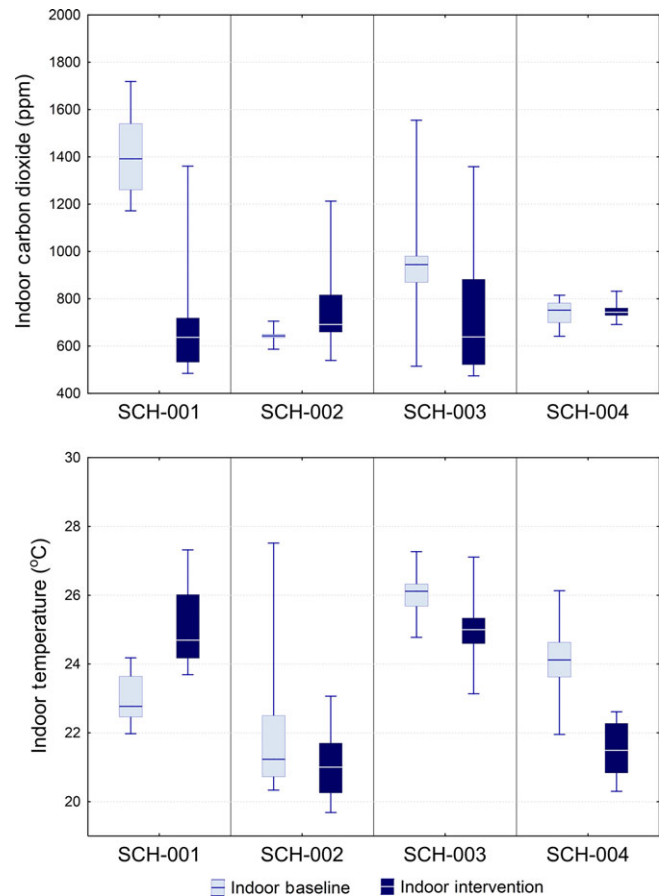


Fig. 6 CO₂ (ppm) and Temperature (°C) during occupied hours by school and intervention status. Boxplots show medians, 25–75 percentiles (box) and min-max (whiskers)

time periods are presented in Figure 6. Tables with the descriptive statistics for these two parameters, as well as relative humidity, can also be found in Tables S9–S11.

These parameters were collected to determine whether the intervention was affecting occupant comfort, although it should be noted that the intervention did not actually provide less ventilation overall, it merely shifted the timing of ventilation. During occupation of the building, the ventilation remained the same for baseline and intervention periods. The intervention did not lead to materially significant increases in CO₂ levels at any of the schools; the largest increase from baseline to intervention was from a geometric mean of 648 ppm to 749 ppm at SCH-002. The American Society of Heating, Refrigerating and Air Conditioning Engineers (ASHRAE, 2013) Standard 62.1-2013 recommends indoor level of CO₂ be no more than 700 ppm above the outdoor level, a standard that was met during the intervention for all schools. During the baseline period, CO₂ levels at school SCH-001 did exceed 700 ppm above outdoor levels, suggesting that there was either a source during this time or that the ventilation levels were below guidance provided by

Table 4 Descriptive statistics for daily F_{inf} estimates by school and intervention status

School	Pollutant	Intervention Status	N ^a	Min	P25	Median	P75	Max	Mean (95% CI)	Geometric Mean (95% CI)
SCH-001 (9 AM start)	FP	Baseline	17	0.04	0.22	0.32	0.47	0.68	0.35 (0.26, 0.44)	0.29 (0.20, 0.42)
		Intervention	15	0.01	0.18	0.35	0.52	1.17	0.40 (0.24, 0.55)	0.27 (0.14, 0.52)
	UFP	Baseline	18	0.03	0.06	0.09	0.12	0.54	0.11 (0.06, 0.17)	0.09 (0.06, 0.12)
		Intervention	16	0.03	0.06	0.13	0.21	0.30	0.14 (0.10, 0.18)	0.11 (0.08, 0.16)
	BC	Baseline	17	0.09	0.15	0.22	0.39	0.59	0.25 (0.18, 0.32)	0.21 (0.16, 0.29)
		Intervention	16	0.08	0.12	0.14	0.19	0.24	0.15 (0.13, 0.18)	0.15 (0.12, 0.17)
SCH-002 (8 AM start)	FP	Baseline	14	0.00	0.78	0.85	0.95	1.13	0.77 (0.58, 0.95)	0.77 (0.60, 1.01)
		Intervention	17	0.02	0.42	0.73	0.91	0.98	0.65 (0.49, 0.81)	0.49 (0.29, 0.84)
	UFP	Baseline	16	0.35	0.46	0.52	0.58	0.69	0.52 (0.47, 0.56)	0.51 (0.46, 0.56)
		Intervention	18	0.20	0.36	0.44	0.48	0.79	0.43 (0.37, 0.50)	0.42 (0.36, 0.48)
	BC	Baseline	13	0.41	0.75	0.78	0.86	0.94	0.77 (0.68, 0.85)	0.75 (0.65, 0.87)
		Intervention	17	0.10	0.57	0.63	0.69	0.95	0.61 (0.52, 0.71)	0.60 (0.52, 0.71)
SCH-003 (8 AM start)	FP	Baseline	18	0.29	0.38	0.49	0.56	0.84	0.50 (0.43, 0.57)	0.48 (0.42, 0.55)
		Intervention	16	0.24	0.34	0.54	0.83	1.17	0.58 (0.43, 0.73)	0.51 (0.39, 0.68)
	UFP	Baseline	17	0.11	0.17	0.22	0.28	0.55	0.25 (0.19, 0.31)	0.23 (0.18, 0.28)
		Intervention	14	0.19	0.29	0.32	0.35	0.84	0.36 (0.27, 0.45)	0.34 (0.28, 0.42)
	BC	Baseline	15	0.04	0.42	0.60	0.71	0.96	0.56 (0.42, 0.70)	0.50 (0.36, 0.70)
		Intervention	14	0.05	0.45	0.57	0.77	0.90	0.57 (0.43, 0.71)	0.58 (0.48, 0.70)
SCH-004 (9 AM start)	FP	Baseline	14	0.45	0.51	0.64	0.72	1.14	0.67 (0.55, 0.78)	0.64 (0.55, 0.75)
		Intervention	17	0.00	0.53	0.67	0.86	1.11	0.68 (0.53, 0.83)	0.68 (0.55, 0.84)
	UFP	Baseline	15	0.25	0.28	0.39	0.46	0.77	0.41 (0.32, 0.50)	0.38 (0.32, 0.47)
		Intervention	18	0.15	0.19	0.27	0.32	0.42	0.26 (0.22, 0.30)	0.25 (0.22, 0.29)
	BC	Baseline	7	0.65	0.66	0.80	0.82	0.86	0.77 (0.69, 0.84)	0.86 (0.75, 0.98)
		Intervention	0	–	–	–	–	–	–	–

^aNumber of daily monitoring events of 6.5 h duration each.

AHSRAE. However, it was noted that the CO₂ levels dropped significantly (to a geometric mean of 682 ppm) during the intervention; the cause of this difference in CO₂ levels is unknown as the building was similarly occupied during the intervention and baseline periods, and no other changes to the HVAC system were made. Both SCH-001 and SCH-003 were much newer schools compared to SCH-002 and SCH-004, and were therefore likely more tightly sealed buildings. This is supported by the infiltration factors (as discussed in The Infiltration Efficiency (F_{inf}) and the Ambient and Indoor-Generated Component Estimates).

Temperature also did not seem to be adversely affected. There were some differences in average indoor temperature between the baseline and intervention periods, but indoor temperature changes were likely the result of changes in outdoor temperature. We minimized the potential impacts of changing outdoor temperatures on the study results by alternating the order of the baseline and intervention periods between schools, and by adjusting the results for the indoor-outdoor temperature difference. No complaints regarding occupant comfort were received during the study.

The infiltration efficiency (F_{inf}) and the ambient and indoor-generated component estimates

Descriptive statistics for daily F_{inf} estimates are presented in Table 4. Median F_{inf} estimates across all schools ranged from 0.32 to 0.85 for PM_{2.5}, 0.09 to

0.52 for UFPs, and 0.14 to 0.78 for BC. The lowest F_{inf} estimates were observed at SCH-001, and the highest F_{inf} estimates were observed at SCH-002. As only four schools were included in this study, the factors influencing this variability could not be determined. However, it is likely that the age of the buildings as well as differences in the HVAC systems' filtration systems contributed to these differences (MacNeill et al., 2012).

In general, UFP F_{inf} was about half that for FP and BC. This has been reported previously and has been attributed to the lower penetration rates due to Brownian deposition and higher deposition velocities (Hinds, 1999; Kearney et al., 2011, 2014; Rivas et al., 2015). With little infiltration of UFPs, the intervention was unlikely to change indoor UFP levels. In this study, we found that PM_{2.5} F_{inf} was comparable to BC F_{inf} , or in some instances lower. Previous research has reported higher or comparable F_{inf} for BC compared with PM_{2.5} and has been attributed to the non-volatile nature of BC (Lunden et al., 2003a,b; MacNeill et al., 2012; Sarnat et al., 2006). However, few studies have examined the infiltration of BC and substantial regional or seasonal differences may exist.

No statistically significant differences were observed in F_{inf} between baseline and intervention time periods. As this intervention was not expected to increase air exchange, deposition, or penetration, these results were expected.

The majority of indoor UFP and BC concentrations were of ambient origin in all schools (see Tables 5 and S18 for details). The median percentage of indoor

Table 5 Descriptive statistics for percent ambient contribution to indoor UFP and BC

School	Pollutant	Intervention Status	N ^a	Min	P25	Median	P75	Max	Mean (95% CI)	Geometric Mean (95% CI)
SCH-001 (9 AM start)	UFP	Baseline	18	4	82	92	97	100	81 (68, 95)	70 (47, 102)
		Intervention	16	15	71	92	95	100	78 (62, 83)	68 (49, 95)
	BC	Baseline	17	59	70	82	92	100	81 (74, 88)	80 (73, 87)
		Intervention	16	44	68	85	92	99	79 (70, 88)	77 (68, 87)
SCH-002 (8 AM start)	UFP	Baseline	16	51	91	100	100	100	92 (85, 100)	91 (82, 101)
		Intervention	18	43	79	93	99	100	84 (75, 94)	82 (71, 94)
	BC	Baseline	13	56	83	90	96	100	87 (79, 95)	86 (77, 95)
		Intervention	17	24	81	93	97	100	85 (75, 95)	81 (68, 97)
SCH-003 (8 AM start)	UFP	Baseline	17	80	93	99	100	100	96 (93, 99)	96 (93, 99)
		Intervention	14	37	96	99	100	100	93 (84, 103)	91 (78, 106)
	BC	Baseline	15	4	84	90	100	100	85 (72, 98)	73 (46, 116)
		Intervention	14	8	75	98	100	100	85 (70, 100)	64 (39, 106)
SCH-004 (9 AM start)	UFP	Baseline	15	35	81	88	91	96	83 (74, 91)	81 (70, 93)
		Intervention	18	84	95	99	100	100	97 (95, 99)	97 (95, 99)
	BC ^b	Baseline	7	46	53	68	78	86	66 (52, 79)	64 (52, 79)

^aNumber of daily monitoring events of 6.5 h duration each.

^bBC instrumentation failure during intervention.

UFPs that were of ambient origin ranged from 88 to 100%. The median percentage of indoor BC of ambient origin ranged from 82 to 98% at the three schools with complete BC data. SCH-004 employed a BC monitor in part of the baseline period and all of the intervention period that was found to be defective, and therefore, the somewhat lower median value of 68% for a less than complete baseline period should be viewed with caution. This is consistent with the questionnaire data that indicated very few indoor sources. Table S18 for the descriptive statistics for the UFP and BC components.

To date, studies have not examined the ambient-/indoor-generated contribution of UFPs and BC in schools. However, while not directly comparable to our data, several studies have included indoor–outdoor ratios (I/O) of particulate matter levels in schools (Laiman et al., 2014; Mullen et al., 2011; Rivas et al., 2015).

Mullen et al. (2011) showed UFP indoor–outdoor ratios increased to 0.59 during student presence compared to 0.41 when the classrooms were unoccupied. In both scenarios, it is evident that outdoor sources were the dominant source. Rivas et al. (2015) found a similar median I/O UFP ratio of 0.66 (ranging from 0.17 to 1.62) across 39 schools in Barcelona. Laiman et al. (2014) were able to identify a number of specific UFP sources—namely printing, heating, and cleaning—within 25 Australian primary schools that resulted in elevated I/O ratios.

The Ottawa-Carleton District School Board reported that costs associated with implementing this intervention were negligible. Overall, any costs associated with changing the HVAC system timing were offset by off-peak energy use incentives. Therefore, this intervention is a very cost-effective mechanism for improving air quality in schools.

Limitations

Limitations of this study include the small number of schools sampled and the fact that sampling occurred during only one season. Sampling in a greater number of schools and in more seasons would allow for further investigation of the factors that influence the efficacy of the intervention (*e.g.*, window opening, HVAC system characteristics, classroom location, etc.). Measurements were also only taken in one classroom. While it was intended to represent the indoor air quality of the school serviced by the HVAC system, it is possible that the school was not well-mixed and localized sources of pollutants could influence air quality. In addition, all schools were located in Ottawa, ON where ambient concentrations were relatively low in comparison with other cities. However, our results suggest that this intervention would be beneficial to locations with high ambient concentrations. Future studies should collect more detailed information on HVAC system characteristics to better understand the applicability of these results to other locations.

Conclusions

The results of this study indicate that changing the timing of ventilation can practically and significantly reduce traffic-related pollutant concentrations in late-start schools located near major roads without negatively impacting occupant comfort and at minimal cost. However, this intervention was limited in its ability to make significant improvements to the indoor air quality in early-start schools, which must resume ventilation at 8 AM when outdoor pollution levels are more elevated due to rush hour. Future research should evaluate the impact of this intervention with a larger number of schools in a more

polluted environment and include measurements of health impacts.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Table S1. Percentage (%) of invalid half hourly averages.

Table S2. Descriptive statistics for FP concentrations during occupied hours ($\mu\text{g}/\text{m}^3$) (DustTrak display readings).

Table S3. Descriptive statistics for UFP concentrations during occupied hours ($\#/ \text{cm}^3$).

Table S4. Descriptive statistics for BC concentrations during occupied hours (ng/m^3).

Table S5. Descriptive statistics for benzene concentrations during occupied hours ($\mu\text{g}/\text{m}^3$).

Table S6. Descriptive statistics for toluene concentrations during occupied hours ($\mu\text{g}/\text{m}^3$).

Table S7. Descriptive statistics for ethylbenzene concentrations during occupied hours ($\mu\text{g}/\text{m}^3$).

Table S8. Descriptive Statistics for m,p-xylene concentrations during occupied hours ($\mu\text{g}/\text{m}^3$).

Table S9. Descriptive Statistics for CO_2 concentrations during occupied hours (ppm).

Table S10. Descriptive Statistics for temperature during occupied hours ($^{\circ}\text{C}$).

Table S11. Descriptive Statistics for relative humidity during occupied hours (%).

Table S12. Detailed model results for FPs.

Table S13. Detailed model results for UFPs.

Table S14. Detailed model results for BC.

Table S15. Detailed model results for benzene.

Table S16. Detailed model results for toluene.

Table S17. Detailed model results for ethylbenzene.

Table S18. Detailed model results for m,p-xylene.

Table S19. Descriptive statistics for UFP and BC of ambient and non-ambient origin.

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