Title
Characterizing Ultrafine Particle Exposures in Two Types of Indoor Environments: San Francisco Bay Area Classrooms and Beijing High-Rise Apartments

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ABSTRACT

Characterizing Ultrafine Particle Exposures in Two Types of Indoor Environments: San Francisco Bay Area Classrooms and Beijing High-Rise Apartments

by

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Ultrafine particles are defined as those particles having a diameter of 100 nm or less. They are emitted by both indoor and outdoor sources and are ubiquitous in the environment. Epidemiological studies have indicated that ultrafine particle (UFP) exposures are associated with adverse health outcomes, and toxicological studies have suggested that this is mechanistically possible. At present, the mass concentrations of fine particles having a diameter of 2.5 µm or less (i.e. PM$_{2.5}$) are monitored and regulated in most developed countries in the world. However, UFP concentrations correlate poorly with PM$_{2.5}$ concentrations; thus, the extensive outdoor PM$_{2.5}$ data available cannot be used to draw inferences regarding UFP exposure concentrations, and efforts to reduce PM$_{2.5}$ levels cannot be expected to reduce UFP levels. While a growing number of studies have investigated UFP concentrations both indoors and outdoors over the last two decades, there remain many microenvironments in which UFP exposures have not been characterized. In this dissertation, UFP exposure concentrations are characterized and the factors influencing those concentrations are explored, within two microenvironments that had hitherto not been investigated: San Francisco Bay Area elementary school classrooms and Beijing high-rise apartments.

Children between the ages of 6 and 11 years old living in California spend an average of 10% of their time in school, second only to the amount of time spent at home (53%). In addition, children are considered to be more susceptible to some health effects resulting from pollutant exposures than are adults. To contribute towards a characterization of children’s exposure to ultrafine particles, a field study was conducted in six classrooms in the San Francisco Bay Area. The purpose of this study was to provide data regarding children’s UFP exposures in school classrooms, the contributions of indoor and outdoor sources to those exposures, and the influence of building parameters and occupant behaviors on those exposures. Additional aims were to characterize the classroom ventilation rates, and to explore the balance between maintaining adequately high ventilation for the removal of bioeffluents and other indoor emitted pollutants, while also seeking to limit the indoor proportion of outdoor particles (IPOP). The data collection phase of this study involved monitoring particle number
(PN) concentrations and the concentrations of three gaseous co-pollutants (CO₂, NO, O₃) for two to four school days in each classroom. Time-resolved data on classroom ventilation characteristics and occupant activities were recorded using temperature and state-change sensing data loggers, and by a researcher who was present in the classroom for the duration of the school day. In all, 18 days of data were collected from June to December 2008.

The average indoor PN concentration during periods of student occupancy in the six classrooms ranged from 5.2 \times 10³ to 16.5 \times 10³ cm⁻³. Indoor sources had a relatively small influence on classroom PN concentrations, with only three significant source events detected during periods of student occupancy across the six classrooms. For this small sample of admittedly limited scope, the classrooms monitored in warmer months (i.e., June through early November) had both a higher outdoor and indoor average PN concentration during periods of student occupancy than those monitored during colder months (i.e., late November and early December). This higher exposure to outdoor generated particles during warm months was influenced by more frequent opening of doors and windows for the purpose of maintaining a comfortable temperature in the classroom. The mean daily-integrated UFP exposures of the students while in their classrooms was 50 \times 10³ cm⁻³ h d⁻¹, which was approximately a factor of 6 less than the mean exposure calculated in a parallel study for a sample of children in San Francisco Bay Area homes. The higher daily-integrated exposure experienced by children in homes is partly attributable to the higher PN concentrations measured in homes during hours of occupancy than in schools, and partly a result of the greater time that children spend in their home on a daily basis as compared to their classrooms. For these classrooms, outdoor PN concentrations measured on-site appear to be a good indicator of the relative exposure concentrations encountered by students within their classrooms. The utility of outdoor data for predicting exposures indoors depends critically on the dominance of outdoor air as the source of indoor PN levels.

The time-weighted average air-exchange rate for the six classrooms ranged from 1.1 to 10.8 h⁻¹, and the accompanying range for the rate of ventilation per person was 4 to 27 L/s. Two of the classrooms utilized mechanical ventilation systems, while four were ventilated by means of doors and windows. In the case of the naturally ventilated classrooms, the ventilation rate generally exceeded the standard specified by the American Society of Heating Refrigerating and Air-Conditioning Engineers (ASHRAE) when doors and/or windows were in an open state, but often fell below the standard otherwise. For the mechanically ventilated classrooms, the air-exchange rate appeared unnecessarily high in one case and too low in the other. Results from five of the six sites were analyzed to see if an increase in the air-exchange rate was accompanied by an increase in the IPOP; for four of the classrooms the data were so correlated. However, reducing the air-exchange rate as a strategy for decreasing the indoor level of outdoor generated particles is not recommended, and instead strategies were investigated for reducing the IPOP using active filtration.

The work presented here suggests that outdoor sources may be a more important contributor than indoor sources to UFP concentrations in Bay Area classrooms. Therefore, strategies to reduce classroom UFP concentrations may be most effective if focused on decreasing the IPOP. The classroom air-exchange rate results indicate that teachers in naturally ventilated classrooms should be encouraged to keep windows and/or
doors in the open state during periods of student occupancy to maintain adequate ventilation. In classrooms with mechanical ventilation systems, more attention may need to go towards ensuring that the classroom ventilation rate is neither too high nor too low. Since the IPOP is expected to and seen to increase with an increase in the air-exchange rate, it is recommended that strategies to increase classroom ventilation be accompanied with active filtration, either via portable fan-filter air cleaners or through use of high efficiency in-duct filters. The results presented here were collected from a relatively small sample of sites. Thus, to the extent that children’s exposure to ultrafine particles is considered an issue of concern, these results should be augmented by further research conducted in a larger sample of Bay Area schools.

Roughly 20% of the world’s population lives in China, and yet research groups have only recently begun to investigate UFP concentrations in this region of the world. Studies investigating UFP concentrations in mainland China have thus far focused on the outdoor environment. Since people generally spend the majority of their time indoors, data are needed on the UFP exposure concentrations encountered in indoor microenvironments in China, so that population exposures in mainland China can be accurately characterized. To contribute towards filling this research gap, a field study was conducted in a sample of high-rise apartments in Beijing, one of the largest cities in China, with a population of roughly 20 million. In the past three decades, newly constructed housing developments in Beijing have primarily taken the form of high-rise buildings. The data collection phase of this study involved monitoring PN within four high-rise apartments for two to four days each. For two apartments, outdoor PN data were also collected. Temperature and state-change data loggers were used to record when occupant activities involving heat (e.g., cooking) were conducted and when door and window positions were changed, respectively. The residents also maintained a journal of their activities and the hours they were present at home. In all, ~9 days of time-series data were collected.

Distinct indoor PN peaks independent of outdoor concentrations were observed on twenty-seven occasions during monitoring at the four apartments. Cooking was responsible for the majority of the observed indoor PN peaks. In one apartment, although the residents cooked infrequently themselves, a large number of indoor peaks appeared to result from the infiltration of emissions from cooking in neighboring apartments. The average indoor PN concentrations at the four apartments ranged from $2.8 \times 10^3$ to $29.1 \times 10^3$ cm$^{-3}$. The apartment with the highest indoor concentration was influenced by the neighbors cooking, and the apartment with the lowest concentration only experienced two indoor PN peaks in two days and had two portable fan-filter air cleaners that operated almost continuously. For the apartments where outdoor PN data were also collected, 58% and 81% of the residents’ total UFP exposure while at home was attributed to outdoor sources. Conversely, in a study of seven single-family homes in the San Francisco Bay Area, an average of 30% of the residents’ exposure was attributed to outdoor sources. The greater indoor exposure to outdoor particles in the former case is expected to have resulted from the higher outdoor concentration during hours the residents spent asleep, the larger fraction of time the residents spent at home and the greater use of natural ventilation. Particle emission rates were calculated for some of the cooking events in the Beijing apartments, and the average was almost identical to the average calculated for natural gas cooking events in the study of Bay Area homes.
The results from this study indicate that cooking makes a significant contribution to exposure in some Beijing apartments. However, outdoor generated particles make a larger contribution overall. An issue that requires further attention is whether or not cross-contamination between apartments is common in Beijing high-rise buildings. This study provides initial results for UFP concentrations in Beijing residences, and elucidates some of the factors that influence exposures in high-rise apartments. It is important that this study be reinforced with further study of different aspects of UFP exposure both in high-rise buildings and other types of residential structures in Beijing and throughout China, so that the UFP exposures of population living in China can be characterized, and, if necessary, controlled.
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Chapter 1. Introduction

This dissertation contributes towards characterizing UFP concentrations and exposures in indoor environments, as well as the factors that influence these parameters. My research specifically focuses on two types of indoor microenvironments in two different geographical areas: elementary schools in Northern California and high-rise apartments in Beijing, China. The analysis presented in this dissertation is based on data collected from two field studies: the first in six San Francisco Bay Area elementary school classrooms, and the second in high-rise buildings in Beijing. The next three sections highlight the motivation for studying these topics and the background information that frames these investigations. The final section of this chapter provides a narrative outline of the dissertation.

1.1 Ultrafine particles: characteristics and concerns

Since the early 1990’s, evidence has existed linking ambient particle mass (PM) concentrations and adverse human health outcomes (Pope and Dockery, 2006). In response, the mass concentration of fine particles having a diameter of 2.5 µm or less (PM$_{2.5}$) was added to the list of criteria air pollutants regulated in the United States (under the National Ambient Air Quality Standards), as well as in many other countries. At present, PM$_{2.5}$ concentrations are monitored in most, if not all, developed countries, and in many developing countries, around the world.

Ultrafine particles are defined as those particles having a diameter of 100 nm or less in size. They commonly comprise the majority of airborne particles when evaluated by number concentration (typically 90% or more); however, because of their relatively small size, they generally make up less than 10% of ambient particle mass concentrations. As a result, ultrafine particle (UFP) concentrations correlate poorly with PM$_{2.5}$ concentrations. Ultrafine particles are therefore not controlled under PM$_{2.5}$ standards, nor can their levels be inferred from the extensive PM$_{2.5}$ monitoring networks that have been instituted around the world.

The first evidence of health effects related to UFP exposure came from toxicological studies, as summarized by Hoek et al. (2010). In the past two decades, the number of toxicological studies investigating different aspects of ultrafine particle toxicity has grown, and several review papers published in the last decade have summarized their results (Donaldson et al., 2001; Ibald-Mulli et al., 2002; Oberdörster et al., 2005; Kennedy, 2007; Knol et al., 2009). Although ultrafine particles make up a minority of mass concentrations, their higher particle number and surface area per unit mass are suggested to be of toxicological importance. Evidence also indicates that ultrafine particles inhibit phagocytosis, and this combined with their small size increases their potential to cross the alveolar epithelium and enter the circulatory system, from which they can be transported to other organs, such as the liver, spleen or heart (Nemmar et al., 2002; Geiser et al., 2005). In addition, there is evidence that ultrafine particles trigger an inflammatory response in the lung that is greater than that of larger particles per unit mass, which has potential to eventually result in a cardiac event (Donaldson et al., 1998; Li et al., 2003). Finally, it has been suggested that ultrafine particles may exert adverse biological effects by the presence of radicals on their surfaces, which is suggested particularly to be the case for combustion particles (Donaldson et al., 1996; Dick et al., 2003).
Relative to the number of toxicological studies that have been conducted, there
have been few epidemiology studies investigating health effects associated with UFP
exposures. The initial epidemiological studies were panel studies investigating short-term
health effects among susceptible populations, specifically, individuals with asthma or
chronic obstructive pulmonary disease (COPD). In reviewing the published scientific
literature, I identified seven panel studies reported in eight published papers that
investigated associations of the concentrations of both ultrafine particles and other
particle size classes with specific health end points among susceptible populations
(Pekkanen et al., 1997; Peters et al., 1997; Tiitanen et al., 1999; Penttinen et al., 2001;
Osunsanya et al., 2001; von Klot et al., 2002; de Hartog et al., 2003; Timonen et al.,
2006), six of which were reviewed by Ibald-Mulli et al. (2002). All of these studies were
conducted in European cities and all used centrally monitored pollutants as the exposure
indicator. Five of these studies used peak expiratory flow rate (PEF) as an investigated
health end point. Of these five, three found that PEF was inversely associated with both
fine and ultrafine particles; in two cases the association with ultrafine particles was
stronger (Peters et al., 1997; Penttinen et al., 2001), and in one case the association with
fine particles was stronger (Pekkanen et al., 1997). One study found an inverse
association of PEF with PM$_{2.5}$ and PM$_{10}$, but not ultrafine particles (Tiitanen et al.,
1999), while another failed to see an association with either ultrafine particles or PM$_{10}$
(Osunsanya et al., 2001). Four of the seven studies characterized medication use or other
disease symptoms as health endpoints. Two of these studies found that both fine and
ultrafine particles were associated with at least one of the investigated health endpoints
(von Klot et al., 2002; de Hartog et al., 2003); one study found no significant association
of any particle size class with health endpoints (Penttinen et al., 2001); and one study
found that PM$_{10}$ was significantly associated with health endpoints, while UFP
concentrations were not (Osunsanya et al., 2001). One of the seven studies also
investigated associations of different particle size classes with an indicator of heart rate
variability (HRV); investigators found that ultrafine particles had a clear association with
HRV in three cities, whereas PM$_{2.5}$ did not (Timonen et al., 2006).

Six studies published in seven papers have investigated short-term associations of
fine and ultrafine particle concentrations with hospital admissions and/or mortality
(Wichmann et al., 2000; Stölzel et al., 2007; Halonen et al., 2008, 2009; Andersen et al.,
2008, 2010; Atkinson et al., 2010). All of these studies were conducted in European
cities and all used centrally monitored pollutant data as exposure indicators. The
analyzed time-series data sets ranged from three to six years in duration. Four of the
studies investigated associations of pollutant concentrations with daily mortality
(Wichmann et al., 2000; Stölzel et al., 2007; Halonen et al., 2008, 2009; Atkinson et al.,
2010), and four investigated associations of pollutant concentrations with hospital
admissions (Halonen et al., 2008, 2009; Andersen et al., 2008, 2010; Atkinson et al.,
2010). Two of the studies investigated associations for different populations based on
age (Andersen et al., 2008; Halonen et al., 2008, 2009). Three of the four studies
reported a positive association of either total or ultrafine particle number concentrations
with mortality (Wichmann et al., 2000, Stölzel et al., 2007; Atkinson et al., 2010). Of the
four studies investigating hospital admissions, two found that both ultrafine particles and
PM$_{2.5}$ or accumulation mode particles were associated with asthma-related hospital visits
among children, whereas only accumulation and coarse mode particle size fractions were
associated with respiratory—or COPD—related hospital visits among adults or the elderly (Andersen et al., 2008; Halonen et al., 2008). Halonen et al. (2009) also found that both PM_{2.5} and the number concentration of Aitken mode particles (defined by the authors as 30-100 nm in diameter) were associated with arrhythmia hospital admissions among the elderly, although cardiorespiratory hospital admissions among the elderly were most strongly associated with accumulation mode particles (defined by the authors as 0.1-0.29 µm in diameter). Atkinson et al. (2010) found that particle number (PN) concentrations were associated with daily hospital admissions of cardiovascular disease, whereas non-primary PM_{2.5} concentrations were more important for respiratory outcomes. Andersen et al. (2010) found that UFP concentrations were significantly associated with hospital admissions due to ischemic and mild strokes, whereas PM_{10} concentrations were not.

To date, there have been no epidemiological studies investigating possible health effects associated with chronic ultrafine particle exposures.

In 2008, an expert elicitation was conducted in Europe to address the possibility of specifying ultrafine particle concentration-response functions for two key health endpoints: mortality and hospital admissions (Hoek et al., 2010). The experts’ estimates for the percentage decrease in all-cause mortality with a permanent 1000 particles/cm^3 decrease in ambient ultrafine particle concentration ranged from 0.1 to 1.2%, with a median of 0.30%. The experts reported that the lack of studies of measured health effects resulting from long-term exposure to ultrafine particles was the most important contributing factor to the overall uncertainty. Estimated effects on cardiovascular and respiratory hospital admissions were considered more uncertain than mortality.

In summary, although there is substantial toxicological research providing evidence of the biological plausibility of health effects elicited by ultrafine particle exposure and the physiological responses that result, there is insufficient epidemiological data to fully validate these toxicological findings. Specifically, the results from epidemiological studies investigating short-term exposures are mixed, although they primarily support a hypothesis that UFP exposure poses health risks. In addition, there is a lack of any epidemiological evidence regarding the effects of chronic exposures. Since the debate over the health risks of PM_{2.5} exposure was significantly strengthened by cohort studies investigating long-term health effects (Pope and Dockery, 2006), there is reason to believe that studies of health effects associated with chronic ultrafine particle exposures may make an important contribution towards understanding the potential public health risk posed by ultrafine particles in outdoor air.

The number of studies investigating UFP sources and concentrations has increased in recent years, as instrumentation has become better able to detect submicron particles and as studies that indicate adverse health risks from UFP exposures have accumulated. The majority of studies characterizing UFP concentrations have focused on the outdoor environment. Table 1.1 provides a summary of the average UFP concentrations and predominant UFP sources observed by studies conducted in urban areas, in which concentrations were measured for a year or more. Studies conducted in urban areas around the world have identified atmospheric nucleation and motor vehicle emissions as two major contributors to UFP concentrations in outdoor air. Atmospheric nucleation events are characterized by a burst of nanometer scale particles resulting from gas to particle conversion processes, and are hypothesized to result from the interactions
of H₂O, H₂SO₄ and possibly other gaseous species (Weber et al., 1996; Ball et al., 1999; Kulmala et al., 2004; Zhang et al., 2004; Sipilä et al., 2010). These events have been observed to occur more frequently during spring and summer months with peak concentrations generally occurring midday, which has been suggested to be related to the higher solar radiation intensity during these periods (Wehner and Wiedensohler, 2003; Jeong et al., 2004; Jeong et al., 2010; Fine et al., 2004; Alam et al., 2003). Studies have shown that these events occur more or less uniformly in air masses that extend distances of several hundred kilometers (Kulmala et al., 2004).

Ultrafine particles emitted by motor vehicles generally consist of elemental carbon particles from fuel-rich combustion, and sulfate and organic particles resulting from the oxidation and subsequent nucleation of volatilized compounds from the engine fuel and lubricating oil (Zhu et al., 2002a; Gramotnev and Ristovski, 2004; Kittelson et al., 2006). UFP concentrations resulting from motor vehicle emissions have been found to be highest during winter months, particularly in the morning hours. It has been suggested that the higher emissions on cold mornings is partly the result of exhaust gases having a lower vapor pressure in colder temperatures, which reduces the energy barrier for particle nucleation (Kittelson et al., 2000; Charron and Harrison, 2003; Virtanen et al., 2006). In addition, the morning hours of the winter months are often characterized by a low mixing height and light winds, which would tend to increase the peak concentration of the emitted pollutants (Stanier et al., 2004; Jeong et al., 2004; Harrison and Jones, 2005). Unlike nucleation events, particle concentrations resulting from motor vehicle emissions are spatially heterogeneous. Studies have shown that PN concentrations near busy roadways decrease by a factor of 3-5 between a distance of ~20 m to ~150 m from the roadside (Shi et al., 1999; Zhu et al., 2002a; Zhu et al., 2002b; Gramotnev and Ristovski, 2004). This relatively high degree of spatial variation in the concentration of locally emitted ultrafine particles calls into questions whether concentrations can be characterized by sparsely deployed central monitors. Hudda et al. (2010) investigated this issue by measuring PN concentrations simultaneously at seven sites in the Los Angeles air basin, and they found the measurements agreed moderately well. Consequently, they suggest that before central monitoring data is used to interpret the PN concentrations of a large area, an understanding should be gained of the spatial variability in that area and of seasonal and diurnal influences on that variability.
<table>
<thead>
<tr>
<th>City</th>
<th>Dates (mo/y)</th>
<th>Mean concentration (particle size range)</th>
<th>Suggested main sources</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Atlanta, GA</td>
<td>8/1999 – 7/2000</td>
<td>36,100 cm$^{-3}$ (3-10 nm) 36,400 cm$^{-3}$ (10-100 nm)</td>
<td>Vehicle emissions, atmos. nucleation</td>
<td>McMurry and Woo, 2002</td>
</tr>
<tr>
<td>Erfurt, Germany</td>
<td>9/1997 – 8/2001</td>
<td>12,600 cm$^{-3}$ (10-100 nm)</td>
<td>Vehicle emissions</td>
<td>Yue et al., 2008</td>
</tr>
<tr>
<td>Helsinki, Finland</td>
<td>5/1997 – 3/2001</td>
<td>7000 cm$^{-3}$ (&lt;30 nm) 6500 cm$^{-3}$ (20-100 nm) 1000 cm$^{-3}$ (90-400 nm)</td>
<td>Vehicle emissions</td>
<td>Hussein et al., 2004</td>
</tr>
<tr>
<td>Rochester, NY</td>
<td>12/2001 – 12/2002</td>
<td>5,800 cm$^{-3}$ (11-50 nm) 1,150 cm$^{-3}$ (50-100 nm) 880 cm$^{-3}$ (100-470 nm)</td>
<td>Vehicle emissions, atmos. nucleation</td>
<td>Jeong et al., 2004</td>
</tr>
<tr>
<td>Toronto, Canada</td>
<td></td>
<td>28,000 cm$^{-3}$ (&gt;7 nm)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pittsburgh, PA</td>
<td>7/2001 – 6/2002</td>
<td>22,000 cm$^{-3}$ (3-500 nm)</td>
<td>Vehicle emissions, atmos. nucleation</td>
<td>Stanier et al., 2004</td>
</tr>
<tr>
<td>Long Beach Riverside</td>
<td>1/2002 – 12/2003</td>
<td>~21,000 cm$^{-3}$ ~19,000 cm$^{-3}$ ~21,000 cm$^{-3}$ ~20,000 cm$^{-3}$ ~17,000 cm$^{-3}$ ~12,000 cm$^{-3}$ (all 14-700 nm)</td>
<td>Vehicle emissions, atmos. nucleation, long range transport</td>
<td>Singh et al., 2006 (Concentrations approximated from results reported in Figure 2)</td>
</tr>
<tr>
<td>Copenhagen, Denmark</td>
<td>5/2001 – 12/2004</td>
<td>8,100 cm$^{-3}$ (6-700 nm)</td>
<td>Vehicle emissions, long range transport</td>
<td>Andersen et al., 2008</td>
</tr>
<tr>
<td>Beijing, China</td>
<td>3/2004 – 2/2006</td>
<td>9,000 cm$^{-3}$ (3-20 nm) 15,900 cm$^{-3}$ (20-100 nm) 7,800 cm$^{-3}$ (0.1 – 1μm)</td>
<td>Vehicle emissions, atmos. nucleation</td>
<td>Wu et al., 2008</td>
</tr>
</tbody>
</table>
While information regarding outdoor UFP sources and concentrations provides an important contribution towards characterizing human exposure, outdoor monitoring data alone does not provide the full picture of exposures that individuals actually experience. In fact, people spend a small minority of their time outdoors. Specifically, a study in the United States found that Americans spend approximately 8% of their time outdoors, while 69% is spent at home, 6% is spent in a vehicle and the remaining 17% is spent in other indoor locations (Klepeis et al., 2001). Likewise, a study in China found that adults living in urban areas of Chongqing spend approximately 13% of their time outdoors, while 60% is spent at home, 1% is spent in transit and the remaining 22% is spent in other indoor locations (Wang et al., 2008b). The UFP exposures that individuals experience in indoor microenvironments differ from those encountered outdoors, depending on the extent to which outdoor particles are able to penetrate and persist within the indoor environment and on the extent to which particles are emitted from indoor sources. Consider for example, a study conducted by Vinzents et al. (2005), in which 15 healthy nonsmoking adults carried a PN monitoring instrument for six 18-hour periods, which included a 20 km bicycle-ride during peak traffic and an overnight stay at home. At the end of each period, researchers measured the amount of oxidative DNA base damage in the circulating mononuclear blood cells of the study subject. On average, during the 18-h period, the study subjects spent 11% of their time bicycling, 7% of their time outdoors not bicycling and 82% of their time indoors. The resulting time-integrated UFP exposure (min x UFP/cm³) had a mean apportionment of 20% from periods of bicycling, 10% from other outdoor periods and 70% during periods indoors. Therefore, the majority of exposure occurred indoors, primarily because of the larger amount of time spent there. In addition, the authors found that the measured oxidative DNA base damage was associated with personal exposure to UFP concentrations, with cumulative outdoor and indoor exposures contributing independently to the association. Importantly, no association was found with ambient concentrations of air pollutants simultaneously measured at a centrally located monitoring station. This study demonstrates an important point: to accurately characterize human exposure to UFP concentrations, it is important to understand the exposures that occur within microenvironments, particularly those where individuals spend the most time.

Several studies have been conducted in the United States to investigate sources and concentrations of ultrafine particles in indoor environments. Studies conducted in residences have found that occupant-related source activities, such as cooking, cleaning, and use of candles can increase the indoor UFP concentration by as much as an order of magnitude above background levels (Long et al., 2000; Wallace, 2006; Santen et al., 2009; Bhangar et al., 2011). Other studies have observed increases in the UFP concentrations in office spaces resulting from the use of laser printers or photocopiers (He et al., 2007; Lee and Hsu, 2007). Two studies conducted in school classrooms observed increases in UFP concentrations resulting from cleaning and art activities (Morawska et al., 2009; Guo et al., 2010). All of these studies found that outdoor UFP concentrations also influenced the indoor levels, with the degree of influence depending on characteristics of the building, such as the ventilation mode and rate.

In reviewing the published scientific literature, I was unable to identify any prior studies that sought to quantify the relative contribution of indoor and outdoor sources to the exposure concentrations experienced in classroom microenvironments. In addition, I
did not find any studies investigating UFP concentrations in urban residences in mainland China or school classrooms in northern California. Consequently, an aim of this dissertation is to fill these specific gaps in scientific information, and thereby to contribute towards improved characterization of population exposures to ultrafine particles.

1.2 Ultrafine particles in school classrooms

Wiley et al. (1991) reported that children between the ages of 6 and 11 years old living in California spend an average of 10% of their time in school, second only to the amount of time spent at home (53%). Based on these statistics, one can assume that reducing the average UFP concentration within a child’s place of residence will most significantly reduce that child’s integrated UFP exposure. However, a different understanding emerges when considered from the perspective of population exposure. Specifically, in the United States in the year 2000, the average number of individuals under the age of 18 across all homes was 0.64, and in homes with at least one individual under the age of 18 was 1.92 (US Census Bureau, 2010). Conversely, the average occupancy of children in classrooms within Alameda County in grades 1-8, during the 2008-2009 school year, was 25.4 (Education Data Partnership, 2010). Therefore, assuming the US census data is approximately representative of the conditions in Alameda County, and using time budget data from Wilson et al. (1991), the 24-hour time-weighted average number of students in a classroom is roughly 2.5, while the 24-hour time-weighted average number of individuals under 18 years old in homes where at least one such individual lives is roughly 1.0. This result suggests that if the UFP concentration is reduced by an equivalent amount in both a home and a classroom, the resulting reduction in time-integrated population exposure will be greater in the classroom. In addition, since most children in the US attend schools that are publicly run, an opportunity exists to reduce pollutant exposures for a susceptible segment of the population without having to impose controls on private individuals or institutions.

Reviewing the literature, I identified eight published studies investigating UFP levels in occupied classrooms, primarily carried out over a few days or weeks in only one season: in Pembroke, Ontario (Weichenthal et al., 2008); in Athens, Greece (Diapouli et al., 2007b); in Denmark (Wargocki et al., 2008); in southern Germany (Zöllner et al., 2007; Fromme et al., 2007); and in Australia (Guo et al., 2008; Morawska et al., 2009; Guo et al., 2010). In Pembroke, a model was developed for predicting classroom UFP exposure levels by investigating the primary factors that influence classroom PN concentrations during the hours that school was in session. On average, indoor concentrations were found to be lower than outdoors, indicating the absence of significant indoor sources. Of ten factors investigated, only wind speed and outdoor UFP concentration were found to be important determinants of the indoor UFP concentration. In Athens, UFP levels were measured in seven primary schools and one home, with the aim of investigating children’s UFP exposures. The ratio of indoor to outdoor PN levels (I/O) was found to be less than one in all cases, and classroom concentrations generally decreased with a decrease in traffic density and for lower degrees of urbanization. In Denmark, 190 children (10-12 years old) from five pairs of classrooms were included in a blind crossover study, which investigated whether the use of an electrostatic particle
(ESP) filter in the classroom was associated with improved student performance. The results indicated that while ESP filters significantly reduced indoor PN concentrations, there was no consistent effect on student performance. In southwest Germany, number concentrations of particles 20-100 nm in diameter were measured in 22 schools, over the course of four months. The I/O ratios for UFP in the classrooms in this study were also found to be less than one, and short-term PN fluctuations observed outside of the classroom as a result of traffic were suppressed indoors. In Munich, the number concentrations of particles larger than 10 nm were measured in 36 classrooms, during the summer months. Levels were in the same range as or lower than levels typically observed outdoors, though outdoor levels were not simultaneously measured. In Brisbane, Australia, particles in the size range of 15 nm to 79 nm were measured in three classrooms over two different monitoring campaigns. In one classroom, indoor PN was statistically associated with outdoor PN, although indoor peaks were observed during the early morning and later afternoon hours, presumably when students were not present, as a result of floor polishing and cleaning with liquid detergents. In another classroom in Brisbane, significant indoor peaks were observed as a result of art activities (painting, gluing, drawing). In sum, while these studies contribute to an understanding of children’s UFP exposure in classrooms, there is still much to be learned regarding how this exposure varies among seasons and across geographic regions, how it is affected by occupant activities and ventilation mode, and how exposure in classrooms compares to exposure in other environments where children spend their time.

Related to the aim of characterizing classroom UFP concentrations, in this dissertation I also seek to characterize classroom ventilation conditions. The ventilation rate of an indoor space has a direct effect on indoor UFP levels. If indoor sources are dominant, then increasing the air-exchange rate will serve to reduce the exposure concentrations. Conversely, if outdoor sources dominate, then increasing the air-exchange rate will tend to increase indoor concentrations. Predominant thinking has generally been that having a high classroom ventilation rate is preferable to having one that is low, primarily because of the high rate of bioeffluent generation during peak occupancy. For this reason, the American Society of Heating, Refrigerating and Air-Conditioning Engineers (ASHRAE) suggests in their standard 62.1-2010 that the rate of ventilation per person within a classroom should not fall below 5 L/s, and that the difference between the indoor and outdoor CO₂ level should remain below 700 ppm. No standard for an upper limit on the classroom ventilation rate is specified. These two metrics provided by ASHRAE for assessing the adequacy of ventilation in a classroom—per person supply rate and CO₂ level—were primarily formulated on the basis of providing a minimum level of comfort to the individuals present, by ensuring low levels of bioeffluents. However, several studies have found that low classroom ventilation rates or high CO₂ levels are associated with not just discomfort, but also with decreased health indicators and learning performance.

The indoor CO₂ concentration during occupancy is often used as a surrogate measure for ventilation because of its abundance and ease of measurement. Most studies investigating classroom CO₂ levels and the health or performance of students or staff have found them to be associated to a statistically significant degree. Specifically, Myhrvold et al. (1996) studied 22 classrooms in 5 Norwegian schools that had been recently renovated with the objective of improving indoor air quality and found a
A statistically significant correlation between indoor CO₂ levels and symptoms of headache, dizziness, heavy-headedness, tiredness, difficulty concentrating, and unpleasant odor. They also observed reduced pupil performance on the Swedish Performance Evaluation System test at high concentrations of CO₂. Shendell et al. (2004b) collected CO₂ data inside and outside 409 traditional and 25 portable classrooms from 22 schools located in six school districts in Washington and Idaho. They found that a 1000 ppm increase in the difference between indoor and outdoor CO₂ was associated with a 0.5-0.9% decrease in annual average daily attendance, corresponding to a relative 10-20% increase in student absence. Madureira et al. (2009) investigated associations between health effects and air quality in 76 classrooms in Oporto, Portugal, and found a statistically significant association of fatigue, headache, heavy-headedness and concentration difficulties with the levels of CO₂ and total volatile organic compounds. Conversely, van Dijken et al. (2006) performed a cross-sectional study in 11 classrooms in 11 different schools in the Netherlands, and, although they observed high CO₂ concentrations, they found no association with health effects.

A few studies have investigated associations between classroom air-exchange rate and the health or performance of students or staff, and most of these observed a statistically significant association. Wålinder et al. (1997 and 1998) conducted two studies in schools in Sweden and found that school staff in classrooms both with a low air-exchange rate and with dilution versus displacement mechanical ventilation systems had increased occurrence of reduced nasal patency and inflammatory biomarker response in nasal lavage. Smedje and Norbäck (2000) found that the reporting of asthmatic symptoms was less common among 143 Swedish pupils after their classrooms were equipped with new ventilation systems that resulted in a mean increase in the air-exchange rate of 4 h⁻¹. Haverinen-Shaughnessy et al. (2011) conducted a study involving 100 elementary schools in the southwest region of the US and found a linear association between classroom ventilation rates and student achievement within the range of 0.9 – 7.1 L/s per person. However, Smejde et al. (1997) found in a study of 762 seventh graders from 11 randomly chosen Swedish schools that there was no significant association between current asthma and classroom ventilation mode or air-exchange rate.

The studies cited in the preceding paragraphs indicate that within a certain general range, increasing the air-exchange rate of a classroom results in an improvement of student health and performance. However, when translating this result into strategic action, it should be balanced by the understanding that in the absence of controls, increasing the air-exchange rate will also result in an increase in the indoor concentration of certain outdoor pollutants. Some research has revealed associations between students’ health and indicators of traffic-emitted pollutants in schools. Specifically, Kim et al. (2004) found in a study of 1,109 students in 10 San Francisco Bay Area schools that both asthma and bronchitis symptoms in the past 12 months were associated with outdoor concentrations of all pollutants measured on the school grounds, especially NOₓ, NO and black carbon. Ultrafine particle concentrations were not measured as part of this study. Ana et al. (2009) found in a study of 400 students in eight secondary schools in Ibadan, Nigeria, that proximity of the schools to certain sources and activities such as refuse burning and major roadways seemed to present substantial risk factors for reported respiratory morbidity among the students. One study reported associations between students’ health and both classroom ventilation and traffic-emitted pollutants.
Specifically, Mi et al. (2006) found among 1,414 students in 30 classrooms in Shanghai, China, that concentrations of both indoor CO₂ and outdoor NO₂ were associated with current asthma among students.

Consequently, there is a need for further study of the relationship between classroom air-exchange rate and indoor levels of pollutants, to develop well-informed strategies that balance the use of increased ventilation to remove indoor generated pollutants with the use of control measures to prevent elevated indoor levels of outdoor generated pollutants. Chapter 3 in this dissertation explores the relationship between classroom ventilation conditions and the indoor proportion of outdoor ultrafine particles.

1.3 Air quality in Beijing high-rise apartments

Over the past 70 years, Beijing has experienced significant changes in its residential housing stock. Prior to 1949, the year the country became the People’s Republic of China, two-thirds of Beijing housing was in a dilapidated condition (Hu and Kaplan, 2001). During this time, the majority of Beijing residents lived with their extended family within courtyard style houses lining narrow alleyways, called “hutongs” (Gaubatz, 1995). From 1949 to 1977, 24 million m² of residential living area was constructed in Beijing (Hu and Kaplan, 2001), in addition to renovation of the existing housing stock, which included subdividing courtyard houses into multiple units (Gaubatz, 1995). Housing was generally built near workplaces, and the goods and services required by the residents were integrated into the neighborhood (Hu and Kaplan, 2001). The year 1978 marked the beginning of economic reforms in China that resulted in rapid economic growth, as evidenced by changes in the gross domestic product, which has had an annual growth rate of approximately 10% for the majority of years over the last three decades (http://databank.worldbank.org). This economic growth has coincided with many other changes. For example, from 1980 to 2005, the urban population in China increased from 19.6% to 40.5%, and the total energy consumption roughly tripled (Chan and Yao, 2008). Within Beijing, from 1978 to 1990, the average urban resident’s wages increased by a factor of 27, and individuals were not only able to exercise more choice regarding where they lived, but an increasing amount of new housing was made available commercially by property developers (Hu and Kaplan, 2001). Most of the newly constructed housing developments have taken the form of groups of high-rise structures that include parks, playgrounds and other public facilities (Gaubatz, 1995).

In parallel to these changes in the Beijing housing stock, air quality in Beijing, both indoors and outdoors, has also undergone change over the last few decades. Since 1978, the vehicle growth rate in Beijing has stayed above 10% per year for most years, and has increased to roughly 15% per year since 1997 (Hao and Wang, 2005). The number of on-road vehicles in Beijing reached the one million mark in 1997, and from 2007 to 2009 increased from three to four million (Wu et al., 2011). In the 1990’s, Beijing was listed as one of the world’s top 10 most polluted cities, and in 1998 the Beijing municipal government began to publish weekly air quality reports (Hao and Wang, 2005). The average concentrations of total suspended particles (TSP), SO₂, NOₓ and CO during the heating season in 1998 were 431, 252 and 201 µg/m³ and 4.4 mg/m³, respectively. The highest hourly O₃ concentration that year was 384 µg/m³ (Hao and Wang, 2005). These levels are high relative to urban norms in Western Europe and the United States (www.epa.gov/airtrends; www.eea.europa.eu/themes/air/airbase/map-
The pollutant concentrations in Beijing at the end of the 1990’s exceeded the national air quality standards of China and, in the cases of TSP and SO₂, were more than twice the standard (Table 1.2). As a result of these high pollutant concentrations, in December 1998, the Beijing municipal government implemented “comprehensive emergency control measures” to reduce pollutant levels (Hao and Wang, 2005). By 2002, the annual average SO₂ concentrations in Beijing had declined by 44%, and there were a higher number of O₃ attainment days (i.e., days for which hourly O₃ concentrations remained below 200 μg/m³), relative to 1998. NOₓ levels decreased from 1998 to 2000, but then rose slightly with the growth of the vehicle fleet, and the levels of PM did not change noticeably (Hao and Wang, 2005). By 2005, the annual average levels of PM₁₀, SO₂ and NO₂ were 141, 50 and 66 μg/m³, respectively, in Beijing (http://english.mep.gov.cn/standards_reports). In preparation for the 2008 Olympic games, the Beijing municipal government implemented a series of control measures to further reduce pollutant concentrations. These included traffic controls prohibiting high emitting vehicles from being on the road and allowing lower emitting vehicles on the road only every other day. Other control measures included enhanced utilization of natural gas to replace coal for electric generation and heat, reduction in local power plant generation by importing electricity from surrounding areas, strict dust control at construction sites, closure or relocation of polluting industrial plants, and the construction of 148.5 km of new subway lines (Wang et al., 2009a). During the summer of 2008, the light-duty gasoline vehicles fleet average emissions of BC, CO and UFP decreased by 33%, 47% and 78%, respectively, and the average UFP emissions of heavy-duty diesel vehicles decreased by 67%, compared to the same period in the summer of 2007 (Wang et al., 2009a). During the 2008 Olympics, the mean concentrations of SO₂, NO₂ and the maximum O₃ 8-h mean were 12.5, 23 and 114 μg/m³, respectively, which represented reductions of 51%, 13% and 20%, respectively, compared to the months immediately preceding the Olympics (Xin et al., 2010). The PM₁₀ and PM₂.₅ concentrations during the Olympics were 82 and 65 μg/m³, respectively, which were 35% and 31% lower than the mean concentrations for the weeks immediately before and after the Olympics (Wang et al., 2009b). Even with the reductions, the PM₁₀ concentration in Beijing during the 2008 Olympics was 1.9 – 3.5 times higher than the concentrations during the previous three summer Olympics, hosted in Atlanta, Sydney and Athens (Wang et al., 2009b). The temporary pollutant controls in Beijing were lifted in late September 2008, and the levels of all pollutants subsequently increased (Wang et al., 2009b; Xin et al., 2010). Nevertheless, the mean PM concentration in Beijing for October-December 2008 was lower than the mean for the same period in 2007, which Wang et al. (2009b) attribute to a 20% reduction in traffic (some of the traffic control measures have not been lifted to date), along with a downturn in the overall economy and so in the rate of energy use.
Indoor air quality in urban China has evolved as characteristics of both the country’s outdoor air quality and indoor environmental factors have changed. Research on indoor air quality in China began in the late 1970’s, with an initial focus on occupational health and safety. In 1979, the Chinese government established the “Hygienic Standard for Industrial Enterprises Designing,” which regulated concentrations of hazardous air pollutants emitted by different production processes in workplaces and residential areas (Bai et al., 2003). In the 1990’s, as living standards were improving rapidly, many individuals in urban environments began to purchase their own apartments or townhouses (Bai et al., 2003). As a result, the remodeling and interior decorating market grew, and large amounts of synthetic decorative products and interior finishes were made available for purchase, often before being carefully examined for potential emissions of hazardous gaseous pollutants (Wang et al., 2004). In parallel, the Chinese government realized the need to offset the rapid increase in energy consumption, and in 1987 the Ministry of Construction released draft standards for energy efficiency of new residential buildings, and thus a movement was made towards constructing buildings that were more air-tight (Lang and Huang, 1993). This combination of increased indoor VOC emissions from building products and decreased residential air-exchange rates exacerbated the indoor pollutant exposure conditions. Researchers in China began to conduct studies of indoor VOC sources and concentrations, and in 1995, the Ministry of Health released the first standards for concentrations of hazardous air pollutants in civil

### Table 1.2. Ambient air quality standards in the United States and China

[http://www.epa.gov/air/criteria.html; Hao and Wang, 2005](http://www.epa.gov/air/criteria.html)

<table>
<thead>
<tr>
<th>Pollutant</th>
<th>Averaging Time</th>
<th>US Standard&lt;sup&gt;a&lt;/sup&gt;</th>
<th>China Standard (Grade II)&lt;sup&gt;b&lt;/sup&gt;</th>
</tr>
</thead>
<tbody>
<tr>
<td>SO₂</td>
<td>Annual</td>
<td>80 µg/m³</td>
<td>60 µg/m³</td>
</tr>
<tr>
<td></td>
<td>Daily</td>
<td>365 µg/m³</td>
<td>150 µg/m³</td>
</tr>
<tr>
<td>NO₂</td>
<td>Annual</td>
<td>100 µg/m³</td>
<td>80 µg/m³</td>
</tr>
<tr>
<td>O₃</td>
<td>8-hour</td>
<td>150 µg/m³</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Hourly</td>
<td>235 µg/m³</td>
<td>200 µg/m³</td>
</tr>
<tr>
<td>CO</td>
<td>Daily</td>
<td>-</td>
<td>4 mg/m³</td>
</tr>
<tr>
<td></td>
<td>Hourly</td>
<td>40 mg/m³</td>
<td>10 mg/m³</td>
</tr>
<tr>
<td>TSP</td>
<td>Annual</td>
<td>-</td>
<td>200 µg/m³</td>
</tr>
<tr>
<td></td>
<td>Daily</td>
<td>-</td>
<td>300 µg/m³</td>
</tr>
<tr>
<td>PM₁₀</td>
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<td>-</td>
<td>100 µg/m³</td>
</tr>
<tr>
<td></td>
<td>Daily</td>
<td>150 µg/m³</td>
<td>150 µg/m³</td>
</tr>
<tr>
<td>PM₂.₅</td>
<td>Annual</td>
<td>15 µg/m³</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Daily</td>
<td>35 µg/m³</td>
<td>-</td>
</tr>
</tbody>
</table>

<sup>a</sup>Mass concentrations approximated by converting from ppm units, assuming P=1 atm and T=298 K.

<sup>b</sup>Grade II standard applies to urban residential, commerce-traffic-residential mixed, common industrial, and rural areas.
buildings (Bai et al., 2003). Currently, the Ministry of Health has defined indoor concentration limits for SO2, NO2, CO, CO2, NH3, O3, B(a)P, PM10, formaldehyde, benzene and total VOCs, as well as a minimum ventilation requirement of roughly 8.3 L/s per person, all of which apply to residential and office buildings (Wang et al., 2004; Environmental Resource Management, 2002). I found two recent studies of VOC concentrations in urban Chinese residences published in English language journals. Wang et al. (2007) conducted a field study in 12 urban residences in different mainland Chinese cities and found that the average indoor formaldehyde concentration in the summer ranged from 19.3 µg/m³ for the homes in Xi’an and 93 µg/m³ for the homes in Beijing. Weng et al. (2010) monitored carbonyl concentrations at three residential locations in Hangzhou, China, and found that the highest average formaldehyde concentration was in bedrooms in the summer at a concentration of 121 µg/m³. For context, both the Chinese government and the World Health Organization give a guideline limit for formaldehyde of 100 µg/m³ (english.mep.gov.cn/standards_reports; WHO, 2010). Also, Wang et al. (2007) reported that the average indoor to outdoor ratios of formaldehyde, acetaldehyde and acetone, respectively, were 3.7, 1.3 and 1.9 in the summer, and 5.2, 1.33 and 1.6 in the winter. Likewise, Weng et al. (2010) found that the average indoor to outdoor ratios of formaldehyde, acetaldehyde and acetone, respectively, were 4.5, 1.6 and 2.3 in the summer, and 2.9, 2.3 and 2.4 in the winter. These results suggest that residential exposures to elevated concentrations of VOCs may be a public health issue of concern within mainland China.

Residential coal combustion has historically been a significant source of indoor air pollution in Beijing, as well as in the rest of China, since space heating and cooking have been traditionally carried out using coal-fired stoves (Lang and Huang, 1993). In 1994, raw coal was the predominant energy source used for residential purposes in both rural and urban areas of China (Zha et al., 2010). By 2004, the primary energy source used in urban residences was liquid petroleum gas followed by electricity; however, in rural residences the primary energy source remains coal (Zha et al., 2010). In Beijing, since 1998, residential coal-fired boilers and household stoves have gradually been replaced with natural gas and electrically powered appliances (Wu et al., 2008). Arvesen et al. (2010) reported that roughly 30% of energy use in Beijing households comes from natural gas, followed by 28% from electricity and 19% from centrally provided heat. The remaining household energy consumption in Beijing comes from coal (14%) and liquid petroleum gas (9%).

Cooking is a significant source of indoor air pollution in China. As in many parts of the world, exposure during indoor cooking has been identified as an important cause of disease among women in rural areas of China, primarily as a result of pollutants generated by the indoor combustion of solid fuels that are not properly exhausted (Kleinerman et al., 2002; Mestl et al., 2006; Zhang and Smith, 2007; Liu et al., 2007). However, unlike other parts of the world, cooking has also been suggested as a cause of disease among women in urban areas of China, not because of pollutants generated by the indoor combustion of fuel (in this case usually natural gas or liquid petroleum gas), but because of the heating of cooking oils to high temperatures. Specifically, Chinese cooking generally involves placing 25-100 mL of oil into a wok and heating it to approximately 280 °C before adding the food. The heating of the oil can generate smoke, which often results in women complaining of eye irritation while cooking (Shields et. al.,
Several population based control studies in urban areas such as Shanghai, Shenyang and Taiwan have identified that women who cook more frequently and who report observing visible smoke while cooking have an increased risk of lung cancer (Wang et al., 1996; Zhong et al., 1999; Ko et al., 2000; Zhao et al., 2006). Laboratory studies have found that that fumes produced by heated unrefined rapeseed oil and soybean oil, which are the two most common cooking oils in China, produce condensates with mutagenic properties and contain 1,3-butadiene, acetaldehyde, n-pentane, acrolein, propanol, n-hexane, propionaldehyde and benzene (Shields et al., 1995; Qu et al., 1992). There have been no studies investigating the possible role played by particles produced as a result of the heated oils in eliciting health effects. Interestingly, two studies have reported that cooking emissions play a role in exacerbating the outdoor PM$_{2.5}$ concentrations in Beijing. Specifically, in two outdoor sampling campaigns of one-year duration, cooking emissions were found to make a significant contribution to the total organic composition of atmospheric PM$_{2.5}$ (Huang et al., 2006; Ho et al., 2010). As a result, pollutants generated by cooking in China are not only expected to elicit health effects among those who cook, but may also contribute to outdoor air pollution.

Only recently have studies begun to investigate ultrafine particle concentrations in China, and those studies have focused on the outdoor environment. To date, studies investigating different aspects of outdoor UFP concentrations in mainland China have been conducted in Guangzho (Yue et al., 2010a), Jinan (Gao et al., 2007), Shanghai (Li et al., 2007), the Yangtze River Delta near Shanghai (Gao et al., 2009), the Pearl River Delta (Yue et al., 2010a; Liu et al., 2008), rural Mount Waliguan (Kivekäs et al., 2009), and Beijing (Yu et al., 2005; Laakso et al., 2006; Wehner et al., 2008; Cheng et al., 2008; Wu et al., 2007; Wu et al., 2008; Yue et al., 2009; Wang et al., 2009a; Westerdahl et al., 2009; Yue et al., 2010b). The published studies investigating UFP concentrations in Beijing can be categorized into those focused on either general outdoor concentrations and sources (Yu et al., 2005; Laakso et al., 2006; Wu et al., 2008, Wehner et al., 2008), vehicular traffic emissions (Cheng et al., 2008; Westerdahl et al., 2009; Wang et al., 2009a), or atmospheric nucleation (Wu et al., 2007; Yue et al., 2009; Yue et al., 2010b). Of the two studies generally investigating Beijing outdoor concentrations and sources, one was conducted over a multi-year period (Laakso et al., 2006; Wu et al., 2008; Wehner et al., 2008), and one was conducted for approximately two weeks of winter and one week of summer (Yu et al., 2005). These studies have found that while the ultrafine particle concentration in Beijing is relatively high, it is in the same range as has been observed in some other urban areas, such as Atlanta, Toronto, Pittsburgh and parts of the Los Angeles air basin (see Table 1.1). However, the accumulation mode particle concentration (i.e. the concentration of particles with a diameter of 0.1 – 2 µm) is noticeably higher in Beijing compared to other urban areas, suggesting rapid growth of recently nucleated particles via gas condensation and potentially particle coagulation, as well as possibly the transport of aged particles from other parts of the region. Regional nucleation events are most frequent in the spring, which results in this season having the highest total particle number concentration. Wind speeds are lowest in the summer (mean of approximately 1.5 m/s compared to 2.2 m/s in the winter and spring), which results in the accumulation of pollutants within slow moving air masses. The minimum particle number concentration in Beijing is observed in the summer, presumably because the higher relative humidity and larger number of accumulated aged particles during this
season produces less favorable conditions for new particle formation. The diurnal variation of 3-20 nm particles appears to be mainly influenced by nucleation events, while the 20-100 nm particles appear to primarily correlate with traffic density, though they are also influenced by the growth of smaller particles.

In reviewing the English language literature, I found only one study that measured UFP concentrations in indoor microenvironments. Specifically, Dong and Yao (2010) measured PN concentrations for a limited duration (exact duration is unclear) in a university dormitory, office, hospital, subway car and train station. The highest average concentration \((33 \times 10^3 \text{ cm}^{-3})\) was observed at the train station, and the lowest concentration \((5 \times 10^3 \text{ cm}^{-3})\) was observed in the office. The concentrations in the subway car, hospital and dormitory were similar, ranging from 11 to 15 \(\times 10^3 \text{ cm}^{-3}\).

To date, there have been no studies published in English that have investigated characteristics of ultrafine particle concentrations in common urban residences in mainland China. As a result, Chapter 4 of this dissertation is focused on an investigation of the sources and concentrations of ultrafine particles in one common type of residential environment in Beijing, apartments in high-rise buildings.

### 1.4 Dissertation outline

The remaining chapters of this dissertation are organized as follows. Chapters 2 and 3 present results of a field study of UFP concentrations and exposures and the factors influencing them in San Francisco Bay Area schools. Chapter 4 presents analogous results from a field study of Beijing high-rise apartments.

Prior to the work reported in this dissertation, ultrafine particle concentrations had not been measured in classrooms in the San Francisco Bay Area, or even in classrooms in the United States. Consequently, there were no data available with which to assess the relative significance of UFP exposures that children anywhere in the US encounter in classrooms, or to identify the most important parameters affecting that exposure. Thus, the field study reported here, was conducted to fill this information gap. In Chapter 2, 18 cumulative days of time-series data collected in six San Francisco Bay Area classrooms are used to characterize students’ exposure to ultrafine particles in that setting. The results presented in that chapter contribute new information towards answering the following questions. Is student exposure to ultrafine particles in Bay Area classrooms significant relative to other microenvironments where children spend their time? How much do indoor UFP sources contribute, relative to outdoor sources, towards exposure levels in Bay Area classrooms? Can classroom exposures be predicted based on measured outdoor concentrations? These are questions that have been previously unanswered and the present dissertation makes significant contributions of information and insight.

Classroom UFP exposure concentrations consist of particles generated both indoors and outdoors. The air-exchange rate is an important parameter influencing the classroom exposure concentrations. It determines the rate at which outdoor generated particles penetrate indoors as well as the rate at which indoor particles are exhausted to the outdoors. In the case of the Bay Area classrooms investigated here, indoor particle concentrations were predominately of outdoor origin. In Chapter 3, the air-exchange rates of the six classrooms are quantified, and the relationship between the classroom air-exchange rate and the indoor proportion of outdoor particles is evaluated. The intent of
this chapter is to explore the role of the classroom air-exchange rate in maintaining a balance between adequately removing indoor generated pollutants, such as bioeffluents, while minimizing the indoor level of outdoor generated pollutants, such as ultrafine particles. This chapter also explores the feasibility of applying active particle control techniques in the classrooms.

The field study from which the results reported in both Chapter 2 and 3 are derived represents a small sample of classrooms within a limited geographical area. Therefore, the results cannot be considered representative of a broad range of schools. However, these results provide an indication of factors that may have an important influence on UFP exposures in school classrooms, and contribute towards an understanding of children’s exposure to ultrafine particles in the microenvironments where they spend much of their time.

Prior to the work reported in this dissertation, there had been no investigations reported in English of ultrafine particle concentrations in urban residences in Beijing, or in any other city in mainland China. Consequently, there were no data with which to characterize the time-integrated exposure of individuals living in major Chinese cities, or to assess the extent to which those exposures result mostly from indoor or outdoor sources. Considering the very large populations that live in Chinese cities, and the great challenges of improving air quality in Chinese cities, it is important that an understanding be achieved of the exposure conditions that exist within this part of the world. In Chapter 4, time-series data from 13 cumulative days of monitoring in 4 apartments in high-rise buildings in Beijing are used to characterize the residents’ exposure to ultrafine particles. I seek to answer the following questions. What is the daily-integrated exposure to ultrafine particles of the residents in the monitored apartments? What are the relative contributions of sources to the indoor exposure levels? To what extent does the building ventilation mode influence the resulting exposures? How do these results compare to those of homes in the San Francisco Bay Area? Again, while this study only represents a small sample of individuals in one type of Chinese urban living environment, it provides insight into the UFP exposure of a large population that has yet to be thoroughly studied.

The research reported in this dissertation lies at the intersection of air quality engineering and environmental health. The motivating purpose of the research is to generate knowledge that will contribute towards the health and well-being of susceptible or sizeable segments of the human population. This motivation is in line with the purpose of the environmental health discipline. However, to define whether an environmental health problem exists and, if so, to design solutions, engineering methods are needed. In this dissertation, I use tools from the discipline of air quality engineering to define the exposure conditions within two important indoor microenvironments, and, in doing so, provide information that will assist with developing strategies to reduce these exposures. To the extent that reducing UFP exposure concentrations is deemed important, the results provided should be augmented with further field studies and analysis of UFP exposures and the factors influencing those exposures, both in school classrooms and in urban living environments in China. As more information is collected and synthesized, informed decisions can be made regarding how population exposures can be reduced, if this is deemed a worthwhile goal.
Chapter 2: Characterizing UFP exposure concentrations in six elementary school classrooms in the San Francisco Bay Area

2.1 Introduction

Children represent a subpopulation generally considered to be at elevated risk of certain adverse effects from air pollutant exposure (Schwartz, 2004). Wiley et al. (1991) reported that children in California between the ages of 6 and 11 years old spend an average of 10% of their time at school, second only to the proportion of time they spend at home (53%). Yet, there have been very few studies investigating particle levels in elementary schools, and no studies investigating levels in elementary schools in California. Many of the studies investigating particle levels in elementary school classrooms have been focused on the accumulation mode (Wheeler et al., 2000; Janssen et al., 2001; Keeler et. al., 2002; Diapouli et al., 2007a; Parker et al., 2008; Stranger et al., 2008; Tippayawong et al., 2009; Guo et al., 2010). These studies found that outdoor sources were the primary contributor to indoor levels; therefore, indoor levels were generally lower than those outdoors, except during periods affected by resuspension. Sites located closer to local traffic and industrial sources appeared to have higher indoor concentrations than those located farther away.

The first studies investigating UFP concentrations in elementary schools were published in 2007. At present, there are eight published studies investigating this issue (Fromme et al., 2007; Diapouli et al., 2007b; Zöllner et. al., 2007; Guo et al., 2008 Weichenthal et. al., 2008; Wargocki et al., 2008; Morawska et al., 2009; Guo et al., 2010), none of which were conducted in the United States. The majority of these studies found that outdoor generated particles were the primary contributors to indoor levels. Two studies, both of which were conducted in Australia, observed significant indoor source events in the investigated classrooms (Morawaska et al., 2009; Guo et al., 2010). A detailed review of these studies is provided in §1.2. While these studies contribute to an understanding of children’s UFP exposure in classrooms, there is still much to be learned regarding how this exposure varies among seasons and across geographic regions, and how it is affected by occupant activities and building-related factors such as ventilation mode and rate.

The aim of the present work is to quantify and characterize UFP concentrations and associated student exposure in several classrooms in the San Francisco Bay Area. I report on the acquisition and analysis of simultaneously measured indoor and outdoor particle number concentrations, classroom occupancy, and classroom ventilation mode, all determined with 1-minute time resolution. Children’s average exposures to particle number (PN) concentrations while they are inside their classrooms are determined for each day of monitoring at the six classrooms, totaling 18 school days in all. The utility of average outdoor PN levels as an indicator of students’ exposure is investigated. The work presented in this chapter contributes to a better understanding of children’s exposure to ultrafine particles, emphasizing conditions in public, elementary school classrooms in an urban area of northern California.
2.2 Methods
2.2.1 Site description

Six northern California elementary school classrooms, located at varying distances from major roadways, and having a range of ventilation configurations, were monitored during June - December 2008. The schools were located in the urban East Bay region in Alameda County, about 20 km east of San Francisco. Table 2.1 summarizes the defining characteristics of the six classrooms, labeled S1-S6, including the dates for which the observational monitoring and manipulation experiments occurred. A floor plan of each classroom is presented in Appendix A. The six classrooms were at a total of four school sites, with S1 and S2 at one site, S3 and S6 at a second site, and S4 and S5 at two other schools. Three of the schools (S3-S6) were located within 2 km of each other, and the fourth school site was located about 10-15 km from the others. Only three of the classrooms (S2, S3 and S5) had a mechanical heating ventilation and air-conditioning (HVAC) system. At S3, the ventilation system operated 24 hours a day, and at S5, the system was on a timer to operate continuously from 6:45 to 18:15. At S2, the HVAC system was operated by the teacher, who appeared to only use it on occasions when the room temperature was perceived to be uncomfortably warm and could not be adequately cooled by opening windows and doors. In the end, this rarely occurred, and the HVAC system at S2 was used on only a few occasions. The other three classrooms were ventilated by a combination of air leakage (infiltration) and natural ventilation through operable windows and/or through open doors.

The Bay Area is characterized by microclimates, with temperatures sometimes differing by 10 °C between cities located less than 50 km apart. Alameda County is one of nine counties designated as part of the greater Bay Area (www.bayareacensus.ca.gov) and has a temperate climate. The coldest months are December through February, with average temperatures ranging from roughly 7 to 15 °C. The warmest months are July through October, with temperatures ranging from roughly 15 to 25 °C. Precipitation is greatest in November through March, with monthly averages ranging from 7 to 11 cm. In October and April, monthly average precipitation is roughly 3.5 cm, and in May through September the monthly average is less than 1.5 cm (www.wrh.noaa.gov/mtr). The Bay Area has a population of roughly 7 million. Alameda County has a population of roughly 1.5 million and a population density of roughly 760 km\(^2\) (http://factfinder.census.gov).
Table 2.1. Summary of classrooms studied.

<table>
<thead>
<tr>
<th>Site</th>
<th>Monitored (^a)</th>
<th>Classroom (^b)</th>
<th>Location</th>
<th>Ventilation</th>
</tr>
</thead>
<tbody>
<tr>
<td>S1</td>
<td>O: June 2-6</td>
<td>3(^{rd}) and 4(^{th}) grade students. Built in 1950’s. Vol. ~ 290 m(^3). Avg. class size~20 students.</td>
<td>Quiet residential street; closest freeway 500 m to E; elevation = 100-150 m.</td>
<td>One door leading to internal hallway; one to outdoor courtyard. Natural ventilation via door and windows to courtyard and door to hallway. No mechanical ventilation.</td>
</tr>
<tr>
<td></td>
<td>M: June 6-11</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>S2</td>
<td>O: Oct 6-15</td>
<td>1(^{st}) and 2(^{nd}) grade students. Built after year 2000. Vol. ~ 240 m(^3). Avg. class size~16 students.</td>
<td>Same as S1.</td>
<td>One door leading to outdoors. Mechanical ventilation with particle filter; sometimes used with fan on but no thermal control. Doors and windows often used for natural ventilation.</td>
</tr>
<tr>
<td></td>
<td>M: Oct 9-10</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>S3</td>
<td>O: Oct 20-24</td>
<td>2(^{nd}) grade students. Built in 1980’s. Vol. ~ 205 m(^3). Avg. class size~17 students.</td>
<td>Intersection of medium traffic urban street and low traffic residential street; closest freeway ~1.5 km to W. Elevation &lt; 50 m.</td>
<td>Two double doors leading to outdoors, and one internal door leading to small shared room. No operable windows. Mechanical ventilation with poorly fitting particle filter operated continuously.</td>
</tr>
<tr>
<td></td>
<td>M: Oct 27-29</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>S4</td>
<td>O: Nov 4-7</td>
<td>5(^{th}) grade students. Originally built in early 1900’s. Vol. ~ 230 m(^3). Avg. class size~20 students.</td>
<td>Residential side street; busy surface street 100 m to N and 200 m to S; closest freeway ~1 km to W. Elevation &lt; 50 m.</td>
<td>One door leading to internal hallway. Building equipped with radiant heating system. Room ventilation and cooling regulated by windows.</td>
</tr>
<tr>
<td></td>
<td>M: Nov 7-10</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>S5</td>
<td>O: Nov 18-21</td>
<td>4(^{th}) grade students. Built in 1970’s. Vol. ~ 260 m(^3). Avg. class size~22 students.</td>
<td>Intersection of moderate and heavily trafficked urban streets; closest freeway ~0.5 km to W. Elevation &lt; 50 m.</td>
<td>Two doors leading to outdoors. Mechanical ventilation with particle filter.</td>
</tr>
<tr>
<td></td>
<td>M: Nov 21-25</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>S6</td>
<td>O: Dec 2-8</td>
<td>2(^{nd}) grade students. Built in 1980’s. Vol. ~ 300 m(^3). Avg. class size~15 students.</td>
<td>Same as S3.</td>
<td>One door leading to outdoors. Two wall-mounted air treatment units. Door opened on occasion for natural ventilation, but windows were never opened.</td>
</tr>
<tr>
<td></td>
<td>M: Dec 4-6</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

\(^a\) “O” refers to the observational monitoring phase, and “M” refers to the manipulation experiment phase. All monitoring occurred during 2008.

\(^b\) Classroom volumes are approximate, and include space occupied by furniture and cabinetry. Average class size calculated from every minute that at least one student was present.
2.2.2 Instrumentation and data acquisition

During observational monitoring, pollutant concentrations were measured simultaneously inside of the classroom and outdoors. Particle number (PN) concentrations were measured using water-based condensation particle counters (CPC, TSI model 3781), with a minimum detection limit of 6 nm (50% response level). Since UFP typically comprise 90% or more of PN concentrations, the data provided by the CPC are deemed a suitable surrogate measure of UFP concentration.

In addition to monitoring PN, indoor and outdoor levels of NO (2B Tech, Model 400), O₃ (2B Tech, Model 202), CO₂ (TSI, Q-Trak Plus 8554; LI-COR, 820), and temperature and relative humidity (TSI, Q-Trak Plus 8554) were also measured. The instruments were organized into two packages, with each set mounted within a commercial polyethylene case, roughly 60 cm in each dimension. A mast was attached to the case, so that air would be sampled from a height of 1.5-1.6 m, corresponding to the approximate breathing zone of a standing person. The gas analyzers were configured so that they sampled through a Teflon-coated line, preceded by a particle filter. The Q-Trak samples air passively; therefore, its sensor probe was mounted at the top of the mast. For particle sampling, each case had a length of aluminum tubing outside of the box and carbon-impregnated tubing inside the box, with an auxiliary pump drawing air at 8-9 liters per minute through a line teed at the entrance to the CPC. Ultrafine particle losses within the CPC sampling line were minimized through use of this high transport flow. Diffusion losses to the walls of the sampling line, estimated based on the work of Ingham (1975), were 13%, 6%, 3% and 2% for 5, 10, 15 and 20 nm diameter particles, respectively. The site-collected data were augmented using outdoor meteorology and pollutant data acquired from the nearest central monitoring station, located < 15 km away from the schools.

The indoor instrument package was placed within the main volume of the classroom. The exact location was determined by balancing the requirement of minimal disruption to classroom activities with that of monitoring as close to the primary occupied space as possible. At S1-S5, the instrument package was placed at the edge of the room, with the sampling line located a horizontal distance of 0.5-1.5 m from the classroom wall. At S6, the instrument package was placed near the center of the room. In all cases, the package was located 1.0-2.5 m from the student seating area.

The outdoor instrument package was located on the school campus as near to the classroom as possible; however, other important locating criteria included safety from tampering and accessibility of power. At four of the sites (S1, S2, S4, S5) the outdoor enclosure was located within 50 m of the classroom. At S3 and S6, the only secure outdoor location was at the edge of the school campus, approximately 150 m away from both monitored classrooms.

Supplementing air contaminant species levels, temperature and state-change data loggers were placed in key locations in each classroom to detect when doors and windows were opened, when heating or air-conditioning was used, or when potential sources, such as laser printers or overhead projectors, were used. Also, a researcher was continuously present in the classroom while school was in session on each monitored day to maintain a continuous record (with 1-min resolution) of classroom occupancy, door and window position, and potential source activities.

Prior to work at each site, CO₂, NO and O₃ monitoring instruments were calibrated in the laboratory using span gases of one or two concentrations, in addition to a zero concentration check. The CPCs were calibrated prior to deployment in the field by a zero concentration and flow rate check. Once in the field, prior to the start of indoor and outdoor monitoring, the instrument packages were set-up to monitor side-by-side for 4+ hours, with the intent of ensuring
that the instrument readings were accurate relative to each other. The goal of the final data processing was to ensure that (a) the readings were accurate, and (b) the instruments measuring the same pollutant indoors and outdoors would agree relative to each other if monitoring from the same location. The final data processing of the CO$_2$, NO and PN readings was thus achieved in two steps. First, one of the instruments was selected as a transfer standard, and the readings from that instrument were adjusted based on the laboratory calibration. Second, the readings of the other instruments were adjusted relative to the transfer standard, based on the results of the side-by-side monitoring data. In the case of CO$_2$ and NO, measured data were adjusted by both a slope and an intercept. In the case of PN, measured data were adjusted only by a slope factor, since the CPCs consistently read zero concentration with high accuracy during the blank check. At five of the six sites, the CPC used as the transfer standard was placed in the indoor enclosure. The mean ± standard deviation of the slope used to adjust the data of the outdoor CPC was 1.07 ± 0.09. In the case of O$_3$, meaningful side-by-side data could not be obtained owing to low ambient levels; therefore, the responses of both instruments were adjusted based only on laboratory calibrations. A summary of the adjustment parameters used to process the data at each site is provided in Appendix B.

2.2.3 Experimental protocols

Observational monitoring at each site was conducted for durations of 70-95 h during 3- to 9-day periods. Observational monitoring at S1, S4 and S5 was completed in a single contiguous period. At S2, S3 and S6, there were one to three breaks of 4-30 h duration in each monitoring period, as a result of field complications. School was in session for four monitored days at S1, two monitored days at S6, and three monitored days at each of the remaining four sites.

In addition to observational monitoring, roughly 15 h was spent in each classroom while it was unoccupied to conduct supplementary (“manipulation”) experiments to measure the air-exchange rate of the classroom and the particle emission rate of potential sources identified during observational monitoring. The air-exchange rate experiments involved elevating the classroom CO$_2$, through dry ice sublimation, to levels above 2000 ppm, and then monitoring the subsequent concentration decay. This procedure was repeated for all common classroom ventilation configurations observed during monitoring, and — depending on the configuration — represented a combination of air exchange owing to natural ventilation, mechanical ventilation and infiltration. Air-exchange rate determinations were also made from portions of the observational monitoring record, when a sudden drop in classroom occupancy resulted in suitable conditions for interpreting the concentration decay of CO$_2$. In this case, air-exchange rate analyses accounted for CO$_2$ emissions by the few occupants still present, if any, in the classroom during the analyzed decay period.

The source-characterization experiments entailed having a researcher conduct the potential source activity, in a similar manner as was observed in the classroom. Particle levels were monitored continuously from before the activity through subsequent decay, during which the room was unoccupied. During source activation, mixing conditions were enhanced through use of three fans; however, fan operation was terminated during the decay period, so as to not interfere with the particle deposition rate. In some cases, the source experiments confirmed that a given activity did not increase the classroom UFP concentration by a discernible degree. For an activity to be identified as a detectable PN source, either from observational monitoring or in a manipulation experiment, it would need to emit particles at a rate clearly detectable above the time-varying baseline level of indoor particle concentrations. For the conditions in this study,
classroom volumes were in the range of 200-300 m$^3$ and baseline indoor PN levels during occupancy were $5 \times 10^3$ - $15 \times 10^3$ cm$^{-3}$, so that the typical total number of suspended particles was in the range 1-5 x $10^{12}$ particles. For a short-term PN emissions episode to be discernible against a varying baseline of this scale, the emitted quantity would need to be at least on the order of 10% of the baseline value, i.e. 0.1-0.5 x $10^{12}$ particles. For a typical air-exchange rate of 2 per hour, the baseline PN removal rate by means of ventilation would be in the range 2-10 x $10^{12}$ particles per hour. Therefore, a persistent indoor PN source would need to emit at a rate of at least ~ 0.2-1 x $10^{12}$ particles per hour to be discernible.

### 2.3. Results and Discussion

The average PN concentrations measured indoors and outdoors at each site are reported in Table 2.2 for the total observational monitoring period, as well as for when students were present and for times when the room was vacant. An important finding was that the outdoor average PN level was consistently higher than the indoor average in all classrooms for each of the conditions. Average indoor PN levels during periods of student occupancy ranged from $5.2 \times 10^3$ to $16.5 \times 10^3$ cm$^{-3}$ at S5 to S1, with an average across all six sites of $10.8 \times 10^3$ cm$^{-3}$. During periods of student occupancy, the ratio of average indoor PN to average outdoor PN varied from 0.48 at S2 to 0.77 at S3 (average = 0.59).

The average indoor PN concentration was higher in every classroom during hours that the classroom was occupied, compared to hours when it was vacant. The explanation for this result lies in three primary factors: (1) indoor sources, though few, tended to correlate with occupancy; (2) external doors and windows were opened frequently when the room was occupied, resulting in an increased air-exchange rate and a concomitant increase in the indoor proportion of outdoor particles; and (3) at most sites, the period of classroom occupancy coincided with periods for which outdoor levels were elevated. These three factors and their implications on student UFP exposure are explored further in §2.3.1 - 2.3.3.

**Table 2.2.** Time-averaged particle number concentrations (in units of $10^3$ particles cm$^{-3}$) and indoor-to-outdoor concentration ratio (I/O) measured during observational monitoring at each site.

<table>
<thead>
<tr>
<th></th>
<th>Total Obs. Period</th>
<th>Occupied by Students</th>
<th>Unoccupied</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>In</td>
<td>Out</td>
<td>I/O</td>
</tr>
<tr>
<td>S1</td>
<td>9.8</td>
<td>16.0</td>
<td>0.61</td>
</tr>
<tr>
<td>S2</td>
<td>5.1</td>
<td>12.9</td>
<td>0.40</td>
</tr>
<tr>
<td>S3</td>
<td>10.5</td>
<td>14.2</td>
<td>0.74</td>
</tr>
<tr>
<td>S4</td>
<td>7.3</td>
<td>21.3</td>
<td>0.34</td>
</tr>
<tr>
<td>S5</td>
<td>3.2</td>
<td>9.7</td>
<td>0.33</td>
</tr>
<tr>
<td>S6</td>
<td>5.6</td>
<td>13.4</td>
<td>0.42</td>
</tr>
<tr>
<td>Avg.</td>
<td>6.9</td>
<td>14.6</td>
<td>0.47</td>
</tr>
</tbody>
</table>

The outdoor WCPC at S4 was not operated overnight (approximately 0:00 to 6:00). The entries represent averages during the periods monitored. The results of the total observation period and for the unoccupied period are likely biased high compared to the true time averages because overnight levels tend to be lower than 24-hr averages.

Entry represents the duration of time for which indoor monitoring took place. Outdoor monitoring occurred for a subset of this period.

I/O was calculated by taking the ratio of the average indoor and outdoor PN concentrations.
2.3.1 Indoor source events

The continuous time-series data on indoor and outdoor PN levels, combined with direct observations of classroom configuration and indoor activities, were used to infer the relative contributions of indoor sources and outdoor air to the indoor PN levels. Indoor emission sources can largely be classified into two distinct categories: episodic and persistent. Episodic sources, such as the use of a candle, produce a telltale signature of a rapid rise in indoor PN level that is decoupled from the outdoor time-trace, followed by a typically slower decay of the indoor PN level representing the post-release effects of removal by means of deposition and ventilation. Persistent sources, such as from unvented natural-gas pilot lights, are more difficult to discern, but clues can be gleaned from investigating how the indoor-to-outdoor concentration ratio varies with building operating conditions and with outdoor PN levels. At these six classrooms, we only observed a few instances of episodic indoor emissions and did not detect any significant contribution from persistent indoor emissions.

During the cumulative observational monitoring phases for all six classrooms, there were four instances when a source resulted in a discernible increase in indoor PN levels. These were (a) the making of pancakes on an electric griddle during a student presentation at S1 on 2 June; (b) the lighting of a candle to celebrate a child’s birthday at S3 on 20 October; and (c, d) the use of a natural-gas heater at S3 and S5 on 22 October and 20 November, respectively. The indoor and outdoor PN concentration time-series during these four events are shown in Figure 2.1. Using a method described in §3.2.2, I estimate that the indoor proportion of outdoor particles during the source events at S1, S3 and S5 were 0.49, 0.69, and 0.40, respectively. To estimate the contribution of outdoor particles to indoor levels during the source event, I apply that ratio to the measured outdoor PN concentration for the period when the indoor source was active. One can then attribute the remaining increment of indoor particles above the estimated outdoor contribution to emissions from the indoor source. In this manner, the cooking event at S1, the candle event at S3 and the heater event at S5 were estimated to contribute 26%, 8% and 38%, respectively, to the cumulative PN exposure experienced by students in their classroom on the respective days. The heater event at S3 occurred before the students had entered the classroom; therefore, this event is assumed to not have made a significant contribution to student exposure.
Figure 2.1. Indoor and outdoor PN time-series during 4 source events. (top left) Making of pancakes on an electric griddle in S1. (top right) Lighting a candle (1-min duration) to celebrate a child’s birthday at S3. Use of vented natural gas heater at S3 (bottom left) and S5 (bottom right). The beginning and end of source activities are shown, where known. The heater at S5 was turned off at approximately 12:30, on the day shown in the bottom right frame.
The heater source events at S3 and S5 produced broader indoor PN peaks than were observed from the cooking and candle events at S1 and S3, respectively. The difference in the persistence of the indoor source emissions is likely due to the difference in the duration of the source activity; specifically, the heaters were on for 0.8 – 3.2 hours, whereas the cooking and candle events each lasted for less than 10 minutes. An interesting feature of the heater source events was that particle emissions appeared to vary and eventually decline, while the heater was still in operation. A similar phenomenon was observed in a home in Alameda County (Bhangar et al., 2011). I hypothesize that particle emissions from the heaters at S3 and S5 resulted from the volatilization of materials that had deposited over time on heater surfaces, which then nucleated into new particles. For this reason, particle emissions decreased once the majority of deposited materials had been volatilized, even though the heater was still in use. If this hypothesis is correct, then it is expected that particle emissions would decrease or even cease with subsequent uses, until enough time had passed to allow for the deposition of more semivolatile materials either from the gas phase or associated with particulate matter or dust. This expectation was explored by conducting replicate experiments of heater use within S3 and S5, under identical ventilation conditions as had characterized the classroom during the original event. At S3, the deliberate heater use manipulation experiment produced a peak that was diminished relative to the peak that was produced during the initial use (Figure 2.2). At S5, the heater use experiment produced no perceptible increase in classroom PN. Consequently, these experiments support the above stated hypothesis; however, more experiments would be needed before a conclusive statement could be made regarding the cause of particle emissions from the heaters in these classrooms.

In addition to emissions detected during observational monitoring, one further activity that was tested in a manipulation experiment at S1 resulted in a significant increase in indoor PN. The classroom floor was mopped with a diluted pine-scented floor cleaner that was routinely used at the school, during a period when indoor ozone levels were approximately 10 ppb. This activity resulted in an increase of particle levels from a baseline of ~ 12 × 10^3 cm^-3 to a peak of 120 × 10^3 cm^-3 (Figure 2.2). Although this floor cleaner was reportedly used regularly at two of the six sites, no peak associated with cleaning was observed in the classrooms during observational monitoring, presumably because the classroom floor was not mopped during this span of days. At the remaining four sites, the liquid cleaning detergents used at the school did not appear to contain terpenes. Other studies have investigated UFP formation owing to ozone-terpene chemistry indoors associated with cleaning activities (Long et al., 2000; Coleman et al., 2008).

In addition to identifying activities in the classroom that were sources of ultrafine particles, activities were also noted that were found not to be discernible sources. Several activities that were anticipated to result in UFP emissions during observational monitoring, but did not appear to do so, were tested during the manipulation experiments to confirm that they in fact did not elevate particle concentrations by a discernible degree. We investigated three laser printers, an industrial grade vacuum cleaner, and a linen-scented disinfectant spray. For the printer experiments, we rapidly printed 350, 120 and 30 pages, respectively, on laser printers in three different classrooms. The vacuum experiment entailed vacuuming a carpeted floor area in a classroom for 10 minutes. For the disinfectant spray experiment, we operated two wall-mounted air cleaners that reportedly generated ozone, while cleaning table surfaces with the spray. In each case, no increase in indoor PN levels was observed as a result of these activities. Overall, the infrequency of UFP source activities observed within the classrooms (only 4 events
in 18 monitored days) resulted in them only having small influence on time-averaged classroom PN concentrations. Therefore, it is inferred that outdoor air was the predominant cause of indoor ultrafine particles in these classrooms during observational monitoring.

Figure 2.2. Time-series data from manipulation experiments involving floor mopping at S1 (top) and heater use at S3 (bottom). The experiments were conducted on 5 June and 29 October, respectively. Durations of the source activities are noted.
2.3.2 Classroom air-exchange rates

The extent to which outdoor particles are able to penetrate from outdoors and persist in classrooms depends largely on the rate and mode of classroom ventilation. The averages of air-exchange rate measurements for each classroom are presented in Table 2.3 for two general ventilation states: the “open” state, such that the classroom had one or more open doors and/or was being mechanically ventilated via an HVAC system, and the “closed” state, such that doors were closed and the HVAC system (if present) was off. Cumulatively, across the six sites, air-exchange rates were measured 18 times for “closed” conditions (14 during manipulation experiments plus 4 during observational monitoring) and 23 times for “open” conditions (10 during manipulation experiments and 13 during observational monitoring). The results determined from the manipulation experiments (i.e. when CO\textsubscript{2} was deliberately released into the room) and from observational monitoring (i.e. when air-exchange rate was determined from CO\textsubscript{2} decay after occupants had exited) were similar. Specifically, the average ± standard deviation of the air-exchange rate when classrooms were in the open state was 3.4 ± 2.7 h\textsuperscript{-1} and 3.3 ± 2.0 h\textsuperscript{-1}, and when classrooms were in a closed state was 0.3 ± 0.2 h\textsuperscript{-1} and 0.5 ± 0.3 h\textsuperscript{-1}, as determined from observational monitoring and manipulation experimental data, respectively.

Table 2.3. Average air-exchange rate (AER) for different ventilation and room configuration conditions at the classroom sites.

<table>
<thead>
<tr>
<th>Site</th>
<th>Closed (doors closed and air off)</th>
<th>Open (door(s) open and/or air on)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>time (h)</td>
<td>AER (h\textsuperscript{-1})</td>
</tr>
<tr>
<td>S1</td>
<td>0.4</td>
<td>0.5</td>
</tr>
<tr>
<td>S2</td>
<td>4.9</td>
<td>0.4±0.1</td>
</tr>
<tr>
<td>S3</td>
<td>0</td>
<td>—</td>
</tr>
<tr>
<td>S4</td>
<td>3.6</td>
<td>0.3±0.2</td>
</tr>
<tr>
<td>S5</td>
<td>0</td>
<td>0.8±0.8</td>
</tr>
<tr>
<td>S6</td>
<td>10.6</td>
<td>0.6±0.2</td>
</tr>
<tr>
<td>Combined</td>
<td>19.5</td>
<td>0.5</td>
</tr>
</tbody>
</table>

\(^a\) For “time (h)” and “N”, the total duration and total number of AER determinations, respectively, is reported for the six sites considering only periods of student occupancy. For “AER (h\textsuperscript{-1})”, weighted averages are reported, using the “time (h)” entries as the weighting factors.

Overall, the mean air-exchange rate of each of the classrooms when in the open state ranged from 1.7 to 5.4 h\textsuperscript{-1}, and when in the “closed” state ranged from 0.3 to 0.8 h\textsuperscript{-1}. Generally, a higher air-exchange rate is deemed preferable for a school, in as much as it results in a higher rate of removal of bioeffluents and other pollutants emitted indoors, such as volatile organic compounds (Godwin and Batterman, 2007; Hodgson et al., 2004). However, since outdoor sources were found to make a larger contribution than did indoor sources to ultrafine particles in these classrooms, a higher air-exchange rate is also expected to result in higher indoor concentrations of ultrafine particles. This expectation was observed to be true in the case of these classrooms. Specifically, S3, which had the highest time-weighted average air-exchange rate, also had the highest ratio of indoor to outdoor PN concentrations during periods of student occupancy (Table 2.2). Likewise, S6, which had the lowest time-weighted average air-exchange rate, also had one of the lowest ratios of indoor to outdoor PN concentrations during periods of
student occupancy (Table 2.2). However, if we look at the average CO₂ concentration, which is
an indicator of accumulating bioeffluents, we see that S3 had the lowest and S6 had one of
the highest average indoor concentrations of CO₂ during periods of student occupancy (Table 2.4).
Thus, although a lower air-exchange rate is expected to result in a lower indoor proportion of
outdoor particles, it may also result in a higher concentration of indoor generated pollutants, and
is therefore not necessarily preferred. This topic is explored more extensively in Chapter 3.

Table 2.4. Indoor and outdoor time-average levels of carbon dioxide, ozone, nitric oxide and
temperature, in each classroom, during hours occupied by students.

<table>
<thead>
<tr>
<th></th>
<th>S1</th>
<th>S2</th>
<th>S3</th>
<th>S4</th>
<th>S5</th>
<th>S6</th>
<th>Average</th>
</tr>
</thead>
<tbody>
<tr>
<td>CO₂ (ppm)</td>
<td>In</td>
<td>Out</td>
<td>In</td>
<td>Out</td>
<td>In</td>
<td>Out</td>
<td>In</td>
</tr>
<tr>
<td></td>
<td>650</td>
<td>350</td>
<td>580</td>
<td>510</td>
<td>1160</td>
<td>400</td>
<td>1080</td>
</tr>
<tr>
<td>O₃ (ppb)</td>
<td>10</td>
<td>28</td>
<td>9</td>
<td>20</td>
<td>5</td>
<td>22</td>
<td>2</td>
</tr>
<tr>
<td>NO (ppb)</td>
<td>0.8</td>
<td>4</td>
<td>15</td>
<td>17</td>
<td>6</td>
<td>10</td>
<td>17</td>
</tr>
<tr>
<td>T (°C)</td>
<td>24</td>
<td>20</td>
<td>23</td>
<td>19</td>
<td>24</td>
<td>20</td>
<td>23</td>
</tr>
</tbody>
</table>

* Average calculated from a combination of on-site and central monitored data, because on-site outdoor temperature
data were only available on the first day.

Of the four classrooms ventilated primarily naturally (S1, S2, S4, S6), the three that were
monitored between June and early November, when outdoor temperatures were still relatively
warm, were in the open state for 60% to 98% of the time occupied by students. Conversely, S6,
which was monitored in early December, was in the open state for only 20% of the time
occupied by students. This finding indicates that classrooms that are ventilated primarily by
natural means may be characterized by a lower air-exchange rate during months when the
outdoor weather produces uncomfortable indoor thermal conditions.

2.3.3 Diurnal trends in indoor and outdoor PN

Figure 2.3 presents time-series plots of the indoor and outdoor PN data measured at each
site. From these plots, both the indoor PN source events and the outdoor PN diurnal trends can
be observed. In reviewing the outdoor profiles, three distinct types of peaks can be observed.
One is a midday peak, which occurred most distinctively at S1, S2 and S4, and was also
observed at S3. A second is a morning peak, beginning shortly after 6:00, which was seen on at
least one monitored day at all sites. The third is an evening peak, occurring after 18:00, and most
prominently observed at S4, S5, and S6. These three types of outdoor peaks occurring diurnally
at the different sites can be more clearly seen in the time-series plots shown in Figure 2.4, which
were calculated for each site by averaging the outdoor PN concentrations measured at each clock
minute, for all days monitored. Similar types of peaks have been reported by several other
studies investigating outdoor PN concentrations in urban centers across the United States (Jeong
et al., 2004; Stanier et al., 2004; Shi et al., 2001). The authors of these studies propose that the
midday peak is the result of atmospheric nucleation, typically occurring on warm days when
solar radiation intensity is high; the morning and evening peaks are the result of elevated
vehicular emissions during rush-hour traffic. While there is not enough information to identify
the exact cause of the outdoor peaks measured at the present study sites, it is observed that the
two sites for which there was no midday peak (S5 and S6) were also the two sites with the lowest average outdoor temperatures, as reported in Table 2.4, and also closest to the winter solstice with its lower level of solar radiation intensity. The potential for atmospheric nucleation events to occur in northern California has been previously suggested in a study conducted at a roadside location approximately 100 km northeast of S1-S6 (Nanzetta and Holmén, 2004).
Figure 2.3. Time-series plots of indoor and outdoor PN concentration measured for the duration of monitoring at the six school sites. Light vertical lines through the plots represent the midnight hour, and heavy vertical lines represent breaks in the monitoring period. Indoor and outdoor PN data for a given site are plotted on the same vertical scale, but scales differ between sites. Outdoor PN could not be measured overnight at S4.
Figure 2.4. The outdoor diurnal average PN trends at sites S1-S6, respectively. Trends represent 1-minute resolution averages of 70-95 h of observational data at each site. Note that the PN concentration scales differ among plots.
With respect to classroom exposure, the midday peak coincides with student occupancy, whereas the rush-hour peaks primarily occur when students are not at school. Consequently, we see in Table 2.2 that S5 and S6 were the only two sites for which the outdoor PN average during occupied hours was lower than when the classroom was vacant. The exposure consequences of elevated midday outdoor PN levels in the warmer months are exacerbated by the more frequent use of open doors and windows to regulate indoor temperatures, therefore increasing the indoor proportion of outdoor particles. The lower outdoor PN levels during occupancy and the less frequent door/window openings that characterized the occupied hours of the winter months were primarily responsible for the lower indoor PN exposure concentrations observed at S5 and S6 as compared with the other sites (see Table 2.2). The influence of the PN diurnal trends on students’ particle exposure is further investigated in §2.3.4.

The four identified indoor source events can be seen in Figure 2.3 for the three classrooms in which they occurred. Specifically, elevated indoor PN levels are apparent corresponding to the pancake cooking event at t~ 89 h in S1, the candle event at t~ 5 h in S3, and the heating events at t~ 30 h and t~ 49 h in S3 and S5, respectively. The sharp peak in both the indoor and outdoor PN profile at 7:00 on 14 October (t~ 47 h) in S2 was the result of an outdoor peak event, the source of which was not identified. In the absence of indoor source events, the indoor PN profile shows a damped version of the outdoor profile, with the degree of damping determined in large part by the classroom air-exchange rate. In particular, we see that at S3 where the air-exchange rate was highest, many of the short-term outdoor PN peaks translate to the indoor time-series profile. In contrast, at S2 and S4 where the air-exchange rate was relatively low, only the persistent features of the outdoor PN trend translate to the indoor profile, particularly in the late evening and early morning hours when the classroom was closed. It should be noted that the outdoor monitor at S3 and S6 was located approximately 10 m from a relatively busy arterial road, while the classrooms were located approximately 150 m from the same road; consequently, some of the indoor damping of the sharp peaks observed outdoors may reflect the greater influence of roadway emissions on the outdoor monitor than would be expected outdoors at the classroom location.

2.3.4 Student PN exposure

The average daily-integrated PN exposure of students inside the classroom was calculated as the ratio of the daily cumulative student exposure to the average student occupancy. The daily cumulative student exposure was determined by multiplying the indoor PN concentration for each minute by the number of students present and summing that product over all of that day’s minutes when at least one student was present. The average student occupancy was determined as the time-averaged number of students present in the classroom when at least one student was present on that day. The units for the average daily-integrated PN exposure (cm$^{-3}$ h d$^{-1}$) can be interpreted as the classroom PN concentration integrated over all hours per day that a student was present. Figure 2.4 shows the results of these analyses for the 18 days of observational monitoring at the 6 sites. The two lowest average daily exposures occurred at S5 and S6, consistent with the low outdoor PN concentrations measured during hours of student occupancy at these schools. The two highest average daily exposures occurred at S1 and S3. The highest exposure occurred on the last day of monitoring at S1 and was influenced by the pancake-cooking event that took place on that day. On the second highest average day (at S3), the average outdoor PN concentration during hours of student occupancy was the 7th highest of the 18 days of monitoring; however, the corresponding value of the average indoor to outdoor
PN concentration ratio was the highest of the 18 days. This combination of a relatively high outdoor PN average and a high indoor to outdoor PN ratio resulted in the observed high average student exposure. For all eighteen days, the average within-classroom daily-integrated exposure for students ranged from $11 \times 10^3$ cm$^{-3}$ h d$^{-1}$ to $100 \times 10^3$ cm$^{-3}$ h d$^{-1}$, with an overall mean of $50 \times 10^3$ cm$^{-3}$ h d$^{-1}$ and a relative standard deviation of 46%.

To put these exposures into context, note that the average daily-integrated PN exposure experienced at home by seven children subjects in a corresponding study of six San Francisco Bay Area single-family dwellings was ~ 6x higher, at a level of $320 \times 10^3$ cm$^{-3}$ h d$^{-1}$, with a relative standard deviation of 71% (Nazaroff et al., 2010). The higher daily exposure experienced by children in homes is partly attributable to the higher PN concentrations measured in homes during hours of occupancy than in schools, and partly a result of the greater time that children spend in their home on a daily basis as compared to their classrooms. To elaborate on the latter point, the average time spent at home by the seven children subjects was 17.3 h d$^{-1}$, as compared to an average of 4.5 h d$^{-1}$ spent in classrooms by the students in this study. Furthermore, children’s exposure to particles in classrooms would generally apply to only ~ 180 days of the year, while exposure in homes would occur about twice as frequently. Consequently, the results of these studies in the San Francisco Bay Area suggest that cumulative annual exposure to UFP by children in their homes would be about an order of magnitude greater than the cumulative exposure in their classrooms.
Figure 2.5. Average daily-integrated exposure of students to PN concentrations in their classrooms for each of the eighteen observational monitoring days. The left-hand axis indicates the classroom school site (S1-S6) and the monitored date. The numbers adjacent to the bars indicate the average daily-integrated exposure per student, in units of 1000 cm$^{-3}$ h d$^{-1}$ (and the average number of students in the classroom when occupied by students).

Figure 2.5 shows time-series results with one-minute resolution for cumulative student exposure, which effectively represents the product of the indoor PN concentration and number of students present. An increase in the student exposure can be observed on each day that a source event occurred, specifically, on 6 June (S1), 20 October (S3) and 20 Nov (S5). On three other
occasions, there were modest peaks in the indoor concentration that occurred independent of outdoors but could not be linked with a specific indoor source activity. The influence of these peaks on student exposure can be seen on 3 June at 13:00 (S1), 19 November at 11:00 (S5) and 2 December at 9:30 (S6). In the absence of indoor source events, student exposure at S1-S4 tended to peak towards the middle or end of the day, when the outdoor PN concentration was at a daily high and doors and windows were opened for cooling purposes. For example, on 4 June at S1 the cumulative student exposure increased steadily with an increase in the outdoor PN concentration. On 13 October at S2, the increase in cumulative exposure in the afternoon relative to the morning resulted from both an increase in the outdoor PN concentration and in the number of open windows. On 6 November at S4, the increase in cumulative exposure for the last ~45 minutes of the school day was associated with both an increase in the outdoor concentration and a sudden change in the ventilation configuration from one window being open to four windows and the classroom door being open. Exceptions to the mid-day/afternoon increase in cumulative student exposure at S1-S4 were observed on 22 October at S3 and 4 November at S4. In the former case, no mid-day peak occurred and the outdoor baseline concentration remained relatively constant throughout the day, and in the latter case, the mid-day peak was modest and comparable in size to the morning peak. Therefore, little difference was observed in cumulative student exposure between morning and afternoon. The mid-day/afternoon peak in exposure in the absence of indoor source events was not observed at S5 and S6 owing to differences in the outdoor PN concentration profile. Specifically, at S5, the cumulative student exposure on 18 and 20 November was highest in the morning consistent with the peaks in the outdoor concentration. The maximum outdoor concentration measured at S5 occurred on the evening of 20 November; however, the students had left by this time, so their exposure was not affected. On 19 November, 2 December and 8 December, there were no significant peaks in the outdoor PN profile and the ventilation configuration of the monitored classroom remained relatively constant. Likewise, the cumulative student exposure on these days (in the absence of indoor peaks) also remained relatively constant at a low level. Thus, the results depicted in Figure 2.5 reinforce the finding that for this sample of classrooms, student exposures were higher on days characterized by a mid-day peak in the outdoor PN profile and by more frequent opening of windows and doors. At these sites, days of this type generally occurred in the summer and fall, when outdoor temperatures were warmer. Time-series plots of indoor and outdoor temperature, student occupancy and door and window openings for each monitored school day are provided in Appendix B.
2.3.5 Outdoor PN as a predictor of classroom exposure

In determining appropriate metrics for health studies, an important issue to address is the extent to which PN exposure within a classroom can be predicted from PN levels measured outdoors. We explored this issue by regressing the calculated average exposure rate of students in each of the classrooms against the 24-h average PN concentration measured outdoors at each site. The resulting regression line has an $R^2$ value of 0.75, as shown in Figure 2.5(a). Since students are only present for a portion of the day, the outdoor average during occupancy should be a better indicator of exposure than the 24-h average outdoor level. Indeed, the data support this finding, as the average in-classroom exposure regressed against the average outdoor PN concentration during occupancy produced a regression line with an $R^2$ value of 0.88 (Figure 2.5(b).) Based on these data, albeit of limited scope, the on-site outdoor concentration of PN, averaged over periods of student occupancy, appears to be a useful parameter with which to predict PN exposures inside classrooms. Before using this idea, however, it should be tested across a broader range of geographic and climatic settings. The utility of outdoor data for predicting exposures indoors also depends crucially on the dominance of outdoor air as the source of indoor PN levels.
2.4. Conclusion

Particle number concentrations in six San Francisco Bay Area elementary-school classrooms were predominantly attributable to particles of outdoor origin. Indoor sources that perceptibly influenced indoor PN levels were only detected in three classrooms and cumulatively had limited effect on average exposures. The classroom PN averages were higher during periods when the classrooms were occupied compared to when they were vacant, primarily because the occupied hours were characterized by (a) increased ventilation rates, (b) the occurrence of indoor source events, and (c) coincidence with daily maxima in outdoor PN. For this northern California region, the classroom PN exposure concentrations appear to be lower in cooler months than they are in warmer months, likely because the cooler months are characterized by less frequent door and window openings and lower midday outdoor PN concentrations. Extrapolating from evidence acquired here, the annual average student PN exposure estimated for classrooms in this study would be approximately an order of magnitude less than the estimated annual exposure determined in a parallel study of the same geographic area for children while in their homes. The large difference results from two factors: (a) much larger influence of indoor sources in homes than in schools; and (b) much greater time spent by children in their homes than in their schools. For these classrooms, the on-site outdoor PN concentration averaged over hours of student occupancy correlated well with students’ in-classroom average PN exposure.

It is reasonable to expect that important conditions that affect indoor UFP concentrations in classrooms and the resulting exposures of school children vary across the population of schools. This study, which investigated six classrooms over a period that cumulatively spanned about six months in only one climatic region of the US, cannot be considered broadly representative of all school conditions. Consequently, to the extent that it is deemed important to understand children’s exposure to UFP in schools, additional work is warranted to better the scope of conditions investigated.
Chapter 3. Characterizing ventilation rates in six San Francisco Bay Area school classrooms and their effect on ultrafine particle concentrations

3.1 Introduction

Ensuring adequate ventilation has long been considered a strategy for maintaining classroom air quality. Almost a century ago, Baker (1918) published a study investigating the influence of natural versus mechanical ventilation on the occurrence of illness among approximately 5,500 students in 134 classrooms. She reported that children in classrooms with closed windows and ventilated by mechanical means had more frequent respiratory diseases severe enough to prevent them from attending school than did children who were in classrooms maintained at the same or lower temperature and ventilated wholly by open windows. Less than a decade later, a debate ensued over whether it was preferable to maintain a high rate of ventilation (30 cfm or 57 L/s per student) using mechanical means, or to allow for a lower ventilation rate via either a mechanical system or open windows (Palmer, 1926; Cole et al., 1930; Cole et al., 1931). By the middle of the 20th century, there was general consensus among the research community that a mechanical ventilation rate of 30 cfm per student was not only unnecessary, but also possibly detrimental to comfort, owing to the higher levels of noise and more frequent occurrence of drafts with which it was associated. Instead, a ventilation rate as low as 15 cfm (i.e. 38 L/s) per student supplied either via a mechanical system or open windows was deemed sufficient for preventing respiratory illness among students (Yaglou, 1956). In recent decades, studies have again begun to investigate whether the health of students and school staff is influenced by the mode and rate of classroom ventilation. Wålinder et al. (1997 and 1998) studied schools in Sweden and found that school staff in classrooms both with a low air-exchange rate and with dilution versus displacement mechanical ventilation systems had increased occurrence of reduced nasal patency and inflammatory biomarker response in nasal lavage. Smedje and Norbäck (2000) found that the reporting of asthmatic symptoms was less common among 143 Swedish pupils after their classrooms were equipped with new ventilation systems that resulted in a mean increase in the air-exchange rate of 4 h⁻¹. Thus, this evidence, although limited in scope, suggests that the classroom air-exchange rate plays an important role in the health and comfort of students and staff.

A higher air-exchange rate in a classroom serves to ensure a high removal rate of indoor emitted pollutants, which is often assumed to be a key determinant of good indoor air quality. However, since classrooms are usually ventilated by minimally filtered outdoor air, an elevated air-exchange rate may also be associated with an elevated indoor level of outdoor air pollutants. This issue is illustrated in Figure 3.1, which presents a schematic of particle sources and losses within an indoor volume modeled as a well-mixed box.
Within this idealized indoor space, the indoor particle concentration has contributions from both indoor and outdoor sources, and decays via airflow to the outdoors as well as by deposition onto indoor surfaces. This model assumes that the mechanical system does not include recirculating air filtration. Another important loss mechanism not shown in Figure 3.1 is coagulation. Being that it is a 2nd order loss mechanism, coagulation is most important at high concentrations, and its influence diminishes as concentrations decrease. Results from the field study discussed in Chapter 2 do not indicate that the influence of coagulation is important to consider in the context of a classroom. For the purposes of this model, coagulation is not included as a loss mechanism. Outdoor particles are expected to enter the space via three pathways: infiltration, natural ventilation and mechanical ventilation. Natural ventilation (i.e., through open doors and windows) provides no resistance to the entrance of outdoor particles. Conversely, particles entering the space via mechanical ventilation commonly must pass through a filter, and via infiltration must pass through small openings in the building envelope; therefore, a fraction of the particles are prevented from entering. Equation (3.1) provides a mathematical description of the particle material balance within the indoor space.

\[
\frac{dN_i}{dt} = N_o \left( P \lambda_i + (1-\eta) \lambda_m + \lambda_n \right) + \frac{E(t)}{10^6 \cdot V} - N_i \left( \lambda_i + \lambda_m + \lambda_n + k \right)
\]  

(3.1)

Here \(N_i\) and \(N_o\) represent the indoor and outdoor particle number concentration (cm\(^{-3}\)), respectively, \(P\) represents the particle penetration efficiency via infiltration pathways, \(\eta\) represents particle collection efficiency of mechanical filtration, \(E(t)\) represents the

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**Figure 3.1.** Schematic of particle sources and losses in a well-mixed classroom. See Equation 3.1 for definition of terms.
emission rate of particles by an indoor source (particles/h), $V$ represents the indoor volume (m$^3$), $k$ represents the deposition rate constant for particles onto indoor surfaces (h$^{-1}$), and $\lambda_i$, $\lambda_m$ and $\lambda_n$ represent air-exchange rate (h$^{-1}$) via infiltration, mechanical ventilation and natural ventilation, respectively. Using time-averaged values to represent the air-exchange rates, outdoor PN concentration, and indoor emissions, and assuming $P$, $k$ and $\eta$ are constant, the steady-state solution to equation (3.1) can be written:

$$N_i = \frac{N_o (P \lambda_i + \lambda_m (1-\eta) + \lambda_n) + \frac{E}{(10^6 \cdot V)}}{\lambda_i + \lambda_m + \lambda_n + k}$$

(3.2)

The two terms in the numerator represent the contribution of outdoor and indoor sources to the indoor particle concentration, respectively. When these terms are equivalent, indoor sources and outdoor sources make equal contributions to the indoor particle concentration. Equation (3.3) presents an expression for the balance point air-exchange rate, $\lambda_{bp}$ ($\lambda_{bp}=\lambda_i+\lambda_m+\lambda_n$), which represents the total air-exchange rate for the point at which the indoor and outdoor source contribution to the indoor particle concentration is equivalent. This expression is derived by setting the two terms in the numerator of the right-hand side of Equation (3.2) equal to each other, assuming that no particles enter via natural ventilation (i.e. doors and windows are closed), and assuming that 90% of particles (on a number basis) are able to pass through both the building frame and mechanical filter (i.e. $P=0.9$ and $\eta=0.1$).

$$\lambda_{bp} = \frac{E}{0.9 \cdot N_o \cdot V}$$

(3.3)

As discussed in Chapter 2, at one Bay Area classroom (S3), there was an indoor particle source event resulting from use of the classroom heater. The emission rate during this event that lasted for roughly one hour was not constant. However, the time-averaged emission rate was calculated as $1.8 \times 10^{13}$ particles/h during the event. Substituting this value into equation (3.3), along with the classroom volume (~205 m$^3$), and the average outdoor PN concentration during this event of $7.2 \times 10^3$ cm$^{-3}$ produces a result for the balance point air-exchange rate of 13.6 h$^{-1}$. Actual air-exchange rates in classrooms are generally much lower. The calculation illustrates the importance of limiting indoor pollutant emission sources to the maximum extent practical as a strategy for ensuring good indoor air quality.

In the case of the six Bay Area school classrooms discussed in Chapter 2, only a few indoor sources were identified (candle, central heaters, cooking pancakes on a portable stove for a student presentation and terpene containing cleaning products reacting with ozone), and these sources were active for a relatively small percentage of the total time that students were present in the classroom. Consequently, the most significant contribution to indoor particle levels in these classrooms was made by outdoor sources. If the indoor source emissions term is removed from equation (3.2), such that all the indoor particles originate from outdoors, then a lower air-exchange rate results in a consistently lower indoor concentration. In fact, according to the idealized situation described by the box-model, the optimal situation for indoor ultrafine particles in the absence of an indoor source would be to have the indoor environment completely isolated.
from the outdoors, such that the air-exchange rate is near zero and outdoor particles are prevented from entering the indoor space. However, having too low an air-exchange rate is not desirable because particles are not the only pollutant in classroom environments. A low air-exchange rate results in uncomfortable and unhealthy conditions as a result of the build-up of indoor emitted pollutants including metabolic gases and other bioeffluents that cannot be avoided in occupied spaces. This conclusion is indicated by the studies cited earlier in this chapter, as well as those reviewed in Chapter 1, which have found an association between the health of students and staff and the classroom air-exchange rate or other surrogate measures of ventilation.

As discussed in Chapter 1, the American Society of Heating Refrigerating and Air-Conditioning Engineers (ASHRAE) has recommended a minimum ventilation rate of 5 L/s per person in school classrooms, which is an order of magnitude below the minimum ventilation rate of 28 L/s per student suggested by Yaglou, in his paper published in 1956. ASHRAE also recommends a maximum steady-state indoor minus outdoor CO₂ (dCO₂) level of 700 ppm for occupied spaces, determined on the basis of maintaining an environment such that “outdoor air will dilute odors from human bioeffluents to levels that will satisfy a substantial majority (about 80%) of unadapted persons (visitors) to a space” (ASHRAE 62.1-2010). Carbon dioxide is often used as a surrogate measure of human bioeffluents, because of its abundance and ease of measurement. Equation (3.4) describes a mass-balance of CO₂ in an occupied classroom, assuming that the only sources of CO₂ are the classroom occupants and the outdoors, and therefore (for example) not occupants from other classrooms in the school.

\[
\frac{dY}{dt} = Y_{out}\left(\lambda_{i} + \lambda_{m} + \lambda_{n}\right) + \frac{E_{CO₂}}{V}\left(Y - Y_{out}\right) - Y\left(\lambda_{i} + \lambda_{m} + \lambda_{n}\right) \tag{3.4}
\]

In this equation, \(E_{CO₂}\) represents the emission rate of CO₂ by the occupants of the classroom (cm³/h), and \(Y\) and \(Y_{out}\) represent the indoor and outdoor levels of CO₂ (ppm), respectively. Equation (3.5) presents the solution for the total air exchange rate when the classroom CO₂ emissions and indoor level are at steady-state.

\[
\lambda = \frac{E_{CO₂}}{V\left(Y - Y_{out}\right)} \tag{3.5}
\]

Using Equation (3.5), we can calculate the air-exchange rate necessary to maintain a value of dCO₂ (i.e. \(Y-Y_{out}\)) less than 700 ppm. As an example, a classroom with a volume of 205 m³ that is occupied by 20 students, each having a CO₂ emission rate of 13,200 cm³/h (Bartlett et al., 2004), would require an air-exchange rate greater than 1.8 h⁻¹ to maintain a value of dCO₂ below 700 ppm. This air-exchange rate corresponds to a ventilation rate of roughly 5 L/s per person.

To summarize, tension exists between the need to exhaust indoor emitted pollutants, while also limiting the introduction of pollutants from outdoor air. Increasing the air-exchange rate helps the former, but exacerbates the latter. This tradeoff is illustrated by Figure 3.2, which shows results for the change in the steady-state indoor CO₂ and PN concentration with increasing air-exchange rate, in an indoor space represented in terms of a well-mixed box model. In this case, increasing the air-exchange rate from a value of 0.1 h⁻¹ to 1 h⁻¹ results in a decrease in the indoor CO₂ concentration.
by a factor of 8, but also increases the indoor PN concentration by a factor of 6. Thus, the situation described by Figure 3.2 emphasizes the point that determining an appropriate classroom air-exchange rate requires accounting for both indoor and outdoor pollutant sources, as well as including consideration of possible pollutant controls.

Figure 3.2. Indoor concentrations of CO₂ and PN versus air-exchange rate. (Box-model was run using the following assumptions: Room volume of 205 m³, ventilation via mechanical system and infiltration only, 20 students with CO₂ emission rate of 13,200 cm³/h each, particle deposition rate of 1 h⁻¹, particle penetration efficiency from outdoors of 0.9, mechanical filter PN collection efficiency of 0.1, and constant outdoor concentration of CO₂ and PN of 400 ppm and 18,100 cm⁻³, respectively.)

To date, there have been a few studies investigating the influence of classroom ventilation rate on indoor pollutant levels in the presence of an indoor source, including CO₂, VOCs, molds, bioaerosols and CO (Daisey et al., 2003; Godwin and Batterman, 2007; Grimsrud et al., 2006). These studies generally conclude that the investigated classrooms were inadequately ventilated, primarily based on the observation of elevated concentrations of indoor generated pollutants. There have been other studies that have investigated the influence of ventilation rate on the classroom concentration of outdoor emitted particles (Guo et al., 2008; Goyal and Khare, 2009; Tippayawong et al., 2009). These studies found that a higher air-exchange rate generally resulted in a higher indoor to outdoor particle concentration ratio (I/O). Apart from these studies, there have been a few others that have considered the influence of ventilation rate on the classroom concentration of pollutants emitted by both indoor and outdoor sources (Kinshella et al., 2001; Blondeau et al., 2005). Kinshella et al. (2001) found that the I/O of particles having a diameter of 0.1-1 µm was less than one (indicating no significant indoor sources) and increased with air-exchange, whereas the I/O of particles having a diameters of 1-3 µm and >3 µm was greater than one (indicating the presence of indoor sources) and was higher at lower air-exchange rates. Blondeau et al. (2005) found that classroom O₃ concentrations were higher at high ventilation rates, and thus suggested that the classroom air-exchange rate should be significantly decreased on days with high outdoor O₃ concentrations, but that it should be increased on days with low outdoor air pollution levels to reduce the indoor concentration of bioeffluents.
In this chapter, I investigate the effect of classroom air-exchange rate on the exposure concentrations of outdoor generated ultrafine particles and indoor generated CO₂. I have identified only two prior papers that report on similar investigations of ultrafine particles. Weichenthal et al. (2008) monitored indoor UFP and CO₂ concentrations as well as window openings in 37 occupied classrooms in one primary and one secondary school for a total of 60 school days (from roughly 8:30 to 15:30) between January and March 2007. They detected a positive relationship between window openings and UFP concentration, but did not find that the relationship was statistically significant, which they attribute to the small number of days that classroom windows were open. They also did not find air-exchange rate to be a statistically significant predictor of classroom UFP concentrations, which they suggest was the result of measurement error. Guo et al. (2008) carried out a 2-week measurement campaign in one primary school classroom at a rural site in Australia, with the intent of investigating the ratio of indoor to outdoor PN concentrations, and the impact of air-exchange rate on indoor PN. These authors report that the I/O of PN concentrations generally decreased with decreasing air-exchange rate, with the exception being occasions when the classroom was sealed following a significant peak in the outdoor PN concentration. In these latter cases, the outdoor PN concentration was observed to decay more rapidly than indoors, resulting in a temporarily elevated I/O. While the results from Guo et al. (2008) contribute towards an understanding of how classroom exposure to ultrafine particles varies with changes in ventilation, there is not a simultaneous consideration of how these same changes in air-exchange rate affected the levels of pollutants primarily generated indoors. In addition, the studies of both Guo et al. (2008) and Weichenthal et al. (2008) were restricted to one season in one locale. Consequently, there is a need for further study of the relationship between classroom air-exchange rate and indoor levels of both indoor and outdoor generated pollutants. Research on this topic can support the development rational strategies that balance the use of ventilation to remove indoor generated pollutants with the use of control measures to prevent elevated indoor levels of outdoor generated pollutants.

The aims of this chapter then are to characterize the ventilation conditions of six San Francisco Bay Area classrooms and to explore the influence of ventilation modes and rates on (i) the indoor level of outdoor-generated ultrafine particles and (ii) indoor-generated CO₂. This aim was met through analyses of PN, CO₂, student occupancy, and classroom ventilation mode time-series data collected for two to four days in each of six classrooms (the same sites and monitoring periods reported in Chapter 2). The following parameters were calculated for periods of student occupancy: indoor minus outdoor CO₂ (dCO₂) level, ventilation rate per person, air-exchange rate, and indoor proportion of outdoor particles (IPOP). The results are used to (1) assess whether these classrooms are adequately ventilated according to ASHRAE standard 62.1-2010 and relative to results from other studies, (2) to explore whether increases in the air-exchange rate result in an increase in the IPOP as predicted by an idealized box model, and (3) to investigate how ventilation, occupancy and meteorological parameters influence both the air-exchange rate and IPOP. Finally, the costs and potential benefits of adjusting the air-exchange rate and applying particle controls in the case of each classroom are discussed. This study is the first to explore the balance between exhausting human bioeffluents from a classroom environment while at the same time limiting the penetration of outdoor ultrafine particles.
It is also the first to investigate the ventilation characteristics of school classrooms in northern California. It should be noted, however, that this study included only a small number of classrooms, with a limited data set for each; consequently, these results should be considered illustrative rather than conclusive.

3.2 Methods

3.2.1 Field sites and data collection

Data on particle number (PN) and CO₂ concentration, as well as classroom occupancy, activities and ventilation mode were collected from six classrooms in four elementary schools in Alameda County, California, during June-December 2008. Details regarding classroom volume, size and ventilation characteristics are provided in Chapter 2 (Table 2.1). The classroom floor plans are presented in Appendix A. Three of the classrooms (S2, S3 and S5) had mechanical heating, ventilation and air conditioning (HVAC) systems, whereas the other three classrooms (S1, S4 and S6) were ventilated via open windows and doors. All six sites would have contributions to ventilation from infiltration. At S3 and S5, the HVAC systems were configured as rooftop package units, and were programmed to provide continuous ventilation during periods of student occupancy, though the teacher had control over the supply air temperature. The HVAC system at S2 was solely operated by the teacher and was rarely turned on during the period monitored. Therefore, S2 is considered a naturally ventilated classroom for the purpose of this study. The average number of students during periods of student occupancy in the classrooms ranged from 15 at S6 to 22 at S5. The schedule at S3 and S6 was structured such that half of the students would arrive at 8:30 and leave at 14:00, while the other half would arrive at 9:30 and leave at 15:00. Thus, for two hours of the school day only half of the students were present, which resulted in these two classrooms having the lowest average number of students present over the full school day, defined here as the daily period when at least one pupil was present in the classroom.

PN and CO₂ concentrations were monitored continuously both indoors and outdoors for four school days at S1, for three school days at S2-S5, and for two school days at S6. PN concentrations were measured using water-based condensation particle counters (CPC, TSI Model No. 3781). Indoor and outdoor CO₂ concentrations were measured using a LI-COR (Model 820) and TSI Q-Track (Model No. 8554), respectively. A researcher was present in the classroom during periods that school was in session to record data on student occupancy, classroom activities and classroom ventilation mode with one-minute time resolution. Researcher-recorded observation data were augmented with data collected by state-change and temperature data loggers (HOBO U9; HOBO U10) that were deployed throughout the room. State-change loggers recorded when windows and doors were opened and closed, and temperature loggers recorded when potential indoor sources (e.g. printers) were used and when heating or air-conditioning was turned on or off. A more detailed description of the field sites, instrumentation and data collection methods is provided in Chapter 2, §2.2.

3.2.2 Data analysis

The data collected in each classroom during periods of student occupancy were divided into discrete segments during which there were no detectable indoor UFP source events, the total number of occupants did not change by more than 50% and the
ventilation configuration did not change for more than 30 seconds at a time (e.g., if the ventilation configuration involved a closed door, the door would have been opened for no more than 30 seconds at a time while a person entered or exited). For each of these periods, both an air-exchange rate and the average IPOP based on particle number concentration were calculated. If more than 50% of the students present on that day were in the classroom during the analyzed period, then the ventilation rate per person was also calculated.

The derivation of the equation used to quantify the air-exchange rate began with the mass-balance differential equation (3.6). Here, $Y$ represents the indoor CO2 level (ppm), $Y_{out}$ represents the outdoor CO2 level (ppm), $E$ represents the emissions of CO2 by the occupants (cm$^3$/min), $V$ represents the room volume (m$^3$), and $\lambda$ represents the air-exchange rate (min$^{-1}$).

$$\frac{dY}{dt} = \frac{E(t)}{V} + \lambda \cdot Y_{out}(t) - \lambda \cdot Y(t)$$ (3.6)

The terms of equation (3.6) can be separated and evaluated over the time interval $t_i - t_f$, to produce the following expression:

$$\int_{t_i}^{t_f} dY = \int_{t_i}^{t_f} \frac{E(t)}{V} \cdot dt + \int_{t_i}^{t_f} \lambda \cdot Y_{out}(t) \cdot dt - \int_{t_i}^{t_f} \lambda \cdot Y(t) \cdot dt$$ (3.7)

Approximating $\lambda$ as a constant for $t_i$ to $t_f$, integrating both sides and rearranging the terms produces equation (3.8) from which the air-exchange rate can be approximated using measured indoor and outdoor CO2 time-series data, classroom volume, and estimated indoor emissions:

$$\lambda = \frac{\int_{t_i}^{t_f} \frac{E(t)}{V} dt - (Y(t_f) - Y(t_i))}{\int_{t_i}^{t_f} Y(t) dt - \int_{t_i}^{t_f} Y_{out}(t) dt}$$ (3.8)

At S3, outdoor CO2 measurements were unavailable for the majority of the monitoring period. Consequently, a constant value of 489 ppm, which was the average indoor baseline concentration in the absence of indoor sources, was used in place of $Y_{out}(t)$.

Emission rate, $E(t)$, was estimated for each minute according to equation (3.9),

$$E(t) = S(t) \cdot e_s(t) + A(t) \cdot e_a(t)$$ (3.9)

where $S(t)$ and $A(t)$ represent the number of students and adults present in the classroom, respectively, and $e_s(t)$ and $e_a(t)$ represent the emission rate of CO2 by students and teachers (cm$^3$ CO2/min per person), respectively. Both $S(t)$ and $A(t)$ were recorded with 1-minute resolution based on visual observation. Values for student and teacher CO2 emission rates were acquired from published literature. Bartlett et al. (2004) calculated that students in a classroom in British Columbia, Canada, had an average CO2 emission rate of 221 cm$^3$/min. Their work was based on a numerical model that was validated with measured time-series data on classroom CO2 concentration and human occupancy. Smith
(1988) estimated that teachers and students in a Rhode Island classroom had a mean CO₂ emission rate of 390 cm³/min and 228 cm³/min, respectively, based on measurement of heart rates and an approximation of heights. Penman and Rashid (1982) estimated a CO₂ emission rate of approximately 320 cm³/min for an adult office worker, based on an assumed energy cost per m² of body area for clerical work. Klein et al. (1999) calculated CO₂ emission rates for 597 adults and 312 children, based on their age, sex, height and weight, and reported results ranging from 192 to 628 cm³/min for adults and 91 to 303 cm³/min for children. According to ASHRAE Fundamentals, the CO₂ emission rate for an average size adult should range from roughly 300 cm³/min when carrying out light office work from a seated position, to 600 cm³/min when carrying out an activity with an energy expenditure comparable to house cleaning (Persily, 1997). For the purposes of the present analysis, an emission rate of 221 cm³/min was used for students, since this value was calculated specifically from data collected in a classroom (Bartlett et al., 2004), and an emission rate of 390 cm³/min was used for teachers, since this value was specifically calculated to represent teachers (Smith, 1988). Both these values fall within the ranges calculated by the other cited sources. (Note: unit conversions were made assuming P=1 atm and T=293 K, so that 221 cm³/min corresponds to 24 g/h of CO₂ and 390 cm³/min corresponds to 43 g/h of CO₂, respectively.)

A sensitivity analysis was conducted to test the influence of what was considered to be the three main sources of error in the air-exchange rate calculation. The first is related to the accuracy of the Q-Trak, which is reported by the manufacturer as ± (3% of reading + 50 ppm). The influence of this source of error was investigated by adjusting the outdoor CO₂ concentration time-series measurements by ± 50 ppm and observing the effect. The LI-COR, which was used to measure the indoor CO₂ concentration, has a reported accuracy of < ±2.5% of the reading. Due to the relatively high accuracy of the LI-COR, error associated with this instrument was considered insignificant. The second source of error investigated was in relation to the estimated occupant CO₂ production rate, which is expected to vary depending on the size, gender and activity level of the occupants. The influence of this source of error was investigated by adjusting the CO₂ emission rates of both teachers and students by ± 25%, which produces a range that spans the values reported for adults in ASHRAE Fundamentals and reported for both adults and children by Klein et al. (1999). The third source of error was related to the assumption that the classroom exchanged air only with the outdoors, when in reality two of the classrooms (S1 and S4) had a door that opened into an internal hallway. The influence of this source of error was investigated for S1 and S4 by increasing the outdoor CO₂ measurements, which in equation (3.8) essentially represent the CO₂ concentration of the air with which the classroom is being ventilated, by 100 ppm. This value was selected based on the assumption that a corridor with a total volume of approximately 2000 m³, with an average of 20 occupants for the majority of time that school is in session, and that is ventilated by approximately 1/2 outdoor air and 1/2 air from adjoining occupied classrooms, will have a CO₂ concentration roughly 100 to 200 ppm above the outdoor concentration. Then, assuming that the classroom is ventilated by an equivalent amount of air from outdoors and from the hallway when connected to both spaces by open doors and/or windows, the average CO₂ concentration of the combined air coming into the classroom will be approximately 50 to 100 ppm greater than the outdoor concentration. Classroom S3 had a door that connected to a small multi-purpose room that was shared
by two other classrooms; however, since the supply air provided by the HVAC system was high in this classroom, the infiltration of air from the neighboring classrooms is assumed to make up a small portion of the total air-exchange. The results of this sensitivity analysis are presented and discussed in §3.3.4.

The derivation of the equation used to quantify the IPOP began with the mass-balance shown in equation (3.10). Here, \( P \) represents the penetration efficiency of particles from outdoors, \( k \) represents the deposition rate of particles to indoor surfaces (min\(^{-1}\)), \( \lambda \) represents the total air exchange rate (min\(^{-1}\)) and \( N_i \) and \( N_o \) represent the indoor and outdoor particle concentrations (cm\(^{-3}\)), respectively.

\[
\frac{dN_i}{dt} = N_o(t) \cdot P \cdot \lambda - N_i(t) \cdot (k + \lambda) \tag{3.10}
\]

The IPOP is represented by the ratio of the indoor and outdoor particle concentration, which, using a steady-state approximation of equation (3.10), is equivalent to the ratio of \( P\lambda \) and \( k + \lambda \), as shown in equation (3.11), where the IPOP is referred to as \( f \).

\[
f = \frac{N_o}{N_i} = \frac{P\lambda}{k + \lambda} \tag{3.11}
\]

Recognizing that steady-state conditions don’t generally hold, I next substitute the expression for \( f \) into equation (3.10) and rearrange to obtain equation (3.12).

\[
\left( \frac{1}{k + \lambda} \right) \frac{dN_i}{dt} = N_o(t) \cdot f - N_i(t) \tag{3.12}
\]

Rearranging equation (3.12) and integrating both sides using the approximation that \( f \) and \( k + \lambda \) are time invariant produces equation (3.13), from which the IPOP can be evaluated.

\[
f = \frac{N_i(t_f) - N_i(t_i)}{k + \lambda} + \int_{t_i}^{t_f} N_o(t)dt \tag{3.13}
\]

The time integrals of \( N_i(t) \) and \( N_o(t) \) were evaluated using measured time-series data. The value of \( \lambda \) used to evaluate \( f \) was determined by the solution to equation (3.8). The value of \( k \) was assumed to be constant at 1 h\(^{-1}\), which was approximated based on evidence from published literature regarding deposition rates of ultrafine particles in indoor environments (Wallace et al., 2004; Tareq et al., 2006; Tareq et al., 2009).

A sensitivity analysis was conducted to test the influence of changes in deposition rate and air-exchange rate on the resulting IPOP. This was accomplished by independently altering parameters \( k \) and \( \lambda \) by \( \pm 50\% \) in equation (3.13). The IPOP values calculated at S3 and S6 were additionally tested for uncertainty related to the assumption that the PN concentrations measured by the outdoor monitor accurately reflected the
concentration immediately outside the classroom. As described in §2.2, the outdoor instrument enclosure at S3 and S6 was located near a relatively busy arterial road, approximately 150 m away from the classrooms. Previous studies have shown that PN concentrations near major roadways, where ambient particle levels are presumably dominated by traffic sources, decrease by a factor of 3-5 between a distance of ~20 m to ~150 m from the roadside (Shi et al., 1999; Zhu et al., 2002a; Zhu et al., 2002b; Gramotnev and Ristovski, 2004). The traffic volume on the arterial road along with which S3 and S6 are located was moderate during the period monitored, therefore, PN concentrations are not expected to have decreased with distance from the roadway as significantly as was observed in the aforementioned studies. However, some difference in the outdoor concentration near the roadway versus near the classrooms is likely. In reviewing the PN time-series for S3 (Figure 2.3), which appeared to have higher outdoor particle penetration factor than did S6, it can be observed that, in the absence of indoor source events, peaks in the indoor PN profile were generally ~50% lower in magnitude than were peaks in the outdoor PN profile. Consequently, possible error resulting from a decay in the outdoor PN concentration from the roadside to these classrooms was tested by decreasing the average outdoor PN concentration used in the IPOP calculation by 50%.

The assumption that the classrooms were single zones linked only to the outdoors should also theoretically contribute to error in the calculation of the IPOP, particularly in the case of S1 and S4 that had doors leading to an internal hallway. However, unlike the case for CO2, there is assumed to be no significant indoor source of UFP within the hallway. This assumption is based on data provided by a third CPC that was placed in the hallway immediately outside of the classrooms. At three different times, moderate PN peaks were observed in the hallway outside of S1 that occurred independently of changes in the outdoor profile, suggesting the possibility of indoor source events. To avoid mischaracterizing indoor generated particles as those originating from outdoors, these periods at S1 were excluded from the analysis. At S4, no evidence of significant PN source events was observed in the hallway. Excluding data from the three periods at S1, particles within the hallway at S1 and S4 are assumed to have originated from the outdoors. Consequently, no significant error is expected to result from the assumption that the only sources of particles within the classrooms during the analyzed periods were of outdoor origin.

3.3 Results and discussion
3.3.1 Classroom CO2 concentrations during student occupancy
Across all sites, during periods when at least one student was present, the mean value of the indoor minus outdoor CO2 (dCO2) level was 388 ppm with a standard deviation of 331 ppm; and the geometric mean (GM) was 268 ppm with a geometric standard deviation (GSD) of 2.8. Figure 3.3 presents the cumulative distribution of the 1-minute average dCO2 concentration for the periods occupied by students at each site. The guideline given by ASHRAE for dCO2 in an occupied space is 700 ppm. A line is drawn at this level in each of the plots, and a text box is provided reporting the GM and GSD, as well as the percent of occupied time during which dCO2 exceeded 700 ppm.
Figure 3.3. Cumulative probability distributions of 1-minute average indoor minus outdoor CO₂ (dCO₂) levels during hours of student occupancy at each site. The GM and GSD of each distribution are reported. Values of dCO₂ below 100 ppm are within the instrument error and should therefore be treated as having a higher relative degree of uncertainty. The total periods of student occupancy represented by each distribution are 18.9, 13.9, 9.5, 14.5, 11.3 and 13.2 h at S1-S6, respectively.
The highest GM dCO₂ level occurred at S4, which is a naturally ventilated classroom. This classroom was characterized by the highest average number of students per unit volume of all the sites (0.087 students/m³), which—all else being equal—would result in the highest level of CO₂ during periods of occupancy. For 56% of the occupied time at S4, the dCO₂ level was above the ASHRAE suggested limit of 700 ppm. The lowest GM dCO₂ level occurred at S3, at a value of 130 ppm (this GM was calculated after excluding the lower 27% of data that were considered unreliable). The highest 1-minute average dCO₂ level at S3 was only 336 ppm; therefore, the 700 ppm suggested standard was never exceeded. S3 had an HVAC system that operated continuously, and the low CO₂ concentrations are an indication that the supply rate of outdoor air was likely higher than the required minimum. Conversely, S5, which was the only other site with continuous mechanical ventilation during periods of occupancy, had the second highest GM dCO₂, at a value of 518 ppm. Normalized occupancy levels were similar at S3 (0.085 students/m³) and S5 (0.083 students/m³). Apparently, S5 was receiving a lower mechanical supply rate of outdoor air than was S3. This inference is reflected also in the calculated air-exchange rates, which are presented and discussed in §3.3.3. The remaining three sites were ventilated naturally. Both S1 and S2 had relatively low GM dCO₂ levels, at values of 237 and 261 ppm, respectively. Each of these sites had windows and doors on opposing walls, creating the potential for enhanced air-exchange due to cross-ventilation. Site S6 had a higher GM dCO₂ than did S1 and S2, at a value of 484 ppm, despite having the lowest average number of students per unit volume (0.05 students/m³) of all the sites. At S6, the doors and windows were all on the same wall, therefore there was no potential for cross-ventilation. In addition the door was only opened for 20% of the occupied period and windows were never opened, compared to 98% and 60% of time that doors and/or windows were open at S1 and S2, respectively. A likely reason for this different behavior in door openings was that S6 was monitored in December, and the average outdoor temperature during student occupancy was 15 °C. Conversely, S1 and S2 were monitored in June and October, and the average outdoor temperature during student occupancy at both these sites was 20 °C. This observation suggests that classrooms in the Bay Area ventilated exclusively by natural means might experience lower air-exchange rates during winter months, due to colder outdoor temperatures.

A comparison of the measured classroom dCO₂ level to the ASHRAE suggested limit of 700 ppm indicates that three of the classrooms have air-exchange rates that adequately remove occupant generated bioeffluents always or almost always, while three of the classrooms exceeded the limit a significant proportion of the time (33-56%). Both the low and high dCO₂ groups contained two classrooms that were naturally ventilated and one that was mechanically ventilated, indicating that the mode of ventilation does not automatically determine whether or not the ventilation rate of a classroom will be adequate. The evidence does, however, suggest that classrooms without mechanical ventilation may be more susceptible to violating the dCO₂ limit of 700 ppm during the winter months when outdoor temperatures are colder.
### 3.3.2 Ventilation per person

The time-weighted average rates of ventilation per person calculated for each site are 18, 13, 27, 7, 4 and 7 L/s for S1 through S6, respectively. The ranking of classrooms from low to high ventilation per person is nearly identical to the ranking from low to high dCO2, which is expected, since both indicators are functions of air-exchange rate and occupancy. Although S4, S5 and S6 all had GM dCO2 levels that were above the ASHRAE suggested limit of 700 ppm for more than 30% of the time occupied by students, only S5 had a rate of ventilation per person below the 5 L/s minimum suggested by ASHRAE. It should be noted, however, that these reported averages do not represent the full period of student occupancy for which data were collected, because air-exchange rate determinations were only made for periods during which there was no significant change in occupancy or ventilation mode.

To provide context for these ventilation results, it is helpful to compare them with results reported for schools studied elsewhere in the world. Santamouris et al. (2008) found, in a review of 21 international papers, that naturally ventilated classrooms had a median ventilation rate of close to 3 L/s per person, and, in a review of 22 papers, they found that mechanically ventilated classrooms had a median of close to 8.3 L/s per person. The small sample of Bay Area classrooms investigated in this study do not conform to the differentiation of ventilation rate based on classroom ventilation mode that is reported by Santamouris et al. (2008), since the two mechanically ventilated classrooms in this study lie on opposite sides of the observed range. However, a comparison of these results does suggest that S1, S2 and S3 have ventilation rates above what has been previously reported for other classrooms investigated by international research groups, while S4, S5 and S6 are close to the ventilation rates previously reported. Overall, a comparison of these results with both those of Santamouris et al. (2008) and with ASHRAE standard 62.1-2010 indicates that none of these six Bay Area classrooms have ventilation rates sufficiently low to require urgent attention; however, they do suggest that during the periods studied classrooms S4, S5 and S6 frequently had ventilation rates that were inadequate for removal of bioeffluents to a satisfactorily low level.

### 3.3.3 Air-exchange rate and indoor proportion of outdoor particles

While the ventilation rate per person provides an indicator of the extent to which bioeffluents will be removed from a classroom, it does not provide a clear indication of how efficiently indoor pollutants that are emitted independent of occupancy will be removed. The classroom air-exchange rate is a more relevant parameter for comparing the pollutant removal efficiency of different classrooms. The time-weighted average air-exchange rates calculated for periods of student occupancy at each of the six Bay Area classrooms are 3.7, 3.6, 10.8, 2.5, 1.5 and 1.1 h⁻¹ at S1 through S6, respectively. The ranking of classrooms from low to high air-exchange rate is not identical to the ranking from low to high dCO2, which is expected since the air-exchange rate is not a function of occupancy. The highest average air-exchange rate among the six classrooms was observed at S3, which was also the classroom with the lowest GM dCO2 concentration. The lowest average air-exchange rate was observed at S6, which, of the four naturally
ventilated classrooms, was the only one for which the door was opened for a minority of the total monitored period.

To provide context for these results, in a study conducted in Los Angeles, California, the mean air-exchange rate calculated for 13 portable classrooms and 7 traditional classrooms in the spring, fall and winter of 2000-2001, during periods that school was in session, was 0.7 h⁻¹ and 0.9 h⁻¹, respectively (Shendell et al., 2004a). In a study conducted in Michigan, the mean air-exchange rate calculated for 64 classrooms in the spring of 2003, during periods school was in session, was 6.3 h⁻¹ (Godwin and Batterman, 2007). The majority of classrooms in both of these studies were equipped with mechanical ventilation systems, but not all were programmed to operate continuously. The air-exchange rates calculated for S1-S6 were generally higher than those calculated for the schools in Los Angeles, but lower than the schools in Michigan. Shendell et al. (2004a) suggested that the low air-exchange rates calculated for the Los Angeles classrooms might have resulted from the mechanical ventilation system either being improperly designed or frequently turned off by the teacher. Interestingly, the highest air-exchange rate measured in the Los Angeles study was in a classroom with no mechanical ventilation system, in which case, the teacher often opened the two doors leading to the outdoors for ventilation. Godwin and Batterman (2007) suggested that the high air-exchange rates measured in the Michigan classrooms resulted from of a combination of a high mechanical ventilation rate and frequent door openings during hours that school was in session.

The average air-exchange rate for different combinations of door and window openings was calculated for each site (Table 3.1). At S3 and S5, since the mechanical ventilation systems operated continuously during periods of student occupancy, the categorization of air-exchange rate according to number of doors open has less significance than it does for those sites that were naturally ventilated. In Table 3.1, results are reported both from calculations made using the integral mass-balance method described in §3.2.2 and the tracer gas decay method described in §2.2.3. The focus of this chapter is on the results produced by the integral mass-balance method, but the tracer gas decay method results are provided for comparison. Ideally, the results calculated by these two methods should agree. In reality, while some agreed well, others did not. The best agreement occurs for the “all closed” configuration at S6, with a percent difference of only 15% between the two results. This was also a configuration for which there were a high number of repeat determinations made for both methods of air-exchange rate calculation. For the five other cases for which multiple determinations were made using both methods, the percent difference between the integral mass-balance and the tracer gas decay results ranged from 74% for the “door cracked (fan on)” configuration at S5 to 77% for the “all closed” configuration at S2. A regression of the air-exchange rates determined from CO₂ decay against those determined using an integral-mass balance is shown in Figure 3.4. A line fit to these points has a slope of 0.48, an intercept of 0.73 and an r² of 0.71. If the “all closed” configuration at S3 is excluded, the best-fit regression line has a slope of 0.70, an intercept of 0.25 and an r² of 0.71. The latter regression line is shown in Figure 3.4.
The difference in results produced by these two methods may be attributed to a few key factors. The first is natural variability in air-exchange rate for a given ventilation configuration, depending on the indoor-outdoor temperature difference, the wind speed and direction, the strength of solar radiation and the degree of turbulence indoors. All else being equal, these meteorological and occupancy factors would be expected to affect the variability within and between the results produced by these two methods of calculation equally. However, since the data used to conduct these two methods of analysis were generally collected at different times of day, there may be differences in how meteorological and occupancy factors affected the calculated air-exchange rate. Specifically, the data sets used for the integral mass-balance analysis were collected during the day when students and/or teachers were present. Conversely, the data sets used for the tracer gas decay analysis were generally collected after school when the classroom was unoccupied and the sun was setting or had set. Since meteorological and indoor air turbulence characteristics will differ between these two periods, there is potential for a systematic bias in the resulting values for air-exchange rate. Greater air-turbulence and a higher indoor-outdoor temperature difference, which are expected to occur during periods that students are present, would tend to result in a higher air-exchange rate. An additional factor possibly influencing the results is related to error in the model assumptions. The model assumes that CO₂ within the classroom comes from two possible sources: the occupants and the outdoors. In reality, S1 and S4 were linked to an internal hallway by a door that was often left open; therefore, some air-exchange is expected to have occurred with the hallway. During school hours the CO₂ concentration in the hallway was likely elevated above the outdoors, which would tend to bias the air-exchange rates calculated using the integral mass-balance method low. This latter source of error is considered as part of the sensitivity analysis discussed in §3.3.4.
Table 3.1. Summary of the average air-exchange rate (AER) and IPOP for observed ventilation configurations at each of the six school sites.

<table>
<thead>
<tr>
<th></th>
<th>Integral Mass-Balance Method</th>
<th>Tracer Gas Decay Method</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>AER (h⁻¹)</td>
<td>AER SD ⁵</td>
</tr>
<tr>
<td><strong>S1</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>All closed</td>
<td>1.5</td>
<td>-</td>
</tr>
<tr>
<td>Outside door open</td>
<td>0.7</td>
<td>-</td>
</tr>
<tr>
<td>Hallway door open</td>
<td>3.2</td>
<td>1.6</td>
</tr>
<tr>
<td>Both doors open</td>
<td>4.4</td>
<td>3.7</td>
</tr>
<tr>
<td><strong>S2</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>All Closed</td>
<td>0.9</td>
<td>0.1</td>
</tr>
<tr>
<td>Door open</td>
<td>2.9</td>
<td>0.34</td>
</tr>
<tr>
<td>Door +1 window open</td>
<td>5.9</td>
<td>0.78</td>
</tr>
<tr>
<td>Door + 2 windows open</td>
<td>6.0</td>
<td>0.9</td>
</tr>
<tr>
<td><strong>S3</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>All closed</td>
<td>12.3</td>
<td>1.2</td>
</tr>
<tr>
<td>Door open</td>
<td>9.1</td>
<td>0.3</td>
</tr>
<tr>
<td><strong>S4</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>All closed</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>1 window open</td>
<td>0.6</td>
<td>-</td>
</tr>
<tr>
<td>3 windows open</td>
<td>1.0</td>
<td>-</td>
</tr>
<tr>
<td>Door only open</td>
<td>1.7</td>
<td>0.3</td>
</tr>
<tr>
<td>Door + 2 to 4 windows open</td>
<td>5.7</td>
<td>3.5</td>
</tr>
<tr>
<td><strong>S5</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>All closed (fan off)</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>All closed (fan on)</td>
<td>1.8</td>
<td>1.4</td>
</tr>
<tr>
<td>Door Cracked (fan on)</td>
<td>1.1</td>
<td>0.3</td>
</tr>
<tr>
<td><strong>S6</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Door Closed</td>
<td>0.7</td>
<td>0.6</td>
</tr>
<tr>
<td>Door Open</td>
<td>4.4</td>
<td>-</td>
</tr>
</tbody>
</table>

⁴“N” represents the number of determinations for which the air-exchange rate was calculated using the given method.

⁵“SD” represents the standard deviation of the results.

At three of the four naturally ventilated sites (S2, S4 and S6) the air-exchange rates with doors and windows closed were close to results found previously for classrooms in Los Angeles (Shendell et al., 2004). When multiple doors and/or windows were opened, the air-exchange rate in the classrooms increased by an order of magnitude. The influence of window openings on air-exchange rate has been previously investigated in homes. Specifically, Howard-Reed et al. (2003) found that opening a window in two different multi-story homes increased the air-exchange rate by an amount roughly proportional to the width of the opening, and opening multiple windows increased the air-exchange rate by an order of magnitude relative to the closed state.
The air-exchange rates calculated for the mechanically ventilated classrooms (S3 and S5) were generally higher than the rates calculated for the “all closed” configurations at the naturally ventilated sites. However, when the naturally ventilated sites were in an open configuration, the resulting air-exchange rates generally exceeded that of S5. The air-exchange rate at S3 exceeded that of all other classrooms for all configurations, which had been indicated by the low dCO2 concentrations at this site.

The indoor proportion of outdoor particles (IPOP) was calculated for the same periods that the integral mass-balance air-exchange rate was calculated at each site. Overall, the highest time-weighted average IPOP was calculated at S3, which is expected considering its high air-exchange rate. All else being equal, a higher air-exchange rate will result in a higher IPOP, as illustrated by Figure 3.2. The highest IPOP was found, counterintuitively, for the “all closed” configuration at S1. An inspection of the PN concentration time-series from this period indicates that there was a decline in the outdoor PN concentration that did not immediately translate indoors (see Appendix B, Figure B.2). Consequently, the high IPOP in this case appears to have resulted from the decoupling that occurred between the indoor and outdoor spaces when the doors were closed, versus resulting from an increased penetration of outdoor particles, which is the most straightforward interpretation of a high IPOP. As stated earlier, this reported result for the “closed configuration” at S1 represents just one determination; therefore, the results should be interpreted with caution. At the remaining sites, the IPOP increased with an increasing number of open windows and doors, as expected.

Details regarding the ventilation mode, occupancy, date and time, indoor-outdoor temperature difference, outdoor wind speed, and indoor and outdoor PN concentration for the periods during which each of the IPOP and air-exchange rate determinations were made, are reported in Tables 3.2-3.7. The periods are ordered within each table according to increasing air-exchange rate. The column titled “∆t” represent the duration of the data interval from which the determination was made. Accompanying time-series plots for each school day are presented in Appendix B.

At S1, 17 air-exchange rate and IPOP values were calculated. The air-exchange rates ranged from 0.7 h⁻¹ for a period when only the door to the outside was open to 10.3 h⁻¹ for a period when both doors were open. The three highest air-exchange rates were calculated for periods when both doors were open. For the remaining 14 determinations, there was no clear pattern between the number of doors open and the resulting air-exchange rate. This lack of association between ventilation mode and air-exchange rate may partly be due to error related to the assumption that air within the hallway was essentially equivalent to air outdoors, when in reality the hallway air is expected to have an elevated CO₂ concentration. Generally, this source of error would bias the air-exchange rate low, with the degree of bias dependent on the extent to which the hallway CO₂ concentration is elevated above outdoors (see §3.3.4 for discussion on sensitivity of results to sources of error). That being said, there is a temporal trend in the air-exchange rates that may be related to meteorology. Specifically, on the last two days of monitoring, the air-exchange rate progressively decreased throughout the day as the indoor-outdoor temperature difference decreased. Conversely, on the first two days of monitoring, the air-exchange rate remained relatively constant for the majority of the day (see Appendix B, Figures B.1-B.4). In general, an increased temperature difference is expected to result in an increased air-exchange rate; however, there is not enough
information in this case to determine conclusively whether temperature played a role in
the observed behavior of the air-exchange rate.

Overall, IPOP values calculated for S1 behaved according to expectation. On a
given day, the IPOP was generally higher when both doors were open and lower when
only one door was open. However, as illustrated in Figure 3.4, there was not a clear
pattern between the IPOP and the air-exchange rate at S1. Overall, the IPOP ranged from
0.39 (hall door only open) to 0.86 (both doors open).

Table 3.2. AER and IPOP calculated for 17 periods when school was in session at S1.

<table>
<thead>
<tr>
<th>Date</th>
<th>Start Time</th>
<th>∆t (h)</th>
<th>Hall Door</th>
<th>Outside Door</th>
<th>No. Occ.</th>
<th>T\textsubscript{in}-T\textsubscript{out} (°C)</th>
<th>WS (m/s)</th>
<th>PN in (10\textsuperscript{3} cm\textsuperscript{3})</th>
<th>PN out (10\textsuperscript{3} cm\textsuperscript{3})</th>
<th>IPOP (-)</th>
<th>AER (h\textsuperscript{-1})</th>
</tr>
</thead>
<tbody>
<tr>
<td>6/6</td>
<td>14:14</td>
<td>0.68</td>
<td>0</td>
<td>1</td>
<td>25</td>
<td>3.0</td>
<td>3.6</td>
<td>17.3</td>
<td>35.5</td>
<td>0.39</td>
<td>0.7</td>
</tr>
<tr>
<td>6/6</td>
<td>12:30</td>
<td>1.40</td>
<td>1</td>
<td>1</td>
<td>2</td>
<td>2.7</td>
<td>2.8</td>
<td>22.7</td>
<td>40.9</td>
<td>0.58</td>
<td>1.1</td>
</tr>
<tr>
<td>6/4</td>
<td>13:57</td>
<td>0.38</td>
<td>0</td>
<td>0</td>
<td>4</td>
<td>3.1</td>
<td>4.4</td>
<td>15.7</td>
<td>15.9</td>
<td>0.84</td>
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<tr>
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<td>12:04</td>
<td>0.37</td>
<td>1</td>
<td>0</td>
<td>10</td>
<td>3.9</td>
<td>5.3</td>
<td>24.1</td>
<td>37.2</td>
<td>0.60</td>
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</tr>
<tr>
<td>6/6</td>
<td>11:20</td>
<td>0.98</td>
<td>1</td>
<td>1</td>
<td>25</td>
<td>4.0</td>
<td>3.4</td>
<td>23.1</td>
<td>42.4</td>
<td>0.53</td>
<td>2.0</td>
</tr>
<tr>
<td>6/4</td>
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<td>0.93</td>
<td>1</td>
<td>0</td>
<td>24</td>
<td>6.6</td>
<td>4.4</td>
<td>10.6</td>
<td>18.4</td>
<td>0.72</td>
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</tr>
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<td>6/4</td>
<td>12:47</td>
<td>0.98</td>
<td>1</td>
<td>0</td>
<td>4</td>
<td>3.9</td>
<td>5.3</td>
<td>24.5</td>
<td>33.3</td>
<td>0.67</td>
<td>2.2</td>
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<td>1</td>
<td>0</td>
<td>30</td>
<td>4.8</td>
<td>4.4</td>
<td>22.8</td>
<td>40.9</td>
<td>0.56</td>
<td>2.4</td>
</tr>
<tr>
<td>6/5</td>
<td>14:05</td>
<td>0.92</td>
<td>1</td>
<td>1</td>
<td>26</td>
<td>-0.2</td>
<td>4.4</td>
<td>22.8</td>
<td>39.7</td>
<td>0.58</td>
<td>2.7</td>
</tr>
<tr>
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<td>0.33</td>
<td>1</td>
<td>0</td>
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<td>5.6</td>
<td>4.4</td>
<td>22.5</td>
<td>37.3</td>
<td>0.63</td>
<td>2.7</td>
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<td>1</td>
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<td>14.6</td>
<td>16.4</td>
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<td>2.8</td>
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<td>1</td>
<td>0</td>
<td>24</td>
<td>5.0</td>
<td>2.2</td>
<td>11.8</td>
<td>36.9</td>
<td>0.43</td>
<td>3.5</td>
</tr>
<tr>
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<td>1</td>
<td>1</td>
<td>21</td>
<td>6.0</td>
<td>2.4</td>
<td>9.9</td>
<td>13.2</td>
<td>0.74</td>
<td>3.6</td>
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<td>0</td>
<td>26</td>
<td>5.0</td>
<td>1.8</td>
<td>10.9</td>
<td>16.1</td>
<td>0.68</td>
<td>5.7</td>
</tr>
<tr>
<td>6/3</td>
<td>13:13</td>
<td>0.77</td>
<td>1</td>
<td>1</td>
<td>26</td>
<td>4.4</td>
<td>3.6</td>
<td>14.0</td>
<td>20.1</td>
<td>0.61</td>
<td>7.5</td>
</tr>
<tr>
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<td>1</td>
<td>1</td>
<td>25</td>
<td>3.4</td>
<td>3.6</td>
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<td>13.3</td>
<td>0.76</td>
<td>10.1</td>
</tr>
<tr>
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<td>1</td>
<td>22</td>
<td>4.1</td>
<td>3.2</td>
<td>7.3</td>
<td>9.9</td>
<td>0.80</td>
<td>10.3</td>
</tr>
</tbody>
</table>

\(^a^\) “1” represents an open door and “0” represents a closed door.

At S2, the air-exchange rate ranged from 0.7 h\textsuperscript{-1} (door and windows all closed) to
6.5 h\textsuperscript{-1} (door and 2 windows open). According to expectation, the air-exchange rate at S2
increased as the classroom became progressively more “open”. The IPOP also generally
followed this trend, with values ranging from 0.07 (doors and windows all closed) to 0.78
(door and one window open). S2 had been the most recently built of all the classrooms,
and the low IPOP calculated when doors and windows were closed indicates that it was
likely the most airtight.
Table 3.3. AER and IPOP calculated for 9 periods when school was in session at S2.

<table>
<thead>
<tr>
<th>Date</th>
<th>Start Time</th>
<th>Δt (h)</th>
<th>Open Doors(^a)</th>
<th>Open Windows(^a)</th>
<th>No. Occ.</th>
<th>T(<em>{\text{in}})-T(</em>{\text{out}}) (°C)</th>
<th>WS (m/s)</th>
<th>PN in (10(^3)cm(^3))</th>
<th>PN out (10(^3)cm(^3))</th>
<th>IPOP (-)</th>
<th>AER (h(^{-1}))</th>
</tr>
</thead>
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<td>0</td>
<td>13</td>
<td>1.2</td>
<td>3.7</td>
<td>8.4</td>
<td>0.07</td>
<td>0.7</td>
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<td>1.2</td>
<td>7.7</td>
<td>13.3</td>
<td>0.08</td>
<td>0.8</td>
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</tr>
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<td>11.4</td>
<td>20.1</td>
<td>0.43</td>
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</tr>
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<td>6.2</td>
<td>13.2</td>
<td>0.27</td>
<td>1.9</td>
<td></td>
</tr>
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<td>1.1</td>
<td>2.2</td>
<td>6.9</td>
<td>25.0</td>
<td>0.34</td>
<td>2.9</td>
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<td>29.5</td>
<td>0.63</td>
<td>5.8</td>
</tr>
<tr>
<td>10/06</td>
<td>10:12</td>
<td>1.45</td>
<td>1</td>
<td>1.0</td>
<td>21</td>
<td>1.5</td>
<td>4.7</td>
<td>7.9</td>
<td>10.9</td>
<td>0.78</td>
<td>5.9</td>
</tr>
<tr>
<td>10/14</td>
<td>10:25</td>
<td>1.55</td>
<td>1</td>
<td>2.0</td>
<td>22</td>
<td>1.6</td>
<td>3.3</td>
<td>11.5</td>
<td>17.7</td>
<td>0.66</td>
<td>6.5</td>
</tr>
</tbody>
</table>

\(^a\)Reported value represents the average number of doors/windows that were open during the period.

Only four air-exchange rate and IPOP calculations were made at S3, because the consistently low dCO\(_2\) values made it difficult to make an accurate determination of air-exchange rate using occupant generated CO\(_2\). Since air-exchange rate determinations were only made for periods when the indoor CO\(_2\) concentration was at least 100 ppm above outdoors, the presented results may disproportionately represent periods when the air-exchange rate was lower than typical for this site. Somewhat counterintuitively, the two air-exchange rates calculated for periods with doors closed were higher than the two calculated for periods with one door open. However, considering the mechanical ventilation rate at this site was high and constant, it is likely that changes in the rate of natural ventilation had a relatively limited influence on the total air-exchange rate. For the three determinations made on 23 October, the IPOP increased with increasing air-exchange rate. The lowest air-exchange rate overall at S3 (8.9 h\(^{-1}\)) was calculated for 21 October, and corresponded with the highest calculated IPOP (0.95).

Table 3.4. AER and IPOP calculated for 4 periods when school was in session at S3.

<table>
<thead>
<tr>
<th>Date</th>
<th>Start Time</th>
<th>Δt (h)</th>
<th>Open Doors(^a)</th>
<th>No. Occ.</th>
<th>T(<em>{\text{in}})-T(</em>{\text{out}}) (°C)</th>
<th>WS (m/s)</th>
<th>PN in (10(^3)cm(^3))</th>
<th>PN out (10(^3)cm(^3))</th>
<th>IPOP (-)</th>
<th>AER (h(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td>10/21</td>
<td>9:42</td>
<td>0.60</td>
<td>1</td>
<td>21</td>
<td>7.3</td>
<td>9.0</td>
<td>10.9</td>
<td>0.95</td>
<td>8.9</td>
<td></td>
</tr>
<tr>
<td>10/23</td>
<td>12:49</td>
<td>0.55</td>
<td>1</td>
<td>23</td>
<td>NA</td>
<td>6.3</td>
<td>7.3</td>
<td>12.7</td>
<td>0.52</td>
<td>9.3</td>
</tr>
<tr>
<td>10/23</td>
<td>9:52</td>
<td>0.75</td>
<td>0</td>
<td>22</td>
<td>NA</td>
<td>11.0</td>
<td>6.7</td>
<td>8.9</td>
<td>0.56</td>
<td>11.6</td>
</tr>
<tr>
<td>10/23</td>
<td>11:01</td>
<td>0.60</td>
<td>0</td>
<td>24</td>
<td>NA</td>
<td>8.0</td>
<td>4.8</td>
<td>9.4</td>
<td>0.67</td>
<td>13.2</td>
</tr>
</tbody>
</table>

\(^a\)Reported value represents the average number of doors/windows that were open during the period.

As at S2, the ventilation modes at S4 included a combination of door and window openings. Also like S2, the air-exchange rate at S4 increased as the classroom became progressively more “open.” Having only the door open appears to have impacted the air-exchange rate to a greater extent than having only the windows open. The lowest air-exchange rate at S4 was 0.6 h\(^{-1}\) (one window open) and the highest was 8.7 h\(^{-1}\) (door and four windows open). The IPOP generally increased with increasing air-exchange rate.
Table 3.5. AER and IPOP calculated for 11 periods when school was in session at S4.

<table>
<thead>
<tr>
<th>Date (mm/d)</th>
<th>Start Time (h)</th>
<th>∆t (h)</th>
<th>Open Doors</th>
<th>Open Win.</th>
<th>No. Occ.</th>
<th>T_in-T_out (°C)</th>
<th>WS (m/s)</th>
<th>PN in (10^3 cm^-3)</th>
<th>PN out (10^3 cm^-3)</th>
<th>IPOP (-)</th>
<th>AER (h^-1)</th>
</tr>
</thead>
<tbody>
<tr>
<td>11/4</td>
<td>11:17</td>
<td>0.55</td>
<td>0.1</td>
<td>20</td>
<td>1.1</td>
<td>3.6</td>
<td>12.6</td>
<td>28.4</td>
<td>0.37</td>
<td>0.6</td>
<td></td>
</tr>
<tr>
<td>11/6</td>
<td>10:16</td>
<td>0.73</td>
<td>0.0</td>
<td>27</td>
<td>3.7</td>
<td>1.3</td>
<td>5.3</td>
<td>13.8</td>
<td>0.44</td>
<td>1.0</td>
<td></td>
</tr>
<tr>
<td>11/6</td>
<td>12:48</td>
<td>0.50</td>
<td>1.0</td>
<td>27</td>
<td>-2.2</td>
<td>1.8</td>
<td>16.8</td>
<td>36.7</td>
<td>0.45</td>
<td>1.1</td>
<td></td>
</tr>
<tr>
<td>11/5</td>
<td>10:01</td>
<td>0.95</td>
<td>0.0</td>
<td>24</td>
<td>6.3</td>
<td>0.4</td>
<td>14.6</td>
<td>23.8</td>
<td>0.37</td>
<td>1.6</td>
<td></td>
</tr>
<tr>
<td>11/6</td>
<td>8:39</td>
<td>0.40</td>
<td>1.0</td>
<td>28</td>
<td>8.1</td>
<td>1.8</td>
<td>6.0</td>
<td>24.9</td>
<td>0.35</td>
<td>1.6</td>
<td></td>
</tr>
<tr>
<td>11/5</td>
<td>8:55</td>
<td>0.75</td>
<td>0.0</td>
<td>21</td>
<td>8.1</td>
<td>0.9</td>
<td>13.3</td>
<td>17.9</td>
<td>0.92</td>
<td>1.7</td>
<td></td>
</tr>
<tr>
<td>11/5</td>
<td>12:49</td>
<td>0.95</td>
<td>0.0</td>
<td>28</td>
<td>5.3</td>
<td>2.2</td>
<td>20.0</td>
<td>38.6</td>
<td>0.59</td>
<td>2.0</td>
<td></td>
</tr>
<tr>
<td>11/6</td>
<td>9:15</td>
<td>0.40</td>
<td>1.0</td>
<td>0.0</td>
<td>28</td>
<td>7.6</td>
<td>1.3</td>
<td>7.5</td>
<td>12.5</td>
<td>0.64</td>
<td>2.2</td>
</tr>
<tr>
<td>11/4</td>
<td>8:51</td>
<td>0.80</td>
<td>3.6</td>
<td>22</td>
<td>6.5</td>
<td>0.9</td>
<td>12.0</td>
<td>18.3</td>
<td>0.74</td>
<td>4.1</td>
<td></td>
</tr>
<tr>
<td>11/5</td>
<td>11:13</td>
<td>0.40</td>
<td>2.0</td>
<td>20</td>
<td>3.5</td>
<td>0.9</td>
<td>24.2</td>
<td>44.2</td>
<td>0.74</td>
<td>5.1</td>
<td></td>
</tr>
<tr>
<td>11/6</td>
<td>14:03</td>
<td>0.50</td>
<td>4.0</td>
<td>25</td>
<td>-1.2</td>
<td>2.7</td>
<td>31.3</td>
<td>34.2</td>
<td>0.89</td>
<td>8.7</td>
<td></td>
</tr>
</tbody>
</table>

The air-exchange rate at S5 ranged from 0.8 h^-1 (door open and mechanical ventilation system on) to 4.7 h^-1 (door closed and mechanical ventilation system on). Like S3, having the door open at S5 did not result in a systematically increased air-exchange rate relative to the closed door configuration. This result is more surprising in the case of S5 compared to S3, since the mechanical ventilation rate at S5 was relatively low. The highest air-exchange rates at S5 were calculated for periods during which the central heater was cycling on and off (11/20 10:51). Particle emissions had resulted from heater use earlier that morning, but the generated peak had decayed and emissions are assumed to have discontinued before 10:51. The IPOP at S5 generally increased with increasing air-exchange rate, ranging in value from 0.15 to 0.48.

Table 3.6. AER and IPOP calculated for 8 periods when school was in session at S5.

<table>
<thead>
<tr>
<th>Date (mm/yy)</th>
<th>Start Time (h)</th>
<th>∆t (h)</th>
<th>Open Doors</th>
<th>No. Occ.</th>
<th>T_in-T_out (°C)</th>
<th>WS (m/s)</th>
<th>PN in (10^3 cm^-3)</th>
<th>PN out (10^3 cm^-3)</th>
<th>IPOP (-)</th>
<th>AER (h^-1)</th>
</tr>
</thead>
<tbody>
<tr>
<td>11/20</td>
<td>12:36</td>
<td>0.60</td>
<td>1.0</td>
<td>28</td>
<td>6.0</td>
<td>2.2</td>
<td>2.1</td>
<td>6.2</td>
<td>0.18</td>
<td>0.82</td>
</tr>
<tr>
<td>11/20</td>
<td>13:15</td>
<td>0.42</td>
<td>1.0</td>
<td>28</td>
<td>6.4</td>
<td>3.1</td>
<td>1.4</td>
<td>6.0</td>
<td>0.20</td>
<td>0.84</td>
</tr>
<tr>
<td>11/20</td>
<td>15:00</td>
<td>0.50</td>
<td>0.0</td>
<td>1</td>
<td>5.3</td>
<td>2.7</td>
<td>2.1</td>
<td>11.3</td>
<td>0.25</td>
<td>1.09</td>
</tr>
<tr>
<td>11/18</td>
<td>12:42</td>
<td>0.97</td>
<td>0.1</td>
<td>28</td>
<td>6.1</td>
<td>3.1</td>
<td>2.9</td>
<td>7.6</td>
<td>0.31</td>
<td>1.25</td>
</tr>
<tr>
<td>11/19</td>
<td>12:36</td>
<td>1.05</td>
<td>1.0</td>
<td>32</td>
<td>6.1</td>
<td>3.1</td>
<td>1.1</td>
<td>3.2</td>
<td>0.28</td>
<td>1.33</td>
</tr>
<tr>
<td>11/18</td>
<td>11:03</td>
<td>0.73</td>
<td>0.0</td>
<td>29</td>
<td>6.0</td>
<td>2.2</td>
<td>3.7</td>
<td>8.1</td>
<td>0.40</td>
<td>1.41</td>
</tr>
<tr>
<td>11/20</td>
<td>11:27</td>
<td>0.32</td>
<td>0.2</td>
<td>30</td>
<td>5.9</td>
<td>2.2</td>
<td>2.6</td>
<td>6.8</td>
<td>0.24</td>
<td>1.75</td>
</tr>
<tr>
<td>11/20</td>
<td>10:51</td>
<td>0.40</td>
<td>0.1</td>
<td>30</td>
<td>5.9</td>
<td>1.8</td>
<td>5.4</td>
<td>9.3</td>
<td>0.48</td>
<td>4.72</td>
</tr>
</tbody>
</table>

*Reported value represents the average number of doors/windows that were open during the period
Although there were multiple windows at S6, the windows were never opened and the door was only left open on a few occasions. Of the 12 determinations made at this site, only one was made when the door was open, which produced the highest air-exchange rate and IPOP, at values of 4.3 h⁻¹ and 0.73, respectively. The air-exchange rate and IPOP values calculated for the 11 closed periods ranged from 0.3 h⁻¹ to 2.5 h⁻¹ and 0.19 to 0.81, respectively. During monitoring at this site, conducted in December, there were periods when the outdoor temperature was cold, resulting in a higher indoor-outdoor temperature difference than seen at other sites (i.e. greater than 10 °C). It was during these periods that the highest closed-door air-exchange rates were observed. The IPOP in this classroom was not found to increase with increasing air-exchange rate.

Table 3.7. AER and IPOP for 12 periods during observational monitoring at S6.

<table>
<thead>
<tr>
<th>Date (mm/dd)</th>
<th>Start Time</th>
<th>Δt (h)</th>
<th>Open Doors a</th>
<th>No. Occupants</th>
<th>T_in-T_out (°C)</th>
<th>WS (m/s)</th>
<th>PN in (10³ cm⁻³)</th>
<th>PN out (10³ cm⁻³)</th>
<th>IPOP (-)</th>
<th>AER (h⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>12/02</td>
<td>8:32</td>
<td>0.90</td>
<td>0</td>
<td>11</td>
<td>7.3</td>
<td>1.8</td>
<td>15.0</td>
<td>15.5</td>
<td>0.81</td>
<td>0.3</td>
</tr>
<tr>
<td>12/02</td>
<td>12:50</td>
<td>1.05</td>
<td>0</td>
<td>20</td>
<td>6.9</td>
<td>3.6</td>
<td>9.4</td>
<td>21.0</td>
<td>0.40</td>
<td>0.3</td>
</tr>
<tr>
<td>12/02</td>
<td>11:26</td>
<td>0.50</td>
<td>0</td>
<td>22</td>
<td>6.1</td>
<td>1.8</td>
<td>6.7</td>
<td>13.8</td>
<td>0.36</td>
<td>0.4</td>
</tr>
<tr>
<td>12/02</td>
<td>14:08</td>
<td>0.90</td>
<td>0</td>
<td>13</td>
<td>7.7</td>
<td>3.6</td>
<td>9.0</td>
<td>13.2</td>
<td>0.45</td>
<td>0.4</td>
</tr>
<tr>
<td>12/02</td>
<td>10:01</td>
<td>0.72</td>
<td>0</td>
<td>27</td>
<td>7.1</td>
<td>1.3</td>
<td>12.0</td>
<td>18.4</td>
<td>0.34</td>
<td>0.5</td>
</tr>
<tr>
<td>12/05</td>
<td>12:45</td>
<td>1.10</td>
<td>0</td>
<td>21</td>
<td>6.7</td>
<td>1.8</td>
<td>3.9</td>
<td>7.6</td>
<td>0.32</td>
<td>0.5</td>
</tr>
<tr>
<td>12/08</td>
<td>14:08</td>
<td>0.97</td>
<td>0</td>
<td>12</td>
<td>12.3</td>
<td>2.2</td>
<td>2.1</td>
<td>8.0</td>
<td>0.26</td>
<td>0.7</td>
</tr>
<tr>
<td>12/08</td>
<td>12:50</td>
<td>1.00</td>
<td>0</td>
<td>20</td>
<td>12.7</td>
<td>3.1</td>
<td>2.0</td>
<td>8.2</td>
<td>0.22</td>
<td>0.9</td>
</tr>
<tr>
<td>12/08</td>
<td>9:50</td>
<td>0.90</td>
<td>0</td>
<td>20</td>
<td>12.7</td>
<td>4.0</td>
<td>1.9</td>
<td>8.1</td>
<td>0.22</td>
<td>1.4</td>
</tr>
<tr>
<td>12/08</td>
<td>10:56</td>
<td>1.00</td>
<td>0</td>
<td>21</td>
<td>12.9</td>
<td>3.6</td>
<td>1.8</td>
<td>8.8</td>
<td>0.21</td>
<td>1.9</td>
</tr>
<tr>
<td>12/08</td>
<td>8:50</td>
<td>0.60</td>
<td>0</td>
<td>10</td>
<td>11.9</td>
<td>3.1</td>
<td>2.5</td>
<td>11.8</td>
<td>0.19</td>
<td>2.5</td>
</tr>
<tr>
<td>12/05</td>
<td>14:09</td>
<td>0.90</td>
<td>1</td>
<td>13</td>
<td>5.8</td>
<td>3.6</td>
<td>4.6</td>
<td>6.5</td>
<td>0.73</td>
<td>4.3</td>
</tr>
</tbody>
</table>

a Reported value represents the average number of doors/windows that were open during the period

To systematically investigate the association of IPOP with air-exchange rate, the analyzed periods at each site were broken into two groups based on whether the air-exchange rate for that period was below or above the site median. The mean IPOP for each group was then calculated, and a t-test was conducted to determine whether the mean IPOP of the high air-exchange rate group was greater by a statistically significant degree than that of the low air-exchange rate group. Linear regressions of the IPOP versus air-exchange rate are plotted for each site in Figure 3.4, with vertical and horizontal lines drawn through the medians. The means used to conduct the test for each site, as well as the resulting P-values, are shown in Table 3.8. Since there were only four determinations made at S3, the statistical test was not performed for this site; however, a regression plot is still shown. For the remaining five sites, four produced the expected result: high air-exchange rates yielded a mean IPOP that was significantly greater than the mean IPOP for lower air-exchange rates. In the case of S2 and S4, the difference achieved a P-value of <0.01. Since the regression at these two sites conformed most
closely to a line, an $R^2$, slope and intercept of the linear regression were calculated and are shown in Figure 3.4. In the case of S1 and S5, the P-value was <0.1. At S6, not only was the expected result not achieved, but the reverse occurred. Specifically, the mean IPOP of the high air-exchange rate group was lower than that of the low air-exchange rate group, with a statistical significance of $P=0.11$. The exact cause of this anomalous outcome is unclear; however, there does appear to be a temporal or meteorological trend within the results. Five of the six periods that had an air-exchange rate below the median but an IPOP above the median occurred on 2 December, and four of the five periods that had an air-exchange rate above the median but an IPOP below the median occurred on 8 December. This temporal pattern indicates that the systematic change in the relationship between IPOP and air-exchange rate from 2 December to 8 December was likely the result of a temporal change in the influencing parameters. Although the classroom doors and windows were generally closed on both days, there were differences in the outdoor meteorology. Specifically, for the five periods investigated on 2 December the indoor temperature was roughly 7 °C above outdoors, while on 8 December the indoor temperature was roughly 12 °C above outdoors. In general, an increased indoor to outdoor temperature difference is expected to increase air infiltration across a building frame because of the greater difference in air-pressure (ASHRAE, 2009); therefore, the higher air-exchange rates observed on 8 December may have been related to the change in outdoor temperature. However, this reasoning cannot be extended to explain why the periods on 8 December were characterized by a lower IPOP. The systematic difference in the IPOP between the two days may be related to a change in the wind direction. Specifically, on 2 December the wind blew predominately from the northeast, whereas on 8 December the wind blew predominately from the northwest. Since the outdoor PN monitor was located approximately 150 m northeast of the classroom, along a busy arterial road that runs from WSW to ENE, it is possible that this change in wind direction resulted in a change in the spatial gradient of the outdoor PN concentration with distance from the roadway. In the end, although there are a few possible explanations for the unusual relationship between IPOP and air-exchange rate at S6, there is no way to definitively identify the cause.

Overall, the results from these six classrooms support the expectation that higher air-exchange rates correspond to higher IPOP values for ultrafine particles. In the case of S2 and S4, this relationship is clearly manifested within a linear regression of these two parameters. In the case of S1 and S5, the relationship is less clear. This lower degree of clarity in the results from S1 and S5 is likely attributable to two general sources of error. The first is temporal variability in the particle deposition rate and in the relative contribution of the three pathways for air-exchange, all of which can influence the degree to which particles can penetrate and persist in the classroom. The second is error introduced into the calculation via instrument accuracy and model assumptions, which may influence the results of some classrooms more than others. The sensitivity of results to these latter mentioned sources of error is explored in §3.3.4.
Table 3.8. Summary of parameters used to determine whether a high air-exchange rate results in a significantly higher IPOP for ultrafine particles.

<table>
<thead>
<tr>
<th>Site</th>
<th>Low AER range (h⁻¹)ᵇ</th>
<th>High AER range (h⁻¹)ᵇ</th>
<th>Low IPOP (mean ± sd)ᶜ</th>
<th>High IPOP (mean ± sd)ᶜ</th>
<th>P-valueᵈ</th>
</tr>
</thead>
<tbody>
<tr>
<td>S1</td>
<td>0.7 – 2.7</td>
<td>2.7 – 10.3</td>
<td>0.60 ± 0.11</td>
<td>0.72 ± 0.14</td>
<td>0.03</td>
</tr>
<tr>
<td>S2</td>
<td>0.7 – 2.9</td>
<td>4.7 – 6.5</td>
<td>0.22 ± 0.15</td>
<td>0.68 ± 0.09</td>
<td>&lt;0.01</td>
</tr>
<tr>
<td>S4</td>
<td>0.6 – 1.6</td>
<td>1.7 – 8.7</td>
<td>0.40 ± 0.05</td>
<td>0.75 ± 0.14</td>
<td>&lt;0.01</td>
</tr>
<tr>
<td>S5</td>
<td>0.8 – 1.2</td>
<td>1.3 – 4.7</td>
<td>0.24 ± 0.07</td>
<td>0.34 ± 0.11</td>
<td>0.08</td>
</tr>
<tr>
<td>S6</td>
<td>0.3 – 0.5</td>
<td>0.7 – 4.4</td>
<td>0.45 ± 0.18</td>
<td>0.31 ± 0.21</td>
<td>0.89</td>
</tr>
</tbody>
</table>

ᵃ S3 is not shown because statistical test was not conducted for this site
ᵇ Range in values for air-exchange rates below (i.e. low) and above (i.e. high) the median
ᶜ Mean IPOP for the high and low air-exchange rate groups
ᵈ P-value from a one-sided t-test of the mean IPOP values for the low and high air-exchange rate groups
Figure 3.5. Regression plots of IPOP versus air-exchange rate calculated for unique periods at each site. A dotted line is drawn through the median value of both parameters.
3.3.4 Sensitivity analysis

The air-exchange rate and IPOP results at each site were tested for their sensitivity to the input parameters estimated to have the greatest influence on uncertainty. The median percent change in the air-exchange rate per change in the outdoor CO₂ concentration and the occupant CO₂ emission rate are presented for each site in Table 3.9.

Table 3.9. Median percent change in the air-exchange rates calculated for each site per ±50 ppm change in the outdoor CO₂ concentrations or ±25% change in the occupant CO₂ emission rate.

<table>
<thead>
<tr>
<th></th>
<th>CO₂ out -50 ppm</th>
<th>CO₂ out +50 ppm</th>
<th>E_CO₂ -25%</th>
<th>E_CO₂ +25%</th>
<th>CO₂ out +100 ppma</th>
</tr>
</thead>
<tbody>
<tr>
<td>S1</td>
<td>-16%</td>
<td>+24%</td>
<td>-27%</td>
<td>+27%</td>
<td>+63%</td>
</tr>
<tr>
<td>S2</td>
<td>-19%</td>
<td>+29%</td>
<td>-28%</td>
<td>+28%</td>
<td>-</td>
</tr>
<tr>
<td>S3</td>
<td>-26%</td>
<td>+54%</td>
<td>-29%</td>
<td>+29%</td>
<td>-</td>
</tr>
<tr>
<td>S4</td>
<td>-6%</td>
<td>+8%</td>
<td>-29%</td>
<td>+29%</td>
<td>+17%</td>
</tr>
<tr>
<td>S5</td>
<td>-6%</td>
<td>+7%</td>
<td>-35%</td>
<td>+35%</td>
<td>-</td>
</tr>
<tr>
<td>S6</td>
<td>-7%</td>
<td>+9%</td>
<td>-59%</td>
<td>+59%</td>
<td>-</td>
</tr>
</tbody>
</table>

*a Sensitivity to +100 ppm change in outdoor CO₂ was also tested for the two classrooms with doors leading to an internal hallway.

The median sensitivity across classrooms to ±50 ppm change in the outdoor CO₂ concentration spanned a wider range (6% to 54% change) than did median sensitivity to ±25% change in the occupant CO₂ emission rate (27% to 59% change). The ±50 ppm change in the outdoor CO₂ concentration had the greatest effect on classrooms with a high air-exchange rate. The highest sensitivity was observed at S3, which had an air-exchange rate well above the other classrooms. This result is expected, since classrooms with a higher air-exchange rate will generally have a lower indoor CO₂ concentration, and therefore the term in the denominator of equation (3.8) will be proportionately more affected by a net change in outdoor CO₂ compared to classrooms with a lower air-exchange rate. The ±25% change in the occupant CO₂ emission rate had the greatest influence on air-exchange rate determinations for which the second term in the numerator of equation (3.8) (i.e. \( Y(t_f) - Y(t_i) \)) was positive and significantly above zero. In other words, periods during which the classroom CO₂ concentration progressively increased, such that the classroom CO₂ level at the end of the period was roughly 100 ppm or more above that of the beginning of the period, were the most sensitive to changes in the occupant CO₂ emission rate. Likewise, periods for which the net change in the classroom CO₂ concentration was -100 ppm or less had the least sensitivity to changes in the occupant CO₂ emission rate. In general, classrooms with a lower air-exchange rate were more likely to have periods with a high net increase in classroom CO₂ from beginning to end than were classrooms with a high air-exchange rate, though this wasn’t always the case. For example, S3, which had the highest air-exchange rate, had an identical sensitivity to changes in the rate of occupant generated CO₂ as did S4, even though S4 had an average air-exchange rate a factor of five less than that of S3. The reason for this was that at S4, seven of the eleven analyzed periods had a relatively steady CO₂ concentration because the levels had declined little during the brief period of vacancy that
preceded it. Likewise, two of the eleven periods had a declining CO2 concentration because a window or door had been recently opened.

Air-exchange rate results at S1 and S4 were also tested for their sensitivity to a 100 ppm increase in the outdoor CO2 concentration. This 100 ppm increase is meant to approximate the net increase in the CO2 level of the air that is exchanged with the classroom, when part of the air comes from an internal hallway versus only coming from the outdoors. In the case of S1, this 100 ppm increase in outdoor CO2 resulted in the greatest median percent change in air-exchange rate of all the tested sensitivities for that site (57% increase). In the case of S2, this 100 ppm increase had a less significant effect on the air-exchange rate than did a ±25% change in the rate of occupant generated CO2. Overall, the results of the air-exchange rate sensitivity analysis indicate that the error and variability introduced by the input parameters results in an increase in the uncertainty of the final calculated value of up to 50-60% for S1, S3 and S6, and up to 30-35% for S2, S4 and S5.

The values for IPOP calculated at each site were tested for their sensitivity to a ±50% change in both the input air-exchange rate and particle deposition rate. The results are reported in Table 3.10. Changes in the air-exchange and deposition rate had comparable effects on the IPOP, since they are present as a sum in the IPOP expression (equation (3.13)). The most sensitive IPOP values were for periods during which the indoor PN concentration was low, such that the second term in the numerator of the IPOP expression, \( \int_{t_i}^{t_f} N(t)dt \), was small. Consequently, S5 and S6, which had the lowest indoor PN averages during student occupancy (Table 2.2), had the most sensitive IPOP results to changes in the air-exchange rate and deposition rate. Likewise, S1, which had the highest indoor PN average during student occupancy (Table 2.2), had the least sensitive IPOP results. The IPOP values at S3 and S6 were tested for sensitivity to a factor of 3 decrease in the outdoor PN concentration, due to the distance of these classrooms from the outdoor PN monitor. Since the time-integrated outdoor PN concentration is represented as the denominator of equation (3.13), a factor of 2 decrease in the PN concentration uniformly results in a 50% decrease in all IPOP values. The actual IPOP values that result from this decrease appear unrealistically low, thus it is unlikely that the outdoor PN concentration differed this significantly between the outdoor PN monitor and the classrooms, despite the outdoor monitor’s closer proximity to the road. That said, the results of this sensitivity analysis indicate that the IPOP values calculated for S3 and S6 should be viewed with a higher degree of uncertainty that the values calculated for the other sites. Overall, excluding the sensitivity of S3 and S6 to error resulting from distance to the outdoor monitor, the IPOP results were more robust than the air-exchange rate results, with the median IPOP sensitivities across the sites being consistently less than 10%.
Table 3.10. Median percent change in the IPOP results at each site per ±50% change in the air-exchange rate (AER) or ±50% change in the particle deposition rate (k).

<table>
<thead>
<tr>
<th></th>
<th>Median percent change in IPOP</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>AER -50%</td>
<td>AER +50%</td>
</tr>
<tr>
<td>S1</td>
<td>0.2%</td>
<td>-0.1%</td>
</tr>
<tr>
<td>S2</td>
<td>1.6%</td>
<td>-0.6%</td>
</tr>
<tr>
<td>S3</td>
<td>1.6%</td>
<td>-0.6%</td>
</tr>
<tr>
<td>S4</td>
<td>4.5%</td>
<td>-2.7%</td>
</tr>
<tr>
<td>S5</td>
<td>-9.2%</td>
<td>5.2%</td>
</tr>
<tr>
<td>S6</td>
<td>-3.9%</td>
<td>2.2%</td>
</tr>
</tbody>
</table>

^a Results at S3 and S6 were tested for a factor 2 decrease in outdoor PN due to the distance between the outdoor monitor and the classrooms

3.3.5 Strategies for improving classroom ventilation and reducing particle concentrations

A summary of results for the parameters used to characterize the adequacy of ventilation in the six Bay Area classrooms is provided in Table 3.11. Also included is a suggested action for improving the ventilation characteristics of each of the monitored classrooms. S2 is included within this discussion, but it represents a special case. Specifically, S2 was equipped with an HVAC system; however, it was rarely used, and there is no record of the start and end time for the periods during which it was used during the monitoring period of this study. As a result, while at S2 there is the option of utilizing the HVAC system for continuous mechanical ventilation, particularly during the colder months, no suggestion can be made regarding its use, since it was not characterized as part of this study. Consequently, S2 is primarily treated as a naturally ventilated classroom in the context of this discussion. While the suggestions provided in Table 3.11 are specific to the six classrooms investigated in this study, they are also generally applicable to other classrooms of similar construction and in a similar climate as those described here.

Table 3.11. Summary results of parameters characterizing ventilation at S1-S6.

<table>
<thead>
<tr>
<th>Vent. Mode^a</th>
<th>Median dCO2 (ppm)</th>
<th>% dCO2 &gt; 700 ppm^b</th>
<th>Mean VPP (L/s)^c</th>
<th>Mean AER^d (h⁻¹)</th>
<th>Suggested Action^e</th>
</tr>
</thead>
<tbody>
<tr>
<td>S1</td>
<td>N</td>
<td>237</td>
<td>3</td>
<td>18</td>
<td>3.7 At least one door open</td>
</tr>
<tr>
<td>S2</td>
<td>N</td>
<td>261</td>
<td>5</td>
<td>13</td>
<td>3.6 Door or multiple windows open</td>
</tr>
<tr>
<td>S3</td>
<td>N</td>
<td>130</td>
<td>0</td>
<td>27</td>
<td>10.8 Decrease mech. ventilation rate</td>
</tr>
<tr>
<td>S4</td>
<td>N</td>
<td>665</td>
<td>56</td>
<td>7</td>
<td>2.5 Door or multiple windows open</td>
</tr>
<tr>
<td>S5</td>
<td>M</td>
<td>518</td>
<td>35</td>
<td>4</td>
<td>1.5 Increase mech. ventilation rate</td>
</tr>
<tr>
<td>S6</td>
<td>N</td>
<td>484</td>
<td>33</td>
<td>7</td>
<td>1.1 Door open</td>
</tr>
</tbody>
</table>

^a N= naturally ventilated; M= mechanically ventilated  
^b Percent of time during which students were present that the indoor minus outdoor CO₂ level (dCO₂) exceeded 700 ppm  
^c VPP= ventilation rate per person  
^d AER= air-exchange rate  
^e Suggested action for maintaining an appropriate ventilation rate in the classrooms
Among the four naturally ventilated classrooms, ventilation appeared adequate when a door or multiple windows were maintained in the open position. Conversely, ventilation appeared inadequate when doors and windows were closed. As a result, the suggestion for these naturally ventilated classrooms is that the open configuration (i.e. one door and/or multiple windows open) be maintained for the duration of student occupancy. A concern with maintaining an open door and/or windows in these classrooms year round is that, during the colder months, the occupants will become thermally uncomfortable. In the ASHRAE Fundamentals Handbook (2009), it is recommended that indoor temperatures in classrooms be maintained between roughly 20 and 27 °C. When formulating strategies for achieving a comfortable indoor temperature, the heat generated within the classrooms by the occupants themselves must be considered. Assuming a classroom has a ventilation rate of 5 L/s per person and that the occupants each generate 50 watts of metabolic power, and ignoring modes of heat removal other than by ventilation and heat sources other than the occupants, the classroom temperature would be roughly 10 °C above outdoors. If the classroom ventilation rate is increased to 10 L/s per person, then the indoor to outdoor temperature difference would drop to roughly 5 °C. According to weather data from 2009 for Oakland, which is the largest city in Alameda County, outdoor temperatures of less than 10° C occurred for some portion of the daylight hours on the majority of days in November through February (www.wrh.noaa.gov/mtr). Thus, maintaining an open door and/or windows in these classrooms during the winter months may necessitate some mode of indoor space heating. Heating capability already exists at S4 and S6 via a hot water radiant system and wall mounted package unit, respectively. At S2, the HVAC system would presumably be used for heating and ventilation during the cold months. Therefore, at only one of the naturally ventilated sites investigated in this study, S1, does the introduction of supplemental heaters need to be considered to prevent the teacher from decreasing the classroom ventilation rate in an attempt to raise the classroom temperature.

For the two classrooms with HVAC systems, S3 and S5, a modification in the supply rate of outdoor air by the mechanical ventilation system is suggested. Specifically, at S3, where the indoor minus outdoor CO₂ concentration never exceeded 700 ppm and the rate of ventilation per person was well above the ASHRAE recommendation of 5 L/s, it is suggested that the mechanical ventilation rate be reduced, which should serve to decrease the IPOP. In the case of S5, where the indoor minus outdoor CO₂ concentration exceeded 700 ppm for 35% of the period of student occupancy and the mean rate of ventilation per person calculated for eight periods was below the ASHRAE recommended minimum, it is suggested that the mechanical ventilation rate be increased. The HVAC systems at both S3 and S5 are rooftop packaged units, with the airflow supply rates designed for the purpose of providing the necessary amount of heating and cooling, and not specifically designed for ventilation. The system at S3 is designed to provide 25% more cooling than the system at S5, which partly explains the higher air-exchange rate observed there. Since rooftop systems are designed to provide a specific amount of thermal conditioning, the airflow rates can only be adjusted by roughly 15% or less. However, the ratio of outdoor ventilation air and recirculated classroom air supplied by the HVAC system is determined by the position of the damper. In the case of S5, the low air-exchange rate indicates that the damper is only
partially open, and thus, should be adjusted. Conversely, at S3, the high air-exchange rate indicates that the damper is likely entirely open, and, thus, should be partially closed to allow for a larger fraction of recirculation air.

The suggested actions provided in Table 3.11 were formulated from the standpoint of effectively exhausting indoor emitted pollutants. Next, strategies by which to limit the indoor proportion of outdoor particles in the classrooms will be considered.

In the case of the naturally ventilated classrooms, the strategy suggested for reducing the classroom IPOP is the operation of portable fan-filter air-cleaning units. These air cleaners operate by using fans to force air through a filter media. The influence of the air cleaners on the classroom particle levels is described by the mass-balance expressed in equation (3.14),

\[
\frac{dN_i}{dt} = N_o \cdot \lambda \cdot P - N_i \cdot \left( k + \lambda + \frac{Q_f}{V} \cdot \eta_f \right)
\]

(3.14)

where \( Q_f \) represents the flow rate through the air-cleaner and \( \eta_f \) represents the particle collection efficiency of the air cleaner (other terms were defined in equation (3.1)). Under steady-state conditions, the IPOP of a classroom with an air cleaner present can be described by equation (3.15). This equation is essentially identical to equation (3.11), except for the additional loss term due to the presence of the air cleaner.

\[
\frac{N_i}{N_o} = \frac{P \cdot \lambda}{k + \lambda + \frac{Q_f}{V} \cdot \eta_f}
\]

(3.15)

A modeling analysis was performed to estimate the reduction in the steady-state IPOP that would result in each of the naturally ventilated classrooms, if a portable air-cleaner was present and operated. A popular commercially available fan-filter air cleaning unit was used for the purpose of this analysis. This air-cleaner is reported by the manufacturer to have three different airflow settings. In this case, it is assumed that the teacher will operate the air-cleaner at the “low” or “medium” settings, which have noise levels reported by the manufacturer of 50 and 55 dB, respectively, versus the “high” setting, which has a noise level of 60 dB. Studies have found that noise levels as 58 dB can result in a reduction in students’ health and performance (Lundquist, 2000; Passchier-Vermeer, 2000; Wålinder et al., 2007). The flow rate of the portable air cleaning unit at the “low” and “medium” fan settings is reported by the manufacturer to be approximately 120 and 330 m³/h, respectively. The clean air delivery rate of the unit is represented as a product of its flow rate and particle removal efficiency. Waring et al. (2008) tested the single pass efficiency of two portable filter air-cleaners that reportedly were equipped with high-efficiency particulate air (HEPA) filters and found their efficiencies both to be approximately 60%. They suggest that this relatively low value likely resulted from the bypass of air around the filters or around the air cleaners themselves, and does not reflect the efficiency of the filter media, which by definition is 99.97% or greater. Since the air-cleaner used in the present modeling analysis is also reported by the manufacturer to utilize a HEPA filter, a single-pass efficiency of 60% is used, per the results of Waring et al. (2008a). The air-exchange rates used to represent each site were 3, 3, 2 and 4 h⁻¹ for
S1, S2, S4 and S6, respectively. These air-exchange rates were selected based on results for the “open” ventilation configuration at each site. The particle deposition rate was assumed to be constant at a value of 1 h⁻¹, and the particle penetration efficiency was assumed to be one.

The results for the steady-state IPOP of each classroom utilizing one fan-filter unit at either the low or medium setting are presented in Figure 3.5. The percent decrease in the IPOP across classrooms when the air cleaner was used on the “low” setting ranged from 5% to 9%, and when used on the “medium” setting ranged from 21% to 37%, relative to the uncontrolled IPOP. The degree to which the air filters were able to impact the indoor particle levels was directly related to the air-exchange rate of the classrooms. Specifically, when the classroom air-exchange rate was high, the air cleaner was less able to compete with the rate of particle removal by ventilation, compared to when the air-exchange rate was low. For this reason, the greatest percent decrease in IPOP was observed at S4, with a base air-exchange rate of 2 h⁻¹, whereas the lowest percent decrease was observed at S6, with a base air-exchange rate of 4 h⁻¹. Overall, these results indicate that operating this commercially available air cleaner at the “medium” setting will lead to a substantial decrease in the classroom IPOP; however, this is a viable option only to the extent that the fan noise is not perceived as disruptive to regular classroom activities.

![Figure 3.5](image)

**Figure 3.6.** Modeling results for steady-state IPOP for ultrafine particles when there is no air cleaner, or when one is present and operated at a “low” or “medium” fan setting.

The costs associated with use of air cleaners in a classroom can be divided into the cost for the initial purchase, the cost for maintenance and the cost for energy use. Since the air cleaner assumed to be used in this analysis is commercially available, these three categories of expenses can be quantified in 2010 dollars using information published by the manufacturer. The air cleaner has an initial cost of approximately $500. Every five years, the manufacturer suggests that the filter be replaced with a $200
replacement filter. The unit is reported by the manufacturer to draw 85 watts of power at the medium setting. Based on the cost of electricity in California of approximately $0.15 per kilowatt hour, and assuming that the air cleaner is operated for approximately 8 hours a day for 20 days per month, the cost of energy is estimated to be approximately $2.00 per month. Assuming that school is in session for 9 months out of the year, the energy cost per year for one air cleaner is roughly $20. Thus, the total cost per year for operating the air cleaner assuming 10 years of use would be roughly $90: $50/year for the air-cleaner, $20/year for a replacement filter and $20/year for electricity. In a letter to the community giving a budget update, the superintendent of the district containing classrooms S3-S6 reported that in 2010, the district had been given $5214 per pupil funding from the California government. Assuming that there are roughly 20 students per classroom. The cost per student of having one air cleaner per classroom per year would be roughly $4.50 or only 0.09% of the budgeted funding per child.

Since the mechanically ventilated classrooms already have a built-in filtration system, employing a higher efficiency filter within the duct is a more straightforward strategy for reducing the classroom IPOP than would be the use of a portable air cleaner. In addition, removing particles via an in-duct filter is a more effective method for reducing the indoor levels of outdoor particles, since it represents a closed path versus open path configuration. Specifically, in the case of an in-duct filter, contaminated outdoor air must pass through the filter prior to entering the classroom. Conversely, in the case of a portable air-cleaning unit, the contaminated air enters the classrooms and may possibly reach the breathing zone of the students without ever passing through the air cleaner (Nazaroff and Weschler, 2009).

The influence on classroom particle levels of an HVAC system supplying a combination of outdoor and recirculated air is described by equation (3.16),

\[
\frac{dN_i}{dt} = N_o \frac{Q_o}{V} (1 - \eta_m) + N_i \frac{Q_r}{V} (1 - \eta_m) - N_i \frac{(Q_r + Q_o)}{V} - N_i k
\]  

(3.16)

where \(Q_o\) is the supply rate of outdoor ventilation air, \(Q_r\) is the supply rate of recirculated indoor air, \(Q_o + Q_r\) is the total airflow supply rate of the HVAC system and \(\eta_m\) is the effective particle removal efficiency of the mechanical filter. The assumptions in this case are that doors and windows are closed (i.e. \(\lambda_n \sim 0\)) and air-exchange rate due to infiltration is negligible (i.e. \(\lambda_i \sim 0\)). Using a steady-state approximation, the IPOP of a classroom under these conditions can be described by equation (3.17).

\[
\frac{N_i}{N_o} = \frac{\frac{Q_o}{V} (1 - \eta_m)}{\frac{Q_r + Q_o}{V} + k - \frac{Q_r}{V} (1 - \eta_m)}
\]  

(3.17)

The greater effectiveness of closed-path versus open-path filtration is illustrated by a comparison of the IPOP expressions represented by equations (3.17) and (3.15). Specifically, in equation (3.15), the clean air delivery rate of the fan-filter unit is one of three competing particle removal mechanisms represented in the denominator of the expression. Conversely, in equation (3.17), the filter efficiency is manifested in the
The efficiency of commercially available in-duct filters are rated according to the Minimum Efficiency Reporting Value (MERV), which is evaluated based on ANSI/ASHRAE Standard 52.2. The MERV rating is specified as an integer from 1 to 20. Filter classification is based on the collection efficiency of 3 to 10 µm particles for the 1 to 8 rating, 1 to 10 µm particles for the 9 to 12 rating, 0.3 to 10 µm particles for the 13 to 16 rating, and <0.3 to 10 µm particles for the 17 to 20 rating. During the period that the classrooms were monitored, S5 was equipped with a MERV 7 filter and S3 was equipped with two filters erroneously installed in parallel, one of which was a MERV 4 filter. Since the MERV rating system is primarily based on the collection efficiency of particles in the accumulation and coarse mode size ranges, the corresponding collection efficiency of ultrafine particles is not well characterized. Kowalski and Bahnfleth (2002) modeled the size-resolved collection efficiency of MERV 6, 8 and 11-16 rated filters down to a 0.01 µm particle size. By extending the results reported by Kowalski and Bahnfleth and assuming that filter efficiency results for a 0.04 µm particle are approximately representative of the average for the ultrafine particle size range, it is estimated that the ultrafine particle collection efficiency of a MERV 7 filter is roughly 20% and of a MERV 4 filter is <10%. Therefore, switching the filters in the HVAC systems at S3 and S5 to ones with either a MERV 12 (~60% collection efficiency of 0.04 µm particles) or MERV 14 rating (~90% collection efficiency of 0.04 µm particles) would result in a decrease in the concentration of particles entering via the mechanical ventilation system by roughly 40-50% or 80-90%, respectively, assuming negligible bypass of air around the filter. However, the assumption regarding filter bypass is an important one to consider, since research has indicated that even a moderate gap can significantly erode the overall efficiency of filtration. Ward and Siegel (2005) found in a modeling analysis that a 10 mm gap between the filter and duct resulted in a decrease in the ultrafine particle collection efficiency of a MERV 15 filter by ~60%, a MERV 11 filter by ~10%, and a MERV 6 filter by ~5%. A 1 mm gap was found to only affect the efficiency of the MERV 15 filter, which decreased by ~1%.

In the present study, we were able to inspect the HVAC system at S3 approximately one year following the completion of monitoring at that site and observed a gap between the duct and MERV 4 rated filter of approximately 5 mm. An inspection of the HVAC system at S5 was not made. Consequently, although changing the filter in the HVAC system of these classrooms to one with a higher MERV rating could reduce the resulting classroom IPOP, the full benefit of this reduction would be eroded if there were gaps around the filter. Gaps around high efficiency filters have a greater influence on the effective particle collection efficiency than for low efficiency filters, so efforts should be made to minimize filter bypass in any system that uses high efficiency filters. Figure 3.6 shows the IPOP for ultrafine particles that would result in S3 and S5 from use of a mechanical filter having a MERV rating of 4, 6 or 12. In the case of the MERV 12 filter, the effect of a 10% loss in collection efficiency due to filter bypass is also considered. The filter removal efficiencies for ultrafine particles were estimated based on results provided by Kowalski and Bahnfleth (2002). The total mechanical airflow supply rates were assumed to be 2500 and 2000 m³/h at S3 and S5, respectively, based on the size of the HVAC systems. Both systems were assumed to supply 70% outdoor air and
30% recirculated air. The results of this analysis indicate that increasing the MERV rating of the in-duct filter from 4 to 7 would result in a ~20% decrease in the IPOP, and from 4 to 12 would result in a 65% decrease in the IPOP. In this case, a 10% decrease in the efficiency of the MERV 12 filter, owing to filter bypass, had a relatively small influence on the outcome. Overall, a comparison of Figures 3.5 and 3.6 indicates that using high-efficiency in-duct filters to treat classroom air results in a more significant reduction in the IPOP for ultrafine particles than does use of a portable fan-filter air-cleaning unit.

![Figure 3.7](image)

**Figure 3.7.** Modeling results for steady-state IPOP for ultrafine particles, assuming the presence of an in-duct filter with MERV rating of 4, 7 or 12. In the case of the MERV 12 filter, the influence of filter bypass is considered.

Based on a 2010 price search, filters with MERV ratings of 12, 7, and 4 were found to have initial costs of roughly $20, $8 and $3, respectively (for filter dimensions of 51 cm x 64 cm with varying thickness). Therefore, if the filters in the HVAC systems at S3 and S5 were changed to ones with MERV 12 ratings, the initial cost would be minimal relative to the cost of purchasing a portable air-cleaning unit. However, replacing the filters in these ducts with ones having higher particle-removal efficiencies is also expected to incur a higher operational cost. Specifically, Waring and Siegel (2008) found in a modeling analysis that a MERV 12 filter in an HVAC system operating in an urban area for 12 hours per day, with 50% outdoor and 50% recirculated air, will have a particle loading rate of roughly 200 g/m² per month, which is almost a factor of two greater than the predicted loading rate for a MERV 6 filter under the same conditions. They further predict that the pressure drop in the mechanical system induced by a MERV 12 filter loaded at this rate will increase by a factor of 2.5 in the first month, which is ~1.5x greater than the increase in pressure drop induced by a MERV 6 filter under the
same conditions. Research has shown that, for large commercial systems, a higher pressure drop filter causes the fan motor to draw more power to deliver the required amount of air (Fisk et al., 2002), thus resulting in increased energy use. This association between energy use and filter pressure drop has generally been assumed to hold true for smaller residential and light-commercial systems. However, in a field study involving measurements in 17 residential and light-commercial buildings it was observed that high-efficiency filters do not have much of an impact on energy consumption in forced air cooling systems (Stephens et al., 2010). This result occurred primarily because smaller systems do not have a required volume of air that must be delivered to the conditioned space. Consequently, rather than increasing fan power, this pressure drop instead generally diminishes the system flow rate. Stephens et al. (2010) found that among 16 sites during fan only operation, the median airflow rates were approximately 4-5% lower with a MERV 11-12 filter installed compared to a MERV 2 filter. Considering the relatively large range of air-exchange rates observed within these classrooms, a 4-5% decrease in the ventilation rate resulting from use of a high efficiency filter is insignificant.

Although leaving an excessively loaded filter within the HVAC systems in these classrooms is not expected to result in an increase in energy use, it could have a negative impact on indoor air quality. Specifically, organic compounds associated with captured particles may desorb into air passing through the filter, or may react with ozone to produce harmful byproducts, such as formaldehyde (Hyttinen et al., 2006; Destaillats et al., 2008). In addition, heavily particle-laden filters have been associated with the decreased comfort and work performance of the occupants of the conditioned space (Clausen, 2004). For these reasons, the HVAC filters in classrooms should be regularly replaced. The lifetimes of MERV 12 filters operating in the HVAC systems at S3 and S5 were estimated using equation (3.17). This equation was derived by Fisk et al. (2002) for the purpose of performing a cost and benefit analysis of the use of high efficiency filters in HVAC systems of large commercial buildings.

\[
\text{Filter Lifetime} = \frac{\text{Dust Holding Capacity}}{\text{(Arrestance) (Inlet PM}_{10}) (\text{Air Flow Rate})}
\] (3.17)

A commercially available MERV 12 filter was selected for the purpose of the cost analysis of S3 and S5. The manufacturer reports that this filter has a dust holding capacity of 110 g. The arrestance of a MERV 12 filter, according to ASHRAE 52.1 test method, is 99%. An airflow rate of 2500 and 2000 m³/h was used for S3 and S5, respectively, and the systems were assumed to operate for 12 hours per day and 20 days per month. A value of 30 µg/m³ was used to represent the inlet PM_{10} concentration for both S3 and S5, based on the average concentration reported by Kim et al. (2004) for measurements made on the grounds of ten schools in the San Francisco Bay Area. It is recognized that the inlet concentration is not likely to be equivalent to that of the outdoors, since there is an assumption that the systems will operate with 30% recirculated air; however, it is unclear whether the indoor concentration will be elevated or suppressed relative to outdoors, since resuspension of particles due to human activity has been found to elevate PM_{10} concentrations in school classrooms (Fromme et al., 2007; Parker et al., 2008; Tippayawong et al., 2009). As a result, 30 µg/m³ is used to represent a reasonable approximation for the inlet concentration.
Based on this analysis, the estimated lifetime of the MERV 12 filter at S3 is 6 months and at S5 is 8 months. The manufacturer of this filter recommends that it be replaced every 3 months, but no information to support this recommendation is provided. For the purpose of this analysis, it is estimated that the filter will be replaced in the classrooms 3 times per year, or roughly every 3 months for the months that school is in session (i.e., the filters would be replaced in August, November and March). Fisk et al. (2002) estimated that the time required to change a filter in a commercial HVAC system ranges between 3 to 13 minutes, with an associated range in labor cost of $5 to $10. Bekö et al. (2008) estimated the cost of disposing a filter as $5 per filter. Thus, a conservative estimate of the cost required to purchase and maintain high-efficiency HVAC filters in each classrooms is $105 per year ($60 for purchase of three filters, $30 for the labor required to replace the filters and $15 for filter disposal). This cost amounts to $5.25 per student, in a 20 student classroom, or 0.1% of the budgeted funding per child. Therefore, the cost of maintaining a high-efficiency in-duct filter in S3 and S5 will be roughly the same as the cost of operating a fan-filter air-cleaner in the four naturally ventilated classrooms.

Quantifying the benefits associated with decreasing the classroom IPOP is a helpful measure towards justifying the cost. In this case, the primary benefit is assumed to be an improvement in the health, and perhaps learning performance, of the students. Unfortunately, at present, the health benefits associated with decreased exposure to ultrafine particles are not well known. However, sufficient information does exist to quantify the decrease in particles inhaled by each child per year, resulting from the implementation of the suggested filtration strategies. Specifically, the average decrease in the classroom PN concentration resulting from improved particle filtration can be estimated by multiplying the fractional change in the IPOP by the average outdoor PN concentration. This result effectively represents the increment in the classroom PN concentration to which the students would not be exposed, owing to enhanced filtration. Multiplying this result by the total volume of air the students are estimated to breath each year while in their classroom yields an estimate of the number of particles that the students would not inhale, as a result of the implemented filtration strategy. The parameters used to make this estimate are summarized in Table 3.10. Specifically, the average decrease in the classroom IPOP estimated to result from the implementation of improved particle filtration (i.e., using a fan-filter air-cleaner on the medium setting or a MERV 12 in-duct filter) relative to the baseline case (i.e., no air-cleaner or a MERV 4 in-duct filter) is approximately 0.2 and 0.5 in the naturally and mechanically ventilated classrooms, respectively. The average outdoor PN concentration measured during hours of student occupancy at the six classrooms is $18 \times 10^3 \text{ cm}^{-3}$. Assuming that this is approximately representative of this region for the duration of the year, the reduction in the indoor particle concentration, owing to the reduction in the IPOP, would be 3.6 and $5.8 \times 10^3 \text{ cm}^{-3}$ for the naturally and mechanically ventilated classrooms, respectively. Then, assuming that elementary school children have a breathing rate of approximately $0.8 \text{ m}^3/\text{h}$ (US EPA, 1989), and that they spend 4.5 hours per day (average for the classrooms in this study) and 180 days per year in their classrooms, the number of particles not inhaled owing to increased particle filtration would be 2.3 and $4.7 \times 10^{12}$ per child per year, for the naturally and mechanically ventilated classrooms, respectively. Thus, for these classrooms, the cost for $1 \times 10^{12}$ fewer particles inhaled per child per year
is roughly $1.90 and $1.10 for the naturally and mechanically ventilated classrooms, respectively. This metric will ultimately allow comparison to other interventions to reduce children’s exposure to ultrafine particles.

**Table 3.12.** Summary information used to calculate the number of particles *not* inhaled per child per year resulting from implementation of improved filtration strategies in classrooms.

<table>
<thead>
<tr>
<th>Classroom Type</th>
<th>Breathing Rate (m³/h)</th>
<th>Hours/day in class</th>
<th>Days/year at school</th>
<th>PN out (10³ cm⁻³)</th>
<th>Δ IPOP</th>
<th>Δ PN in (10³ cm⁻³)</th>
<th>Particles not inhaled (#/y/child)</th>
</tr>
</thead>
<tbody>
<tr>
<td>NV</td>
<td>0.8</td>
<td>4.5</td>
<td>180</td>
<td>18</td>
<td>0.2</td>
<td>3.6</td>
<td>2.3 x 10¹²</td>
</tr>
<tr>
<td>MV</td>
<td>0.8</td>
<td>4.5</td>
<td>180</td>
<td>18</td>
<td>0.5</td>
<td>9.0</td>
<td>4.7 x 10¹²</td>
</tr>
</tbody>
</table>

*“NV” = naturally ventilated classroom; “MV” = mechanically ventilated classroom*

### 3.4 Conclusion

Observational monitoring data collected for 2 to 4 days in each of 6 classrooms in the San Francisco Bay Area were analyzed to characterize classroom ventilation rates, as well as to assess the relationship between classroom air-exchange rate and the indoor proportion of outdoor ultrafine particles (IPOP). Across the 6 sites, the median value for the indoor minus outdoor CO₂ concentration during periods of student occupancy ranged from approximately 130 ppm at a classroom with a mechanical ventilation system operating at a high flow rate, to 665 ppm at a classroom that was naturally ventilated. The mean rate of ventilation per person calculated for periods of student occupancy at the 6 sites ranged from 4 to 27 L/s with the 2 mechanically ventilated classrooms lying on opposite ends of the spectrum and the 4 naturally ventilated classrooms falling between. The mean air-exchange rates calculated for a subset of periods during student occupancy at the two mechanically ventilated classrooms were 1.5 and 10.8 h⁻¹, and at the 4 naturally ventilated classrooms ranged from 1.1 to 3.7 h⁻¹. At the naturally ventilated classrooms, the air-exchange rate when multiple doors and windows were open was generally an order of magnitude higher than when the classroom was occupied yet closed. Consequently, classrooms that had doors and windows open for a larger fraction of time were characterized by a higher time averaged air-exchange rate. Not surprisingly, the extent of door and window opening was related to thermal comfort conditions, with the minimum occurring at the site monitored in December when the outdoor air was cold.

The results for the naturally ventilated sites indicate that the classrooms were adequately ventilated for removal of bioeffluents when in the open configuration, but inadequately ventilated when in the closed configuration. Consequently, it is suggested that teachers in naturally ventilated classrooms maintain the room in the open configuration for the duration of student occupancy, and that accommodations be made for the control of outdoor pollutants and for heating when necessary. The mechanically ventilated classrooms produced differing results, with the air-exchange rate at one appearing unnecessarily high and at the other appearing too low. These results indicate that greater attention may need to go towards ensuring that classrooms are neither inadequately ventilated nor are they supplied with an unnecessarily high ventilation rate, leading both to an increased energy cost and to elevated indoor proportions of outdoor pollutants.
Since higher air-exchange rates were found to be associated with a higher mean IPOP, it is suggested that actions to increase classroom ventilation should be accompanied by strategies to reduce indoor PN levels. In naturally ventilated classrooms, the suggested strategy for reducing particle concentrations is to utilize a portable fan-filter air-cleaning unit. In the mechanically ventilated classrooms, the suggested strategy is first to replace the existing in-duct filters with ones having high MERV-ratings. The results of this study indicate that PN levels can be reduced in classrooms for a relatively low cost. However, justification of this expense, no matter how small, would benefit from increased evidence from health studies of a link between elevated PN concentrations and degraded student health and performance.

Since this study included only a small number of classrooms, these results cannot be considered as broadly representative of all school conditions, but rather lend insight into the challenge of ensuring that classrooms not only have adequate ventilation for the removal of indoor generated pollutants, but also provide appropriate protection from outdoor pollutants.
Chapter 4: Ultrafine particle concentrations and exposures in four high-rise Beijing apartments

4.1. Introduction

Despite the fact that roughly 20% of the world’s population lives in China, relatively few studies have been published in international journals investigating ultrafine particle (UFP) concentrations in the Chinese atmosphere (Cheng et al., 2008; Gao et al., 2007; Gao et al., 2009; Kivekäs et al., 2009; Laakso et al., 2006; Li et al., 2007; Liu et al., 2008; Wang et al., 2008a; Wehner et al., 2008; Westerdahl et al., 2009; Wang et al., 2009a; Wu et al., 2007; Wu et al., 2008; Yu et al., 2005; Yue et al., 2010a; Yue et al., 2010b). I have found no studies published in English-language journals reporting UFP concentrations measured inside urban residences in mainland China. Since people living in urban environments in China have been found to spend roughly 90% of their time indoors (Wang et al., 2008b), it is important that the research community gain an understanding of indoor air contaminant levels experienced by urban Chinese residents, including those living in Beijing.

Indoor UFP concentrations are strongly influenced by outdoor levels; however, removal processes reduce the indoor UFP levels that originate outdoors to degrees that vary among buildings and with time. Furthermore, indoor sources can contribute significantly to indoor UFP levels, sometimes causing indoor levels to exceed those outdoors. In the past two decades, several published studies from the US, Europe, Australia, Taiwan and Singapore have investigated UFP concentrations in homes (Wallace, 2006; He et al., 2004; Li et al., 1993; Balasubramanian and Lee, 2007; Matson, 2005; Bhangar et al., 2011). These studies have found that average UFP levels within residences can vary by as much as two orders of magnitude, and that indoor levels can be an order of magnitude or more above those outdoors during indoor peak events resulting from activities such as cooking, using candles, and other indoor combustion or high-temperature activities. Other studies have found that cleaning activities involving terpene-scented products can also lead to the formation of high concentrations of ultrafine particles, owing to reactions with ozone (Long et al., 2000; Coleman et al., 2008).

To begin to address the lack of information about UFP levels and exposures in mainland Chinese residences, we measured particle number (PN) concentrations inside four apartments in high-rise buildings in Beijing. The PN measurements were made continuously with 1-minute time resolution for 48-71 hours at each site, using a water-based condensation particle counter with a minimum particle size detection limit of 6 nm. At two of the apartments, outdoor PN concentrations were measured at the same time as indoor monitoring occurred. Coincident data on apartment occupancy and resident activities were collected using data loggers and resident-maintained journals. The data were interpreted to characterize occupants’ exposures to ultrafine particles within their home, to apportion those exposures between hours spent asleep and awake, and to link observed indoor PN peaks with specific resident activities. At sites for which both indoor and outdoor PN data were available, residential exposures were apportioned to indoor and outdoor sources. Where possible, indoor peak events were further analyzed to calculate the magnitude of source emissions, as well as the rate of subsequent particle decay. These results were compared with those from a similar study recently carried out in the San Francisco Bay Area to explore the influence of differences in occupancy patterns, source activities and
ventilation characteristics on the resulting exposure levels. The aim of this study is to contribute to a characterization of three main factors: (1) the indoor UFP exposure concentrations experienced by individuals living in high-rise buildings in the Beijing urban environment; (2) the contribution of indoor and outdoor sources to those exposures; and (3) the source activities that are responsible for indoor generated particles.

4.2. Methods
4.2.1 Site description

A convenience sample of four apartments in different high-rise buildings in Beijing was monitored between June and August 2009. Table 4.1 presents a summary description of the sites. A1, A3 and A4 were considered middle-class apartments, and apartment A2 was considered an upscale apartment. Three of the apartments (A1, A3, A4) were in the northern half of the Haidian district in NW Beijing, and the fourth apartment (A2) was in the northern half of the Chaoyang district in NE Beijing. A map of Beijing with the locations of these four apartments identified is shown in Figure 4.1. The majority of industry in Beijing is concentrated in the southwest and west districts, whereas the majority of high-quality commercial housing is located in the north and northwest districts (Hu and Kaplan, 2001).

The distance between the ground floor of the apartment building and the nearest major roadway varied from 20 m at A4 to 150 m at A2. The building floor on which the apartment was located varied from the 7th floor at A1 to the 23rd floor at A2. Apartment A1 is situated on the northern half of a 17-floor building with an identical building located ~20 m to the west and ~100 m to the north. Apartment A2 is situated on the southeast corner of a 28-floor building with an identical building located ~30 m to the east and two 25-floor buildings located ~100 m to the south. Apartments A3 and A4 are located in identical 19-floor buildings separated by 50 m of open space. Apartment A3 is situated on the southwest corner of the building, immediately north of A4, with an identical building situated ~20 m southeast, which is ~20 m to the northeast of A4. Apartment A4 is situated on the southeast corner of a building that is adjacent to a roadway. Diagrams illustrating the location of each apartment relative to neighboring buildings and major roadways are shown in Figure 4.2.

The four apartments were similar in volume, with a mean ± standard deviation of 223 ± 42 m³. A floor plan of each apartment is shown in Appendix C. Each apartment was equipped with a natural gas stove without pilot lights. A married couple lived in each apartment. At A2, the resident couple also lived with their 3-year-old daughter, and at A4 the resident couple lived with their adolescent son, elderly mother, and two housekeepers. For most of the year at A4 only one housekeeper is present, but for two days of the three-day monitoring period a new housekeeper was in training, increasing the number of residents to six from the usual five. Throughout the monitoring period at each site, the residents were requested to carry out their typical daily activities. At all sites excluding A2, windows were open for a majority of the monitoring period, enhancing air exchange with outdoors.

We note that this sample represents a small set of buildings that were only monitored during one summer season. Results might vary for other types of Beijing housing (e.g., low-rise buildings and courtyard houses) and for other seasons (e.g., during cold winters).
Figure 4.1. Location of apartments sites sampled in Beijing (map from maps.google.com)
Table 4.1. Description of apartments.

<table>
<thead>
<tr>
<th>Site</th>
<th>Building Location</th>
<th>Apartment</th>
<th>Residents</th>
<th>Ventilation</th>
</tr>
</thead>
<tbody>
<tr>
<td>A1</td>
<td>Haidian District, Suzhoujiej (NW); ~650 m S of 4th Ring Road; ~50 m E of Wanquanhe Road.</td>
<td>7th floor</td>
<td>Multi-cultural couple: R1 - Young adult male (American) R2 - Young adult female (Chinese)</td>
<td>Windows open for “awake” hours and 1st night “asleep” hours. Windows closed for 2nd night “asleep” hours.</td>
</tr>
<tr>
<td></td>
<td></td>
<td>2 bedrooms</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>1 bathroom</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>180 m³ N-facing windows</td>
<td></td>
<td></td>
</tr>
<tr>
<td>A2</td>
<td>Chaoyang District, Wangjing (NE); ~1.9 km N of 4th Ring Road; ~2.3 km S of 5th Ring Road; ~150 m W of Guangshun North Street.</td>
<td>23rd floor</td>
<td>American family: R1 - Young adult male R2 - Young adult female R3 – Female toddler</td>
<td>Windows closed for duration of monitoring.</td>
</tr>
<tr>
<td></td>
<td></td>
<td>2 bedrooms</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>2 bathrooms</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>280 m³ S&amp;E-facing windows</td>
<td></td>
<td></td>
</tr>
<tr>
<td>A3</td>
<td>Haidian District, Wudaokou (NW); ~700 m N of 4th Ring Road; ~70 m N of Chengfu Road.</td>
<td>16th floor</td>
<td>Chinese couple: R1 - Elderly male R2 - Elderly female</td>
<td>Windows open during “awake” hours and closed during “asleep” hours.</td>
</tr>
<tr>
<td></td>
<td></td>
<td>3 bedrooms</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>1 bathroom</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>210 m³ S&amp;W-facing windows</td>
<td></td>
<td></td>
</tr>
<tr>
<td>A4</td>
<td>Haidian District, Wudaokou (NW); ~700 m N of 4th Ring Road; ~20 m N of Chengfu Road; ~50 m S of A3.</td>
<td>14th floor</td>
<td>Chinese family: R1 - Middle-aged female R2 - Middle-aged male R3 - Adolescent male R4 - Elderly female R5, R6 - Young adult females (housekeepers)</td>
<td>Windows open for duration of monitoring.</td>
</tr>
<tr>
<td></td>
<td></td>
<td>3.5 bedrooms</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>1 bathroom</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>220 m³ S&amp;E-facing windows</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Figure 4.2. Diagrams illustrating location of apartments relative to roadway, predominant wind direction, and neighboring buildings (not to scale). North is at approximately 0º (i.e. vertical in the upwards direction) in all cases.
4.2.2 Instrumentation and data collection

Particle number (PN) concentrations were measured using a water-based condensation particle counter (CPC, TSI model 3781). Since ultrafine particles typically comprise a large majority of PN concentrations, the data provided by the 3781 are deemed a suitable surrogate measure of UFP concentration, at least for those particles larger than the minimum detectable size. The manufacturer reports that the instrument has a minimum detection limit of 6 nm (50% response level) for a hydrophilic organic aerosol (sucrose). The detection limit for aged ambient aerosol and for fatty acid aerosol produced by cooking would likely be comparable, whereas the lower particle-size detection limit of hydrophobic organic aerosols (e.g., from fresh motor vehicle exhaust) would likely be higher. Wallace et al. (2008) characterized UFP emissions from natural gas cooking for particle sizes of 2-64 nm. For frying they reported (in their supporting information, Table S2) count-based particle size distributions with GM = 16-20 nm and GSD = 1.7-1.8. Lognormal distributions with these parameters would have more than 95% of the particles larger than 6 nm. Particles associated with simmering (cooking rice and reheating food) were smaller, with GMs of 8.5 and 5.5 nm, respectively. In the apartments we monitored, water was boiled using electric kettles or hot-water dispensers. Given the 6 nm cutoff of the CPC, it is possible that we missed some particles from certain types of cooking.

A second CPC, available for the full monitoring period at A1 and A3 and for a 12-hour period at A4, was used to measure outdoor concentrations at the same time as indoor monitoring was conducted.

The instruments were received from the manufacturer immediately prior to the commencement of fieldwork, and had therefore been recently calibrated. Side-by-side monitoring with the two instruments was conducted for three hours in the field prior to monitoring at A1 and for four hours in the laboratory prior to monitoring at A3. Readings from the two instruments agreed to within 6%. Because of the recent calibrations and because of the close agreement in side-by-side sampling, the raw instrument data were used without adjustment.

The indoor CPC was placed in a location assumed to be representative of the primary occupied space during both day and night. While using a single monitor to represent the entire occupied space is not ideal, other studies have found minimal room to room variation of particle mass concentrations within a home (Jones et al., 2007; Ju and Spengler, 1981). At A1, A3 and A4 the CPC was placed in the living room; at A2 it was placed in the master bedroom near the doorway. Carbon-impregnated flexible tubing was attached to the inlet of the CPC and positioned such that air was sampled from a height of 1–1.5 m above the floor. When the second CPC was available, it was placed indoors near a window and carbon-impregnated flexible tubing was used to sample from an outdoor position roughly 1 m outside of the window.

Data on the residents’ indoor occupancy and basic activities were collected via temperature and state-change sensors as well as by means of an activity journal maintained by the residents themselves. Four state-change sensors (HOBO U9) were placed on windows and doors that were reported by the residents to be the most frequently used, to collect information pertaining to air-exchange between rooms in the apartment, and between the apartment and the outdoors. Temperature sensors (HOBO U10) were placed in locations where they would provide a record of times when activities suspected to generate particles took place (e.g. stove-top cooking or use of a toaster oven). Prior to the commencement of monitoring at each site, residents were presented with an activity and occupancy journal, and they were asked to record the time, duration and description of potential source activities (e.g. cooking, indoor combustion, use of heated surfaces, and use of scented products), as well as the times when each resident was
at home awake, at home asleep, or away from home. Near the end of each 24-hour segment of monitoring, a researcher visited the field site to check on instrument performance and to ensure that the journal had been appropriately completed for the prior day.

Centrally monitored PM$_{2.5}$ concentrations, as well as wind speed, wind direction, temperature, and relative humidity were obtained from a monitoring site on the Tsinghua University campus, where average measurements were recorded at 30-minute intervals. Apartments A1, A2, A3, and A4 were located at distances of approximately 4 km, 12 km, 1 km and 1 km from the central monitoring site, respectively.

4.2.3 Data analysis

Average indoor PN exposure concentrations and daily-integrated exposure

Average PN concentrations inside the apartments, and outside when available, were calculated for the full monitoring period, as well as for the times when all residents were (a) at home awake, (b) at home asleep, and (c) away from home. At A4, three of the residents were away from Beijing for some portion of the monitoring period; consequently, for those periods, their activity status was not considered when determining these averages.

The average daily-integrated PN exposure resulting from residential indoor exposures was calculated for each resident of each of the four monitored apartments using equation (4.1),

\[
\overline{\text{Exp}} = \frac{N_{\text{awake}}h_{\text{awake}} + N_{\text{asleep}}h_{\text{asleep}}}{d_{\text{monitored}}}
\]  (4.1)

where \( \overline{\text{Exp}} \) is the resident’s daily-integrated exposure to PN concentrations during hours spent at home (cm$^{-3}$ h d$^{-1}$), \( N_{\text{awake}} \) is the average indoor concentration during the monitored hours spent at home and awake \( (h_{\text{awake}}) \), \( N_{\text{asleep}} \) is the average indoor concentration during the monitored hours spent at home and asleep \( (h_{\text{asleep}}) \), and \( d_{\text{monitored}} \) is the number of days over which monitoring occurred, evaluated as the number of hours monitored divided by 24 (h d$^{-1}$). The additive terms on the right side of equation (4.1) represent, respectively, the contributions to daily residential PN exposure when awake and when asleep.

Indoor PN source events

For all periods of monitoring during which both indoor and outdoor PN data were available, indoor peak events were analyzed to obtain two parameters: the theoretical peak incremental particle concentration generated by the event \( (\alpha V) \), in units of particles per cm$^3$, and the particle loss rate following the event owing to internal processes plus air exchange \( (k+\lambda) \), in units of h$^{-1}$. The first-order internal loss-rate coefficient, \( k \), would include loss by deposition to indoor surfaces, internal air filtration (when present) and any decrease in particle number concentration owing to coagulation or evaporation.

Analysis was conducted in four stages. The first stage involved determining the indoor proportion of outdoor particles (IPOP) (Riley et al., 2002). The starting point is this material balance equation:

\[
\frac{dN_{\text{in}}}{dt} = \frac{E}{V} + (\lambda_N + P\lambda_I)N_{\text{out}} - (k + \lambda_N + \lambda_I)N_{\text{in}}
\]  (4.2)
Here, $N_{in}$ is the indoor PN concentration (cm$^{-3}$), $N_{out}$ is the outdoor PN concentration (cm$^{-3}$), $E$ is count-based particle emission rate from indoor sources (h$^{-1}$), $V$ is the apartment volume (cm$^3$), $\lambda_N$ is the air-exchange rate owing to natural ventilation (h$^{-1}$), $\lambda_I$ is the air-exchange rate owing to infiltration (h$^{-1}$), $P$ is the particle penetration efficiency via infiltration pathways (-), and $k$ is the particle loss-rate coefficient for all interior processes other than ventilation (h$^{-1}$). For time-averaged periods during which $N_{in}$ is approximately at steady-state (i.e., $\frac{dN_{in}}{dt}$ is approximately equal to zero), equation (4.2) can be simplified as follows:

$$\overline{N_{in}} = \frac{E}{V(k + \lambda)} + \frac{(\lambda_N + P\lambda_I)}{(k + \lambda)}\overline{N_{out}}$$

(4.3)

where $\lambda = \lambda_I + \lambda_N$, the overbars denote time averages, and the ventilation and deposition terms are assumed to be either time invariant or appropriate time averages. In the absence of indoor sources ($E = 0$), equation (4.3) can be rewritten as follows,

$$\overline{N_{in}} = f \times \overline{N_{out}}$$

(4.4)

The parameter $f$ represents the IPOP for PN concentrations. A comparison of equations (4.3) and (4.4) shows that $f$ is a function of the air-exchange rate, internal particle loss rate, and particle penetration efficiency.

Values of $f$ were calculated for specific segments of experimental data at sites where both indoor and outdoor PN data were available, and for periods when the indoor PN level was not evidently influenced by indoor sources. The appropriate data were grouped according to common ventilation modes, and the values of $f$ were computed as the ratio of the indoor and outdoor average PN levels for each grouping.

In the second stage of analysis, we characterized indoor peak events by estimating a net indoor PN concentration attributable to indoor sources ($N_{in\_net}$), using equation (4.5).

$$N_{in\_net}(t) = N_{in}(t) - f \times N_{out}(t)$$

(4.5)

This expression implicitly assumes that $f$ is appropriately applied in a time-dependent manner even though it is derived from a time-average relationship between indoor and outdoor particle levels. This approach works well when $N_{out}(t)$ does not vary strongly with time during an indoor peak event, or when the peak indoor concentration is very much larger than the estimated contribution from outdoor particles (i.e., when $N_{in}(t) \gg f \times N_{out}(t)$). Using this approach, we are unable to detect the influence of small episodic or persistent indoor emission sources. There is no clear indication that such sources contributed substantially to indoor levels at these sites.

For the third step, the particle loss rate for each peak event was calculated by fitting an exponential curve to the decay period of the peak in $N_{in\_net}$ and interpreting the decay constant as an estimate of $k + \lambda$ for the respective event. For the fourth and final step, the time integral of $N_{in\_net}$ was calculated for each peak over the period when $N_{in\_net}$ was significantly greater than zero, and then multiplied by $(k+\lambda)$ to estimate the total volume-normalized particle emissions associated with the event:
\[
\frac{\sigma}{V} = \int \frac{E(t)}{V} \, dt = (k + \lambda) \int N_{\text{in}_\text{net}}(t) \, dt
\] (4.6)

Multiplying equation (4.6) by the volume of the apartment yields an estimate of the total PN emissions associated with the peak event, \(\sigma\). To minimize error in the calculation of \(\sigma\) and \((k + \lambda)\), events were analyzed only if they resulted in a peak \((N_{\text{in}_\text{net}})\) concentration that was at least 1.5 times greater than the simultaneous outdoor PN concentration for at least 5 minutes.

In the monitored apartments, \(N_{\text{in}_\text{net}}\) may be influenced not only by emissions from sources within the apartment itself but also from episodes of particles infiltrating from neighboring apartments. Both types of phenomena are treated as contributions to \(E(t)\) in the material-balance equation. Consequently, for the purpose of this analysis, \(E(t)\) represents emissions from any indoor source event that influences \(N_{\text{in}_\text{net}}\), whether it strictly originates from within the apartment itself or instead reflects infiltration of particles generated in a neighboring apartment.

**Exposure apportionment between indoor and outdoor sources**

For the two sites at which outdoor PN concentrations were available for the full monitoring period (A1 and A3), we apportioned each occupant’s exposure into contributions from outdoor particles and from indoor sources. The first step in the apportionment was determining the fraction of the indoor PN concentration attributable to outdoor particles, using the parameter \(f\). Then, the outdoor-attributable PN concentration was substituted into equation (4.1) to calculate the daily-integrated indoor exposure to particles of outdoor origin. The difference between the daily-integrated indoor exposure attributed to outdoor particles and the calculated total daily-integrated exposure to PN in the home was attributed to exposure to particles generated by indoor sources. The indoor source contribution determined in this manner was checked by means of comparing it to the calculated exposure to indoor generated particles during the indoor source events analyzed at each site. Results produced by these two methods agreed to within 5%, with some difference between them expected, since the decision of which peaks to attribute to indoor sources was made conservatively. Differences in the indoor proportion of outdoor particles during the awake- versus asleep-periods were accounted for by calculating two distinct values of \(f\), based on differences in the apartment ventilation mode corresponding with these periods. At both sites, during the day, at least one window or door in the occupied space was open for most of the time, whereas for most nights, the windows and doors were all closed in the space where the residents slept.

At A3, the residents closed windows and doors throughout the apartment on both nights before sleeping, and the subsequent reduction of indoor PN was recorded by the indoor CPC, which was placed in the living room. Therefore, the estimated overnight \(f\) at this site was based on direct evaluation of the ratio of the indoor and outdoor average PN levels during the hours that the residents slept. At A1, however, we could not use the indoor CPC to determine directly the particle number concentration during the second of two monitored nights. On this night, the residents shut the door and windows and turned on a wall-mounted air conditioner in the bedroom. The indoor CPC was deployed in the living room where the windows were left open overnight. As a result, the direct indoor CPC readings overnight at this site would overestimate the occupants’ exposure. Instead, to estimate the overnight PN exposure of the A1 residents, 12 hours of supplementary overnight data were collected with the indoor CPC placed in the bedroom where the residents slept with the windows and door closed and the air conditioner
operating. The indoor proportion of outdoor particles in the bedroom was then calculated by taking the ratio of the average indoor (bedroom) to outdoor PN concentration on this night; that parameter is referred to as \( f_2 \). Using this parameter, the overnight PN concentration in the bedroom, in the absence of indoor peak events, can be estimated as the product of \( f_2 \) and the outdoor PN level. However, as a further complication, a peak event was detected in the living room while the A1 residents slept in their bedroom on the second monitored night. Consequently, a new parameter representing the ratio of PN levels within the bedroom to those in the living room was estimated using equation (4.7) and is referred to as \( f_{21} \):

\[
f_{21} = f_2 / f = \frac{N_{in2}/N_{out2}}{N_{in}/N_{out}} = \frac{N_{in2}}{N_{in}}
\]

(4.7)

We evaluated \( f_{21} \) as the ratio of \( f_2 \) to \( f \) and then applied \( f_{21} \) to estimate the bedroom PN concentration, \( N_{in2} \), for the second monitored night based on the measured living room concentration, \( N_{in} \). The primary assumption made in this process is that conditions on the night of the supplementary bedroom monitoring were similar to those on the second night of monitoring in the living room, such that the factors influencing \( f \) and \( f_2 \) are comparable. Since the indoor to outdoor temperature difference and the apartment ventilation conditions varied minimally over these days, the approximation seems reasonable.

4.3 Results and discussion

4.3.1 PN time-series and average concentrations

Cumulatively, over the course of 215 h of observational monitoring at the four apartment sites, distinct indoor PN peaks were observed on twenty-seven occasions (Figure 4.3). Each peak was characterized by a sudden increase in the indoor PN level, by a factor of 2-10, followed by a rapid decay back to the baseline. Fourteen of the observed indoor PN peaks were linked to specific source activities using information provided by temperature sensors and the occupant journals. Thirteen of these fourteen peak events were related to food preparation, with the highest peaks occurring at A4, where multiple dishes were often prepared at once. The 3rd peak at A4, which coincided with a report of “house cleaning,” was the only PN peak linked to an activity unrelated to cooking. (No additional details on the nature of the cleaning event are available.) There were no outdoor PN data available during this period at A4, and since this peak caused PN levels to rise only a factor of ~ 3 above the baseline, it is uncertain that it did, in fact, result from cleaning rather than from fluctuations in the indoor contribution from outdoor PN. At A1, ten of the twelve peaks could not be linked to any specific occupant activity. Indeed, two episodes (peak b and an unlabeled peak on July 10) occurred when the occupants were not home, and three others began when they were asleep (peak a, peak f and an unlabeled peak on July 11).

Upon investigation, I came to suspect that these unknown peak events were most likely related to cooking by neighbors. The basis for this inference was the strong smell of cooking that the residents reported entering through their range hood exhaust, which was documented on two occasions to coincide with increases in indoor PN (peaks a and c). It appears that, when in use, the exhaust fan induces cooking emissions to enter a common duct shared by other residents in the building, which was then intended to exhaust to the outdoors. However, when the monitored apartment’s fan was not in use, cooking emissions from other building residents could leak from
the common duct through the range hood and into the apartment. I suspect that many of the eleven unidentified peaks at A1 resulted from this sort of cross-contamination of cooking emissions between apartments. The only other site at which the residents reported being able to smell the neighbors cooking during the monitoring period was A4. The first report was coincident with the second peak at this site, which resulted from an otherwise unknown cause shortly after the residents had completed their own cooking. The second such event at A4 occurred within minutes of when the residents had begun cooking their own dinner, and so was nearly coincident with an identified peak. Since both of these reports were made in close association to the residents’ own cooking, the effect of cooking by the neighbors cannot be easily discerned at A4. Nonetheless, the correlation of indoor PN peaks at A1 and A4 with the report of neighbors’ cooking along with the quantitative findings described in the following sections suggests that this source may have an important influence on indoor PN levels in some Beijing apartments. The International Mechanical Code (IMC), which has been adopted throughout most of the United States, includes provisions to guard against cross-contamination of exhaust air between the rooms of a home and between neighboring apartments. In section 501 of the 2009 version of the IMC, it is stated that, “The air removed by every mechanical exhaust system shall be discharged outdoors at a point where it will not cause a nuisance and not less than the distances specified…The air shall be discharged to a location from which it cannot again be readily drawn in by a ventilating system.” The Chinese government has not adopted the IMC, and instead uses a code developed domestically. According to a translation by the Sustainable Urban Housing in China research group at the Massachusetts Institute of Technology, China’s building code specifies that kitchens are required to have either mechanical or natural means of ventilation, and that residential buildings in cold climates should be equipped with a “ventilation duct” (chinhousing.mit.edu/english/resources/BuildingCode.html). There is, however, no specific instruction regarding the design of the kitchen mechanical exhaust system. If future studies confirm that the exchange of cooking emissions between apartments is common in Chinese high rise buildings, it is suggested that the Ministry of Construction add more detailed instructions regarding the design of kitchen exhaust systems to the Chinese building code.
Figure 4.3. Concentration time-series plots for indoor particle number (PN) concentration at sites A1, A2, A3, and A4 and outdoor PN at A1 and A3. Dotted vertical lines denote midnight. Analyzed indoor source peak events are labeled with lower-case letters and correspond with the peak identifications in Table 4.4. Indoor peak events that were not analyzed are labeled according to the source category responsible (FP = food preparation, Uk = unknown indoor source, Cl = cleaning activity).
In the absence of indoor source events, time variations in the indoor PN concentrations were influenced by fluctuations in outdoor PN levels and by changes in the ventilation conditions of the apartment. Some insight into the effect of outdoor PN fluctuations on the indoor exposure levels can be gained by studying the outdoor PN time-series at A1 and A3. At A1, outdoor levels varied between approximately 10,000 and 50,000 cm$^{-3}$. Levels remained elevated overnight and, on the first day of monitoring, peaked at around midnight. At A3, outdoor levels were affected by precipitation events that included two thunderstorms, one beginning at 23:30 on July 29th lasting for roughly one hour, and one beginning at 23:30 on July 30th ($t \sim 36 \text{ h}$) lasting for roughly 5 hours (http://www.wunderground.com/history/airport/ZBAA), with intermittent rain occurring in between. For the first 1.5 days of monitoring at A3, outdoor levels fluctuated between $10 \times 10^3$ to $40 \times 10^3$ cm$^{-3}$, but, after the onset of the second thunderstorm, levels declined to $3 \times 10^3$ to $30 \times 10^3$ cm$^{-3}$. The persistently elevated PN levels seen overnight at A1 might be related to a 10-year-old regulation enforced by the Beijing Traffic Management Bureau that forbids heavy-duty trucks from entering the city center before 23:00 and after 6:00, with the city center being defined as on or within the 4th ring road (Westerdahl et al., 2009). An exposure implication of elevated nighttime PN levels is that, assuming constant ventilation conditions, a relatively high rate of residential exposure to outdoor generated UFP could occur during the hours that an individual spends asleep, which makes up a substantial proportion of time spent at home.

The parameter $f$, calculated for periods with distinct ventilation modes at A1 and A3, can be used to illustrate the influence of changes in apartment ventilation conditions on the indoor proportion of outdoor particles (Table 4.2). At both A1 and A3, the residents left at least one window open during the hours they were awake, resulting in relatively high $f$ values during these times, 0.78 and 0.65, respectively. Conversely, during hours that the residents slept at A3, the exterior windows and doors were closed throughout the entire apartment, resulting in a lower value, $f = 0.39$. At A1, the windows and doors to the residents’ bedroom were closed on the second night, resulting in an $f$ value of approximately 0.27 in their bedroom. Because of these behaviors, and despite the persistently elevated outdoor PN levels overnight (absent rain), indoor exposure levels were reduced significantly overnight.

Table 4.2. Calculated infiltration factors ($f$) for the three sites at which indoor and outdoor PN concentrations were simultaneously measured.

<table>
<thead>
<tr>
<th>Configuration</th>
<th>A1</th>
<th>A3</th>
<th>A4</th>
</tr>
</thead>
<tbody>
<tr>
<td>Windows open</td>
<td>0.78</td>
<td>0.65</td>
<td>0.76$^b$</td>
</tr>
<tr>
<td>Windows closed $^a$</td>
<td>0.27</td>
<td>0.39</td>
<td></td>
</tr>
</tbody>
</table>

$^a$ For A1, the reported results represent the average infiltration factor for the bedroom, when the windows were closed and the air conditioning was on. For A3, the reported result represents the infiltration factor for the entire apartment, when all windows were closed and there was no air conditioning.

$^b$ This value of $f$ was calculated from a period of 5.6 hours, when the outdoor CPC was available and there were no detectable indoor source activities.

Average PN concentrations for the full duration of monitoring, as well as for periods when residents all were asleep, awake or away, are shown in Table 4.3. The lowest indoor PN
exposure average for all sites occurred during hours that the occupants were asleep, with concentrations ranging from $0.27 \times 10^3$ cm$^{-3}$ at A2 to $24 \times 10^3$ cm$^{-3}$ at A4. At A2, the large reduction in the average PN level when the residents were asleep compared to when they were awake resulted from the use of portable HEPA filter air-cleaning units in each bedroom. Although at least one unit operated for the entire monitoring period, the influence of these units was greatest overnight, when the bedroom doors were closed and the effective volume of treated air was reduced. Apartment A4 was the only site at which overnight PN exposure was not affected by any changes in the apartment conditions other than the absence of indoor peak events.

Conversely, indoor PN concentrations tended to be elevated at all sites during hours that the occupants were awake at home, with averages ranging from $5.4 \times 10^3$ cm$^{-3}$ at A2 to $40 \times 10^3$ cm$^{-3}$ at A1. Higher averages occurred at apartments A1 and A4, primarily owing to the more frequent indoor source events at these apartments. The lowest average occurred at A2, not only because of the operation of HEPA filter units, but also because the residents did not maintain open exterior doors and windows at any time during the monitoring period. The A2 residents expressed at the start of the monitoring that they perceived the outdoor air to be “dirty” and consequently attempted to minimize outdoor-air ventilation, seeking to improve their indoor air quality via filtration. Different viewpoints were held by residents at the other sites, who utilized natural ventilation (an open window or door) for 65% (A3) to 100% (A1 and A4) of the time. Occupants’ decisions about whether or not to keep exterior windows and doors open would clearly influence exposure to ultrafine particles. Maintaining open windows and exterior doors tends to increase the indoor proportion of outdoor particles but to reduce the persistence of particles emitted indoors. The relative contribution of indoor and outdoor generated particles to indoor exposure concentrations and the decay rates of particles generated during indoor peak events is discussed further in §4.3.3-4.3.4.
Table 4.3. Average monitoring data at the 4 apartments, sorted according to occupancy state.

<table>
<thead>
<tr>
<th></th>
<th>A1</th>
<th>A2</th>
<th>A3</th>
<th>A4</th>
<th>Average</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Total Period</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>PNin ($10^3$ cm$^3$)</td>
<td>29.1</td>
<td>2.8</td>
<td>13.0</td>
<td>20.8</td>
<td>16.4</td>
</tr>
<tr>
<td>PNout ($10^3$ cm$^3$)</td>
<td>25.0</td>
<td>-</td>
<td>18.6</td>
<td>-</td>
<td>21.8</td>
</tr>
<tr>
<td>I/O (—)</td>
<td>1.16</td>
<td>-</td>
<td>0.70</td>
<td>-</td>
<td>0.93</td>
</tr>
<tr>
<td>Duration (h)</td>
<td>49.5</td>
<td>47.6</td>
<td>46.8</td>
<td>71.4</td>
<td>53.8</td>
</tr>
<tr>
<td>PM$_{2.5}$ (µg/m$^3$)</td>
<td>63</td>
<td>49</td>
<td>43</td>
<td>54</td>
<td>52</td>
</tr>
<tr>
<td>Tout (°C)</td>
<td>29.1</td>
<td>25.0</td>
<td>1.16</td>
<td>49.5</td>
<td>63</td>
</tr>
<tr>
<td>Tin (°C)</td>
<td>33.1</td>
<td>27.2</td>
<td>29.6</td>
<td>28.6</td>
<td>29.6</td>
</tr>
<tr>
<td><strong>Occupants Awake</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>PNin ($10^3$ cm$^3$)</td>
<td>40.1</td>
<td>5.4</td>
<td>17.3</td>
<td>24.9</td>
<td>21.9</td>
</tr>
<tr>
<td>PNout ($10^3$ cm$^3$)</td>
<td>24.1</td>
<td>-</td>
<td>20.8</td>
<td>-</td>
<td>22.4</td>
</tr>
<tr>
<td>I/O (—)</td>
<td>1.66</td>
<td>-</td>
<td>0.83</td>
<td>-</td>
<td>1.25</td>
</tr>
<tr>
<td>Duration (h)</td>
<td>14.6</td>
<td>13.7</td>
<td>27.0</td>
<td>27.3</td>
<td>20.7</td>
</tr>
<tr>
<td>Tout (°C)</td>
<td>28.3</td>
<td>28.2</td>
<td>27.4</td>
<td>28.7</td>
<td>28.2</td>
</tr>
<tr>
<td>Tin (°C)</td>
<td>32.6</td>
<td>26.6</td>
<td>29.7</td>
<td>28.8</td>
<td>29.4</td>
</tr>
<tr>
<td><strong>Occupants Asleep</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>PNin ($10^3$ cm$^3$)</td>
<td>24.0</td>
<td>0.27</td>
<td>5.5</td>
<td>11.0</td>
<td>10.2</td>
</tr>
<tr>
<td>PNout ($10^3$ cm$^3$)</td>
<td>26.9</td>
<td>-</td>
<td>14.4</td>
<td>-</td>
<td>20.7</td>
</tr>
<tr>
<td>I/O (—)</td>
<td>0.89</td>
<td>-</td>
<td>0.39</td>
<td>-</td>
<td>0.64</td>
</tr>
<tr>
<td>Duration (h)</td>
<td>19.0</td>
<td>13.5</td>
<td>15.5</td>
<td>21.5</td>
<td>17.4</td>
</tr>
<tr>
<td>Tout (°C)</td>
<td>25.7</td>
<td>23.8</td>
<td>22.4</td>
<td>24.2</td>
<td>24.0</td>
</tr>
<tr>
<td>Tin (°C)</td>
<td>33.0</td>
<td>27.5</td>
<td>29.5</td>
<td>28.3</td>
<td>29.6</td>
</tr>
<tr>
<td><strong>Occupants Away</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>PNin ($10^3$ cm$^3$)</td>
<td>24.8</td>
<td>2.7</td>
<td>13.1</td>
<td>-</td>
<td>13.5</td>
</tr>
<tr>
<td>PNout ($10^3$ cm$^3$)</td>
<td>23.5</td>
<td>-</td>
<td>20.3</td>
<td>-</td>
<td>21.9</td>
</tr>
<tr>
<td>I/O (—)</td>
<td>1.06</td>
<td>-</td>
<td>0.65</td>
<td>-</td>
<td>0.85</td>
</tr>
<tr>
<td>Duration (h)</td>
<td>15.4</td>
<td>10.9</td>
<td>3.3</td>
<td>0</td>
<td>7.4</td>
</tr>
<tr>
<td>Tout (°C)</td>
<td>31.9</td>
<td>29.8</td>
<td>22.5</td>
<td>-</td>
<td>28.1</td>
</tr>
<tr>
<td>Tin (°C)</td>
<td>33.6</td>
<td>27.2</td>
<td>29.2</td>
<td>-</td>
<td>30.0</td>
</tr>
</tbody>
</table>

*a The status of A4 occupants who were out of town were not considered in the calculation of PN averages.
*b Outdoor PM$_{2.5}$ data were available for 67% of the observational monitoring period at A1, 100% of the period at A2, 81% of the period at A3, and 85% of the period at A4.
*c Attenuation of A1 residents’ exposure overnight, owing to the closing of windows and doors within their bedroom, is not captured in the indoor average reported here, which represents the average concentration measured in the living room, where windows remained open.
Centrally monitored PM$_{2.5}$ and outdoor temperatures are presented in Table 4.3. The highest PM$_{2.5}$ levels occurred during monitoring at A1, which also corresponded to the period of the highest outdoor temperature. Centrally monitored wind speeds were generally low during monitoring at all sites, with average values of less than 1 m/s for 89%, 94%, 98% and 100% of monitored time at A1-A4, respectively. Weak winds would tend to diminish the distinction between “upwind” and “downwind” orientations from roadways and would also allow more time for vertical transport of particles from roadways to upper floors on the outside of buildings. The apartments were on the windward side of the building for 32%, 21%, 22% and 23% of monitored time at A1-A4, respectively. For all apartments excluding A2, significant natural ventilation would have occurred during monitoring via open windows. Natural ventilation of buildings is driven by temperature differences and by the wind (Linden, 1999). In the case of weak winds, the primary driver of natural ventilation is thermal buoyancy produced by the indoor-outdoor temperature difference. At these sites, the greatest mean temperature difference occurred at A1 during both awake and asleep hours, while the lowest mean occurred at A4 during awake hours and A2 during asleep hours (Table 4.3). Hence, the driving force for natural ventilation would have been the largest at A1 and the weakest at A4. Note, though, that while all the apartments primarily had windows located on a single wall within each room, at A4 the front door was located on the wall opposite the windows in the living room and was often left open. The resulting cross flow would tend to enhance the ventilation rate of this apartment.

### 4.3.2 Daily-integrated PN exposure

The daily-integrated PN exposure while indoors at home of the thirteen residents of the four households ranged from $45 \times 10^3$ cm$^{-3}$ h d$^{-1}$ for resident R2 at apartment A2 to $494 \times 10^3$ cm$^{-3}$ h d$^{-1}$ for resident R6 at apartment A4 (Figure 4.4). Overall, the arithmetic mean and standard deviation (SD) of daily-integrated exposures was $294 \times 10^3$ cm$^{-3}$ h d$^{-1}$ (SD=$161 \times 10^3$ cm$^{-3}$ h d$^{-1}$). The relatively high daily indoor exposure that occurred for three of the residents at A4 can be attributed to two factors. First, the overall indoor PN concentration at this site was the second highest of the four sites, primarily resulting from a high frequency of indoor source events. Second, these three residents spent a greater percentage of their time inside the apartment than did the other study subject at A4. The relatively high exposure rate of the A1 residents resulted from the high number of peak events that corresponded with their period of occupancy, despite the short percentage of time they spent at home. The residents at A2 had the lowest exposure rates, in part because of their use of HEPA filter units.

The percentage of daily exposure occurring while the occupants were asleep ranged from 5% for R1 and R2 at A2 to 38% for R2 at A4 (Figure 4.4). The small proportion of exposure occurring during the sleeping hours at A2 resulted from the high overnight effectiveness of the HEPA filter units. Conversely, the higher proportion of exposure occurring during the sleeping hours at A4 resulted from the higher overnight PN concentration owing to open windows, and to the relatively high proportion of time spent at home and asleep versus at home and awake by some of the A4 residents. In sum, the relative proportion of daily exposure to indoor PN that occurred when residents were asleep versus awake was a function of the relative proportion of time they spent in these states, as well as the degree to which indoor PN was reduced during hours spent asleep.
Figure 4.4. Estimated daily-integrated residential PN exposures of the 13 occupants in the four apartments studied, compared with the average of 21 subjects from 7 homes in the San Francisco Bay Area, CA (Avg CA). The integrated exposure represents the product of the average concentration during occupancy \((10^3 \text{ cm}^{-3})\) and the daily average duration of occupancy \((\text{h d}^{-1})\). The values to the right of each bar report the following parameters: daily-integrated PN exposure during hours at home and awake + daily-integrated PN exposure during hours at home and asleep \((10^3 \text{ cm}^{-3} \text{ h d}^{-1})\); (% of time at home and awake + % of time at home and asleep). The remaining percentage of the resident’s time represents hours away from home. The following gender/age codes are provided for each resident: M/F = male/female greater than 15 years old; m/f = male/female fifteen years old or younger.

4.3.3 Indoor PN peak events

The calculated loss rate coefficients \((k + \lambda)\) of the fifteen analyzed indoor peak events ranged from 1.3 to 13 h\(^{-1}\), with a mean of 5.6 h\(^{-1}\) (SD=3.4 h\(^{-1}\)) (Table 4.4). The calculated total particles emitted \((\sigma)\) for these events ranged from \(11 \times 10^{12}\) to \(88 \times 10^{12}\) particles, with a mean of \(43 \times 10^{12}\) particles (SD=24 \(\times 10^{12}\) particles). The geometric mean (GM) and geometric standard deviation (GSD) calculated for the \(k + \lambda\) distribution were 4.7 h\(^{-1}\) and 1.9; corresponding values for the \(\sigma\) distribution were \(37 \times 10^{12}\) particles and 1.8. All the analyzed peak events are assumed to have resulted from cooking, within either the apartment itself or a neighboring apartment.
Table 4.4. Characterization of emissions and decay for observed peak events.  

<table>
<thead>
<tr>
<th>Site</th>
<th>ID</th>
<th>$k^+$$\lambda$ ($h^{-1}$)</th>
<th>$\sigma$ ($10^{12}$)</th>
<th>$(\sigma/V)/(k^+\lambda)$ ($10^3$ cm$^{-3}$ h)</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>A1</td>
<td>a</td>
<td>2.3</td>
<td>23</td>
<td>59</td>
<td>Unknown</td>
</tr>
<tr>
<td></td>
<td>b</td>
<td>13</td>
<td>19</td>
<td>9</td>
<td>Unknown</td>
</tr>
<tr>
<td></td>
<td>c</td>
<td>11</td>
<td>45</td>
<td>23</td>
<td>Unknown</td>
</tr>
<tr>
<td></td>
<td>d</td>
<td>3.3</td>
<td>35</td>
<td>60</td>
<td>Unknown</td>
</tr>
<tr>
<td></td>
<td>e</td>
<td>2.5</td>
<td>25</td>
<td>58</td>
<td>Unknown</td>
</tr>
<tr>
<td></td>
<td>f</td>
<td>4.7</td>
<td>55</td>
<td>67</td>
<td>Unknown</td>
</tr>
<tr>
<td></td>
<td>g</td>
<td>8.8</td>
<td>88</td>
<td>57</td>
<td>Cooked eggs, bacon and toast</td>
</tr>
<tr>
<td></td>
<td>h</td>
<td>5.7</td>
<td>19</td>
<td>19</td>
<td>Unknown</td>
</tr>
<tr>
<td></td>
<td>i</td>
<td>4.3</td>
<td>55</td>
<td>74</td>
<td>Unknown</td>
</tr>
<tr>
<td></td>
<td>j</td>
<td>4.0</td>
<td>34</td>
<td>50</td>
<td>Fried beef and vegetables; toaster</td>
</tr>
<tr>
<td>A3</td>
<td>k</td>
<td>2.7</td>
<td>29</td>
<td>50</td>
<td>Fried eggs</td>
</tr>
<tr>
<td></td>
<td>l</td>
<td>1.3</td>
<td>11</td>
<td>38</td>
<td>Cooked porridge</td>
</tr>
<tr>
<td>A4</td>
<td>m</td>
<td>8.9</td>
<td>84</td>
<td>43</td>
<td>Fried vegetables</td>
</tr>
<tr>
<td></td>
<td>n</td>
<td>5.6$^b$</td>
<td>66</td>
<td>53</td>
<td>Cooking (type unknown)</td>
</tr>
<tr>
<td></td>
<td>o</td>
<td>5.9</td>
<td>54</td>
<td>42</td>
<td>Fried vegetables. Reheated soup, chicken</td>
</tr>
</tbody>
</table>

$^a$Parameter definitions: $k^+$$\lambda$ = total first-order loss-rate coefficient for particles emitted during the event; $\sigma/V$ = total particle number emitted per house volume; $\sigma$ = total number of particles emitted; $(\sigma/V)/(k^+\lambda)$ = contribution to integrated exposure for an occupant present for the entire event.

$^b$For the first 3 minutes of the peak $n$ decay period, the apparent particle decay rate was 42 h$^{-1}$. For the remaining 15 minutes of decay, the rate decreased to the value reported in the table. It is assumed that the much higher initial decay rate was the result of incomplete mixing.

4.3.4 Exposure apportionment between indoor and outdoor sources

For sites A1 and A3, we have sufficient information to apportion the residential PN exposure experienced by occupants to indoor and outdoor PN sources (see Figure 4.5). Results were similar for residents in each of these two apartments but differed markedly between apartments. The distribution of exposure attributions was approximately 42% owing to indoor sources and 58% owing to particles of outdoor origin for the residents of A1 and 19% (indoor) versus 81% (outdoor) for the residents of A3. On an absolute basis, the residents at A1 and A3 had comparable values of daily-integrated exposure to outdoor-generated PN. However, the residents of A1 had significantly higher integrated exposure to indoor-generated particles than did the residents of A3, primarily because the time they spent at home coincided with a larger number of indoor peak events. Specifically, there were ten indoor peak events during the occupied period at A1, compared to only two at A3. This $5 \times$ difference in the number of indoor peak events within the two apartments resulted in the A1 residents having an approximately $3.4 \times$ greater daily exposure to indoor-generated particles than did the A3 residents. For these two
apartments, the number of indoor source events was an important factor determining the magnitude of the resident’s daily-integrated PN exposure.

**Figure 4.5**: Respective contributions of outdoor and indoor sources to daily-integrated residential PN exposures, on average, for residents at A1, A3 and averaged for 7 homes in the San Francisco Bay Area, California (Avg. CA). The values to the right of each bar represent the daily-integrated PN exposures attributable to particles of outdoor origin + daily-integrated PN exposures attributable to particles of indoor origin, each in units of $10^3$ cm$^3$ h$^{-1}$.

**4.3.5 Comparison with results from the San Francisco Bay Area**

To provide some context for these results, I compare them to those of a recent study in the San Francisco Bay Area of seven single-family homes, where a similar analysis was performed of the exposure of twenty-one residents. In that study, the total monitoring duration was 625 hours (average 3.7 d per site). Cumulatively, 59 peak events were attributed to episodic emissions from indoor sources (Nazaroff et al., 2010; Bhangar et al., 2011).

The average value of daily-integrated residential PN exposure calculated for these Bay Area residents ranged from $70 \times 10^3$ to $726 \times 10^3$ cm$^3$ h$^{-1}$ with a mean of $297 \times 10^3$ cm$^3$ h$^{-1}$ (SD=$195 \times 10^3$ cm$^3$ h$^{-1}$). Remarkably, these results are similar to those calculated in the present study for the thirteen residents of the four Beijing apartments. The mean daily-integrated exposure of the two study samples differ by less than 2%, with the individual rates in both cases spanning a range of roughly 10x. The distribution of the average daily-integrated exposure between hours spent asleep and awake was also comparable, with 75% and 80% occurring when residents were awake in the Beijing and Bay Area study, respectively. Consequently, for these two admittedly small samples of individuals, the average daily-integrated PN exposure that occurred at home and the distribution of this exposure between two activity states are similar.

In the Bay Area study, particle decay rates and source strengths were characterized for approximately 20 events that involved cooking on a gas stove. The $k + \lambda$ values for these events had a GM of 1.8 h$^{-1}$ and a GSD of 1.4. The higher GM ($\sim 3x$) for the $k + \lambda$ associated with peak events in the Beijing apartments might be attributable to the greater utilization of natural ventilation at these sites compared to the Bay Area homes. At six of the seven homes studied in
the Bay Area, external windows and doors were opened for less than 20% of the monitored time, in contrast to the 65-100% of time that windows were open at A1, A3 and A4. The emission source strength of the gas cooking episodes in the Bay Area study had a GM of $38 \times 10^{12}$ particles per event and a GSD of 2.1, similar to the results found in the present study in Beijing apartments (GM = $37 \times 10^{12}$ particles per event and GSD = 1.8).

On average, residential PN exposure was ~ 70% attributable to indoor sources in the Bay Area homes, much higher than in the two apartments in Beijing (A1 and A3) for which an assessment could be made. I can suggest three reasons for this difference. First, the outdoor PN concentration at A1 and A3, particularly during the asleep hours, was considerably higher than was observed on average in the Bay Area homes. Second, the residents of A3 spent a higher proportion of their time at home than did the Bay Area residents. Third, and perhaps most importantly, natural ventilation was more frequently utilized at A1 and A3 than observed on average in the Bay Area homes. As a result, although the residents of A1 and A3 had overall daily-integrated PN exposures in their homes that were comparable to those of the Bay Area study residents, the relative contributions of indoor versus outdoor sources to the respective exposures were markedly different.

4.4. Conclusion

Of the four apartments studied in Beijing, those where residents most frequently utilized natural ventilation had higher baseline PN concentrations, but also experienced more rapid decay of indoor-generated PN peaks. Overall, the apartments with the highest number of indoor peak events had highest indoor PN concentrations. The daily-integrated PN exposure while at home calculated for the residents of these four apartments were comparable with those previously determined for residents of seven San Francisco Bay Area homes. However, in the cases studied, a larger fraction of exposure appears to be made up of outdoor-generated PN for the Beijing apartments. The analyzed peak events in the Beijing apartments had particle emissions comparable to those found for natural-gas cooking events in the Bay Area homes; however, the average decay rate of indoor generated particles was higher in Beijing apartments, probably owing to the greater use of natural ventilation.

This study represents a start in characterizing the levels, sources, dynamic behavior, and indoor exposures to ultrafine particles in Beijing residences. Our study was limited to a small number of apartments investigated during just one season. Further investigation of indoor UFP exposure concentrations and influencing factors in urban Chinese homes over all seasons is necessary if we are to more completely understand the nature of this important aspect of air pollution.
Chapter 5. Conclusion

Ultrafine particles are ubiquitous in the environment. In addition to natural sources, there are numerous anthropogenic sources that result in elevated outdoor concentrations in populated areas, relative to the rural background. Indoor concentrations of ultrafine particles are influenced by both indoor and outdoor sources. Examples of important indoor sources include cooking, cleaning with terpene scented products and indoor combustion activities; outdoor sources include motor vehicles and atmospheric nucleation. Human exposure to ultrafine particles is a topic that has recently received increased attention from the research community. Interest arises partly because recent health studies have indicated that UFP exposures may lead to adverse health effects and partly because advances in instrumentation have made monitoring UFP concentrations in microenvironments more convenient and affordable. That said, there are still numerous geographic regions and microenvironments within which human exposure to ultrafine particles remains poorly characterized.

This dissertation contributes towards understanding human exposure to ultrafine particles. It makes its contribution by providing and synthesizing new data for two categories of microenvironments within which exposure concentrations had hitherto not been studied: elementary school classrooms in the San Francisco Bay Area and high-rise apartments in Beijing, China.

Children’s exposure to air pollution is a topic of special interest because children are generally considered to be at greater risk of adverse health effects resulting from pollutant exposures than are adults (Schwartz, 2004). The classroom microenvironment is of particular interest in this context because children spend a significant proportion of their time at school. Since UFP concentrations in classrooms in the San Francisco Bay Area had yet to be investigated, the aim of Chapter 2 was to provide new information regarding ultrafine particle exposures of students attending schools in the Bay Area. Specifically, average daily-integrated exposures of students to ultrafine particles were quantified, the relative importance of indoor and outdoor sources to those exposures were assessed, the parameters most significantly influencing exposure concentrations were explored, and the potential of using outdoor concentrations as an indicator of indoor exposures was investigated.

The classroom air-exchange rate is an important parameter influencing the exposure that students encounter to outdoor-generated particles in their classrooms. In Chapter 3, a more detailed assessment was presented of air-exchange rates in the sample of classrooms studied in Chapter 2. Furthermore, Chapter 3 provided an exploration of what classroom factors affected whether the air-exchange rate was high or low, how the classroom ventilation characteristics affected the levels of both outdoor-generated particles and indoor-generated bioeffluents (as indicated by metabolic CO2 levels), and how particle levels in the classrooms that originate outdoors might be reduced without reducing the air-exchange rate. While these findings are specific to the investigated classrooms, the contributions to identifying and characterizing mechanistic relationships can help establish a more complete understanding of the UFP exposure levels encountered by school children, the factors that influence those exposures, and the parameters that can be controlled to limit them.

Relatively little is known regarding human exposure to ultrafine particles in mainland China, despite the fact that this country is home to roughly 20% of the world’s
population. Rapid economic growth over the last few decades in China has led to significant changes in the housing stock and very probably to characteristics of both indoor and outdoor pollutant sources, all of which are expected to influence indoor pollutant exposures. However, there have been relatively few studies investigating indoor air quality in Chinese homes, and no prior work investigated residential concentrations of ultrafine particles. To date, investigations of ultrafine particles in mainland China published in English-language journals have been conducted exclusively in the outdoor environment, and even then, there are few published studies relative to the number reporting investigations in the United States and Europe. To contribute information and knowledge to help fill this gap, results were presented in Chapter 4 on particle concentrations and daily-integrated particle exposures of the residents in four Beijing high-rise apartments. In addition, the distribution of the resident’s exposure between awake and asleep periods and the apportionment between indoor and outdoor sources was discussed, observed indoor particle source events were characterized, and the influence of the apartment ventilation mode on indoor particle concentrations was explored. These results represent an initial yet substantial step towards characterizing the UFP exposure of China’s urban population.

In the sections to follow, results and discussion from Chapters 2 through 4 are summarized, and then possible topics for future research are discussed.

5.1 Summary of results: UFP exposure in six San Francisco Bay Area classrooms

From June 2nd to December 6th 2008, pollutant concentrations and occupant activities were monitored for 70-95 h in each of six classrooms in four different schools in the San Francisco Bay Area. The classrooms were all located in Alameda County and were situated within 15 km of each other. The monitored pollutants included particle number (PN), CO₂, NO and O₃ concentrations and instrument packages were deployed both in a classroom and outside, on the school grounds. The monitored PN concentration of particles having a diameter of 6 nm or greater was used as a surrogate measure for ultrafine particles, since ultrafine particles have been found to make up 90% or more of PN concentrations in urban areas in the US. A researcher was present in the classroom for the duration of student occupancy to document the number of occupants present, the activities taking place, and the status of window and door openings and HVAC operation. All data were collected with 1-minute time resolution.

In Chapter 2, results for different aspects of student UFP exposure in the 6 classrooms were reported. The average indoor PN concentration during periods of student occupancy ranged from $5 \times 10^3$ to $16 \times 10^3$ cm⁻³. Indoor sources had a relatively small influence on classroom PN concentrations. In all, considering 18 school days monitored cumulatively at the six sites, only three indoor source events were observed during periods of student occupancy: cooking of pancakes on an electric stove for a student presentation, the use of a candle to celebrate a child’s birthday, and use of a central heating system. The particles emitted during these events contributed 26%, 8% and 38%, respectively, to the cumulative PN exposure experienced by students in their classroom on the respective days of occurrence.

For the four classrooms monitored from June to early November, peak outdoor PN concentrations occurred mid-day, corresponding with periods of student occupancy.
For the two classrooms monitored from late November to early December the outdoor PN concentration profile either had peaks occurring in the morning and evening hours, often when students were not at school, or lacked distinct peaks. This feature, potentially of seasonal origin, contributed to the classrooms monitored in warmer months having both a higher outdoor and a higher indoor average PN concentration during periods of student occupancy. The higher exposure to outdoor generated particles during warm months was exacerbated by more frequent opening of doors and windows for the purpose of maintaining a comfortable temperature in the classroom, which served to increase the indoor proportion of outdoor particles.

Considering all eighteen school days monitored, the average calculated daily-integrated exposure per student in their classrooms ranged from $11 \times 10^3$ cm$^{-3}$ h d$^{-1}$ to $100 \times 10^3$ cm$^{-3}$ h d$^{-1}$, with an overall mean of $50 \times 10^3$ cm$^{-3}$ h d$^{-1}$ and a relative standard deviation of 46%. To put these exposures into context, corresponding average daily-integrated PN exposures experienced at home by seven children subjects in a parallel study of seven San Francisco Bay Area single-family dwellings was ~ 6x higher, at a level of $320 \times 10^3$ cm$^{-3}$ h d$^{-1}$ (Nazaroff et al., 2010). The higher daily-integrated exposure experienced by children in homes is partly attributable to the higher PN concentrations measured in homes during hours of occupancy than in schools, and partly a result of the greater time that children spend in their home on a daily basis as compared to their classrooms.

In the case of these classrooms, regressing the average student daily-integrated PN exposure for each of the 18 days of monitoring against the outdoor 24-hour PN average for those same days produced an $R^2$ of 0.75. Regressing the same exposure results against the daily outdoor PN averages calculated only for periods of student occupancy produced an $R^2$ of 0.88. Thus, in the case of these classrooms, outdoor PN concentrations measured on-site appear to be a good indicator of the relative exposure concentrations encountered by students in their classrooms. However, the utility of outdoor data for predicting exposures indoors depends critically on the dominance of outdoor air as the source of indoor PN levels.

The CO$_2$, PN, occupancy and ventilation mode time-series data collected at each site were used to assess whether the classrooms were adequately ventilated for the removal of bioeffluents, and whether an increase in the classroom air-exchange rate was accompanied with an increase in the indoor proportion of outdoor particles (IPOP). The results of this analysis were presented and discussed in Chapter 3. The question of whether bioeffluents were being adequately removed from the classroom was assessed by calculating the classrooms’ indoor minus outdoor CO$_2$ (dCO$_2$) level and the rate of ventilation per person, and then comparing results from the six classrooms to each other and to a standard established by the American Society of Heating Refrigerating and Air-Conditioning Engineers (ASHRAE). The geometric mean (GM) dCO$_2$ level calculated for the period of student occupancy at each of the sites ranged from 130 ppm to 518 ppm. The time-weighted average ventilation rate per person, calculated from segments of the period of student occupancy at each site, ranged from 4 to 27 L/s. The classroom with the highest dCO$_2$ and the lowest rate of ventilation per person was mechanically ventilated, but had a low supply rate of outdoor air. In addition, this classroom was monitored in late November, and—owing to cool outdoor temperatures—doors were infrequently open. The classroom with the lowest dCO$_2$ and highest rate of ventilation
per person was also mechanically ventilated, but with a high supply rate of outdoor air. The remaining four classrooms were all naturally ventilated and exhibited intermediate dCO$_2$ levels in central tendency. The time-weighted average air-exchange rate for the six classrooms ranged from 1.1 to 10.8 h$^{-1}$. The highest air-exchange rate was calculated for the mechanically ventilated classroom with the lowest GM dCO$_2$ concentration, and the lowest air-exchange rate was calculated for a naturally ventilated classroom monitored in December that had doors and windows closed for 80% of the occupied period. The classroom with the lowest air-exchange rate did not have the lowest rate of ventilation per person, as might be expected, owing to it having the lowest average number of students per unit volume (0.05 students/m$^3$) of all the sites. The time-weighted average IPOP across sites varied from 0.30 to 0.67. The lowest IPOP occurred at the mechanically ventilated site with the low ventilation rate, and the highest IPOP occurred at the mechanically ventilated site with the high ventilation rate. Consequently, although the high air-exchange rate in the latter classroom resulted in it having the lowest dCO$_2$ level, it also resulted in it having the highest indoor proportion of outdoor generated particles among the six classrooms studied.

Results from five sites were analyzed to discuss whether an increase in the air-exchange rate was accompanied by an increase in the IPOP. For four of the classrooms, the data were so correlated. However, reducing the air-exchange rate as a strategy for decreasing the indoor level of outdoor generated particles is not recommended, since a lower air-exchange rate would result in a higher concentration of indoor generated pollutants. Instead, strategies were investigated for reducing the IPOP without reducing the air-exchange rate. In the case of the naturally ventilated classrooms, a modeling analysis indicated that operating a commercially available fan-filter air-cleaning unit within the classroom could reduce the PN level by a substantial amount for an additional cost of only $4.50 per student per year, assuming the air-cleaner is used for 10 years. In the case of the mechanically ventilated classrooms, a modeling analysis indicated that installing a moderate-efficiency filter (MERV 12) in the HVAC duct and replacing it three times a year would cost only $5.25 per student per year. Analysis indicates that the implementation of these filtration measures would cost $1 to $2 for every $10^{12}$ fewer particles inhaled by a student in the classrooms.

5.2 Future research: UFP exposure in elementary school classrooms

This work represents an initial investigation of UFP exposure concentrations in classrooms in the San Francisco Bay Area, and one of the first published studies of the relationship between the air-exchange rate of any building type and the indoor concentration of ultrafine particles. This work was based on data collected at only a small sample of sites in one location for a limited portion of the year, and—like all research—the results are subject to some error. Consequently, to the extent that children’s exposure to ultrafine particles is deemed an important issue, this work should be supplemented with further study of UFP exposure concentrations in additional elementary school classrooms.

Besides a general study of classroom UFP concentrations, there are a few specific issues that warrant further investigation. First, the observation made in the present study, as well as in a few studies conducted in Europe, that outdoor UFP sources are the primary contributor to classroom concentrations, should be investigated in a larger sample of
classrooms. If this observation is proven to be generally true, then it may be practiced to use outdoor PN data to predict students’ UFP exposures while in their classrooms. Secondly, the present study suggests that in both mechanically and naturally ventilated classrooms, measures may need to be taken to ensure that classrooms have an adequate air-exchange rate. Specifically, it was found in the naturally ventilated classrooms that when opening doors and/or windows resulted in thermal discomfort, the teacher tended to leave the classroom in a closed state, which led to elevated concentrations of CO₂. Further investigation is needed to see if teachers would be willing to leave the classroom “open” if arrangements are made for providing supplemental temperature control when necessary. In addition, it would be worthwhile to investigate whether supplying teachers with a classroom CO₂ monitor, and thus providing them with knowledge of when the classroom was receiving insufficient ventilation, would influence the frequency of door and window openings. In the case of one of the mechanically ventilated classroom, a low air-exchange rate was suspected to have resulted from the HVAC damper being insufficiently open. Future studies should investigate whether this issue is one that occurs frequently in mechanically ventilated classrooms, and, if so, whether an increase in either the training or the staffing of building maintenance personnel would decrease the likelihood of its occurrence. Finally, in the present study, strategies were suggested for reducing the IPOP for ultrafine particles in both naturally and mechanically ventilated classrooms. Further research is needed to investigate whether implementation of these strategies accomplishes the desired aim, or whether they prove ineffective because, in the former case, the noise induced by the fan-filter air cleaners is found bothersome by the teachers and students or, in the latter case, the school has insufficient staffing to provide the maintenance required for achieving good filtration performance. In addition, the cost and benefit of strategies for reducing children’s UFP exposure in their classrooms should be compared to the cost and benefit of strategies to reduce exposures in other microenvironments where children spend their time, so that informed decisions can be made regarding the most effective means by which children’s UFP exposure can be improved.

5.3 Summary of results: UFP exposure in four Beijing high-rise apartments

From June to August 2009, PN concentrations were measured in four high-rise apartments in Beijing, China, for two to four days each during normal occupancy and use. As was the case with the study in San Francisco Bay Area schools, PN concentrations were used as a surrogate measure for UFP concentrations. During the measurement period, residents were asked to maintain a diary of the hours they were present in the apartment and the activities they conducted while at home. In addition, temperature sensors were used to document occupant activities that involved heat generation (e.g., cooking), and state-change sensors were used to monitor the positions of doors and windows. All data were collected with 1-minute resolution.

The overall average PN concentration, during hours that at least one resident was home, ranged from $2.8 \times 10^3$ to $29.1 \times 10^3$ cm$^{-3}$. The PN concentration during hours that all residents were asleep ranged from $0.27 \times 10^3$ to $24.0 \times 10^3$ cm$^{-3}$, and during hours that all residents were awake levels ranged from $5.4 \times 10^3$ to $40.1 \times 10^3$ cm$^{-3}$. One apartment had the lowest PN concentration for all three of these reported ranges. At this apartment, windows were never opened and two portable HEPA filter air-cleaners were utilized for
the majority of time. The residents of this apartment were the only ones among the study subjects in Beijing that perceived the outdoor air to be “dirty,” and therefore they took steps to prevent intrusion of outdoor air pollutants and to remove indoor particles. At the other three apartments, windows were left open for the majority of the monitored periods. The apartment with the highest PN concentrations also had the highest number of PN peaks resulting from indoor source events. Although the residents of this apartment cooked infrequently themselves, emissions appeared to infiltrate from neighboring apartments, where cooking occurred frequently.

In aggregate for the four sites, distinct indoor PN peak events (i.e., peaks in the indoor PN concentration resulting from indoor emissions and clearly distinct from changes in the outdoor concentration) occurred on twenty-seven occasions. Fourteen of these peak events were linked to specific activities, which included thirteen food preparation activities and one cleaning activity. Fifteen of the indoor peak events were analyzed, and were found to have a GM source strength of $43 \times 10^{12}$ particles emitted per event and a GM first-order particle decay rate coefficient of 5.6 h$^{-1}$.

The daily-integrated PN exposure during hours spent at home for the thirteen residents of the four households ranged from $45 \times 10^3$ to $494 \times 10^3$ cm$^{-3}$ h d$^{-1}$, with an arithmetic mean of $294 \times 10^3$ cm$^{-3}$ h d$^{-1}$. The highest daily-integrated exposures occurred at residences where there were a high number of indoor source events, a high indoor proportion of outdoor particles, and where the residents were at home for a relatively large proportion of the day. The percentage of daily residential exposure occurring while the residents were asleep ranged from 5% to 38%, with the lowest proportion occurring in the apartment that had fan-filter air-cleaners operating overnight in the closed bedrooms, and the highest occurring in an apartment where windows remained open overnight. At two of the apartments, outdoor PN concentrations were measured for the duration of monitoring. For these apartments, the calculated apportionment of the residents’ exposure attributable to indoor and outdoor generated particles was 42% (indoor) and 58% (outdoor) at one site, and 19% (indoor) and 81% (outdoor) at the other site. The much higher fraction of exposure attributable to indoor-generated particles at the first site was due to the higher number of indoor source events that occurred there.

These results were compared to those of a study of seven single-family homes in the San Francisco Bay Area. The GM source strength of natural-gas cooking events at the homes in the Bay Area was only 20% less than that of the analyzed peak events in the Beijing study; however, the GM particle decay rate from the natural-gas cooking events was ~3x higher in Beijing than in northern California. This latter result was assumed to have largely occurred because of the greater use of natural ventilation in the Beijing apartments compared to the Bay Area homes. Remarkably, the average value of daily-integrated residential PN exposure calculated for the Bay Area residents was nearly the same as that of the Beijing residents. However, an average 70% of the Bay Area residents’ exposure was attributable to particles of indoor origin, considerably greater than the result for the two apartments for which this percentage was calculated in Beijing. The higher contribution of outdoor particles to indoor exposures in the Beijing apartments was suggested to have primarily occurred because of the greater amount of natural ventilation used there, which is expected to result in a more rapid decay rate of indoor generated particles and in a higher indoor proportion of outdoor particles.
5.4 Future research: UFP exposure in Beijing residences

To date, few studies of ultrafine particle concentrations have been conducted in mainland China, and those that have been conducted have focused on the outdoor environment. As a result, there is no way in which to characterize the personal exposure to ultrafine particles of the 1.2 billion people living in China, or to understand which sources primarily contribute to these exposures or in which microenvironments they primarily occur. To the extent that the research community is able to collect data characterizing ultrafine particle exposures of different populations living in China, national and international policy makers and development groups will be better able to assess whether there is a need to reduce UFP exposures in China, and, if so, what strategies might be used to accomplish those reductions. Therefore, it is important that the recent emergence of research on UFP concentrations outdoors in China be accompanied by studies investigating concentrations indoors, so that population exposures can be more accurately characterized. Since it has been shown that people generally spend a majority of their time in their home, it is logical that initial studies of indoor exposures would include a major focus on the residential environment. In the paragraphs to follow, I describe research opportunities that exist specifically within Beijing residential environments.

The present work represents an initial investigation of PN concentrations in Beijing residences. Beijing is not only the capital city of China, it is also one of the most populous urban areas in the country. Common types of housing in Beijing vary from courtyard style houses converted into multiple units, to both new and old low-rise and high-rise apartment buildings, to newly constructed single-family homes in the suburbs. The indoor UFP sources predominantly influencing exposure concentrations in these various types of Beijing residences are likely to differ, since indoor fuels in the city vary from coal to liquid petroleum gas to natural gas, and cross-contamination between residences is likely to occur in some types of housing structures more so than in others. The outdoor UFP sources influencing residences are also expected to differ depending on the residence height above ground and proximity to local traffic and industry. Additionally, newer buildings are likely to be more airtight, which would affect both the IPOP and the decay rate of indoor generated particles. Therefore, UFP exposure concentrations should be investigated in a wide range of Beijing residential structures, so that an understanding can be built of the factors that govern microenvironmental UFP levels and exposures across the spectrum of Beijing residences, versus assuming the findings from one type of housing are representative of other types.

The present work was focused on ultrafine particle concentrations in high-rise apartments in Beijing. High-rise buildings have become a common and popular style of design for new housing developments in China; however, prior to the present work, there had yet to be any studies systematically investigating UFP exposure concentrations in this type of residential structure in China. Therefore, it is important that the results from this study be built upon by further study of Beijing high-rise apartments, with attention given to the following questions that arose out of this study. Is the infiltration of cooking emissions from neighboring apartments common in Beijing high-rise buildings, and, if so, what can be done to prevent this from occurring? Does enhanced natural ventilation in Beijing high-rise apartments have an overall beneficial or detrimental affect on UFP
exposure concentrations? To what extent are portable HEPA filter air-cleaning units an effective tool for reducing Beijing residential UFP exposures?

Another issue that should be explored is how the contribution of indoor and outdoor-generated particles to indoor exposure varies with season. Specifically, the present work indicated that outdoor-generated particles on average made a more significant contribution to residential exposures in the Beijing apartments than was observed in San Francisco Bay Area homes; however, this result for Beijing only was obtained during the summer months, when windows were generally left open for the majority of time. An issue that remains to be addressed is how the relative contribution of indoor and outdoor sources to residential concentrations would change in the winter, when windows are open less commonly. Since some studies have indicated that the health effects elicited by outdoor generated particles may be different from those generated indoors, and since strategies for reducing residential exposures will vary depending on whether UFP concentration are primarily of indoor or outdoor origin, understanding the distribution of exposures between indoor and outdoor generated particles on an annual basis is of particular importance towards characterization of Beijing residents’ UFP exposure.

An important issue that was not investigated in the present study is the extent to which an apartment’s elevation above ground influences the indoor concentration of outdoor generated particles. To date, there have been only a few studies investigating particle number concentrations at different elevations along the outside of a building envelope (Hitchens et al. 2002; Li et al., 2007; Kumar et al., 2009); however, no studies have investigated the vertical variation of particle concentrations inside a building. Since the indoor concentration is a function of not only the outdoor concentration, but also the air-exchange rate, which is predicted to vary with pressure changes along the vertical axis of a building envelope, the indoor and outdoor vertical distribution of particle concentrations is likely to differ, and therefore both should be investigated. An understanding of the indoor concentration of outdoor generated particles in apartments at varying heights above ground would provide insight into the question of whether residents of lower floors are at greater risk of elevated exposure to motor vehicle emissions than are residents of upper floors.

Besides these specific issues, UFP concentration data should be collected in a larger sample of Beijing high-rise apartments so that population exposures can be more accurately characterized and systematic differences in exposure conditions in buildings of different age, different construction and in different areas of the city can be discerned. Ultimately, understanding Beijing UFP concentrations in both indoor and outdoor microenvironments will allow for the time-integrated exposures of Beijing residents to be more accurately characterized, and this will allow for results from Beijing to be put into context with those from other regions of the world. Similarly, understanding the factors leading to differences in exposure within the various types of Beijing residences will allow for discernment of whether population exposures can be practically reduced or limited by modifying certain building design practices or occupant behaviors.
5.5 Closing remarks: future of ultrafine particle exposure research

For the past two decades, there has been regular and frequent reporting of research studies providing evidence of adverse health effects resulting from exposure to ultrafine particles, as well as information regarding ultrafine particle concentrations within different microenvironments where people spend their time. However, despite accumulated scientific research regarding ultrafine particle concentrations, exposures and toxicology, no effort has been made as yet, by the majority of governmental agencies around the world, to either regulate concentrations or to control source emissions. This inaction is not the result of a decision that no action is needed, but is rather the result of a lack of consensus regarding whether or not existing UFP exposure concentrations pose a risk to public health. Consequently, more research is needed to fill the information gap that currently stands in the way of reaching a consensus regarding whether population exposures to ultrafine particles should be reduced, and if so, how those reductions might best be realized.

In a document published by the World Health Organization on air-quality guidelines for particulate matter, O₃, NO₂ and SO₂ (WHO, 2006), it is stated that, “While there is considerable toxicological evidence on potential detrimental effects of ultrafine particles on human health, the existing body of epidemiological evidence is insufficient to reach a conclusion on the exposure-response relationship of ultrafine particles. Therefore, no recommendations can be provided as to guideline concentrations of ultrafine particles at this point in time.” Since introducing ambient air quality regulations can have economic implications, there is generally a hesitancy to do so until compelling evidence exists of health effects at environmentally relevant concentrations. This same reluctance was observed when the US Environmental Protection Agency first considered introducing fine particulate matter into the US National Ambient Air Quality Standard (NAAQS). It was not until a lawsuit by the American Lung Association and a comprehensive review of scientific literature, that, in 1997, the US EPA added a PM₂.₅ standard to the NAAQS, only to be met by further legal challenges (Pope and Dockery, 2006). It was not until 2002 that the US Court of Appeals ruled that the PM₂.₅ standard was not “arbitrary or capricious” (Pope and Dockery, 2006). Consequently, before policy decisions are likely to be made regarding regulation of UFP concentrations, more research is needed from the public health and medical research communities regarding adverse health effects associated with ultrafine particle exposures. Some questions of particular interest include the following. Are there threshold exposure levels below which no health effects would occur, and if so, what are they? What is the dose-response relationship for population exposures to ultrafine particles? What are the short- and long-term health effects associated with exposures? How do health effects from UFP exposures depend on specific physical and chemical properties of the particles?

Designing epidemiological studies that accurately characterize population exposure to ultrafine particles will require innovation on the part of research groups. Specifically, the majority of studies investigating associations of PM₂.₅ exposure with health outcomes, including those that are most widely cited (Dockery et al., 1993; Pope et al., 1995; Pope et al., 2002), have used air pollution data from central monitors located at regional sites to characterize population exposures. While using central monitoring data is convenient, studies have suggested that this approach might not be appropriate for characterizing exposure to ultrafine particles, due to the high spatial variability in UFP
concentrations (Pekkanen and Kulmala, 2004; Sioutas et al., 2005; Hudda et al., 2010). Thus, epidemiologic research groups have the challenge of designing cost effective methods for approximating the personal exposure to ultrafine particles of study subjects, in order to avoid exposure misclassification.

If consensus is reached that existing UFP concentrations pose risks to public health, then the next question becomes, how can population exposures be most effectively reduced? One approach to answering this question is to break it into two parts. First, can exposures be effectively reduced via source controls, and if so, which sources should be controlled and using what approaches? Second, can exposures be effectively reduced by decreasing microenvironmental concentrations, and if so, in which microenvironments and using what strategies? These questions can be best answered if estimates are available regarding time-integrated population exposures to ultrafine particles and the apportionment of those exposures both among the microenvironments where people spend their time and among the contributing UFP sources. In addition, the design of control strategies will require that characteristics of both contributing sources and microenvironments be well understood. At present, data are available on UFP concentrations for several types of environments and on UFP emissions from a variety of sources. However, there remains much about UFP sources and concentrations that is not well understood. For example, studies investigating outdoor concentrations have been primarily conducted in North America and Europe; therefore, little can be said about the exposures to outdoor UFP concentrations encountered by the majority of people in the world. Even in North America and Europe, atmospheric nucleation, which is considered an important outdoor source of ultrafine particles, has not been fully characterized. Even less is known about UFP concentrations encountered indoors, despite the fact that people typically spend a majority of their time indoors. Studies conducted thus far in residential environments have focused on single-family dwellings in only a few geographic regions in the world. Fewer studies have been conducted in indoor environments other than homes. The information gaps summarized here represent only a few of the many that exist in the vast landscape of UFP exposure research. Thus, before population exposures to ultrafine particles can be accurately characterized, before the factors influencing those exposures can be well understood, and before decisions regarding strategies for reducing exposure concentrations or controlling source emissions can be made, these knowledge gaps must be filled.
References


densities of traffic-related aerosol particles at a busy highway in Helsinki. *Atmospheric Chemistry and Physics* 6, 2411-2421.


particle number size distributions and chemical compositions at the urban and downwind regional sites in the Pearl River Delta during summertime pollution episodes. *Atmospheric Chemistry and Physics* **10**, 9431-9439.


Appendix A: Floor plans of the six investigated San Francisco Bay Area classrooms

This appendix presents floor plans for the six investigated classrooms.
Figure A.1. Floor plan for classroom S1. “D” designates a doorway. “Bag Storage” refers to cubbyholes for the students’ school bags.
Figure A.2. Floor plan for classroom S2. “D” designates a doorway. “Bag Storage” refers to cubbyholes for the students’ school bags. Enclosure 2 was located outdoors, roughly 25 m northeast of the classroom doorway. Enclosure 3 was located in the school cafeteria.
Figure A.3. Floor plan for the building containing S3. The building contains three classrooms total, as well as a middle room connecting the three (“Pod” room). S3 is designated the “Main Classroom”. Enclosure 2 was located outdoors, roughly 150 m east of the classroom. Enclosure 3 was located in the school cafeteria.
Figure A.4. Floor plan for classroom S4.
Figure A.5. Floor plan for classroom S5. Enclosure 2 was located outdoors, roughly 10 m east of the classroom. Enclosure 3 was located in an interior hallway in another section of the school.
Figure A.6. Floor plan for classroom S6. Enclosure 2 was located outdoors, roughly 150 m east of the classroom. Enclosure 3 was located in the school cafeteria.
Appendix B. Calibration parameters calculated for pollutants measured at S1-S6.

To the extent practical, instrument calibrations were conducted throughout the period of monitoring at S1-S6. The approach is described in §2.2.2. Calibration results are presented in Tables B.1 (particles), B.2 (carbon dioxide), B.3 (ozone), and B.4 (nitric oxide). The results presented for the gaseous pollutant monitors were determined from laboratory span gas checks. The results presented for particles were determined from side-by-side monitoring of the three instruments.

Table B.1. Slopes used to adjust PN data collected by the outdoor (QMEd) and supplementary indoor (QMEc) particle monitors. a

<table>
<thead>
<tr>
<th>Site</th>
<th>QMEc</th>
<th>QMEd</th>
</tr>
</thead>
<tbody>
<tr>
<td>S1</td>
<td>—</td>
<td>1.03</td>
</tr>
<tr>
<td>S2</td>
<td>1.16</td>
<td>1.19</td>
</tr>
<tr>
<td>S3</td>
<td>1.08</td>
<td>0.99</td>
</tr>
<tr>
<td>S4</td>
<td>0.92</td>
<td>0.89</td>
</tr>
<tr>
<td>S5</td>
<td>1.12</td>
<td>0.82</td>
</tr>
<tr>
<td>S6</td>
<td>1.08</td>
<td>1.06</td>
</tr>
<tr>
<td>Average</td>
<td>1.07</td>
<td>1.03</td>
</tr>
<tr>
<td>St. Dev.</td>
<td>0.09</td>
<td>0.11</td>
</tr>
</tbody>
</table>

a The primary indoor CPC (QMEa) was used as the transfer standard, and slopes represent a regression of data collected from QMEc or QMEd against data collected side-by-side from QMEa.
b At S2-S6, QMEa was indoors, QMEc was outdoors and QMEd was used as a supplementary indoor monitor. At S1, QMEc was indoors, QMEa was outdoors, and QMEd was a supplementary indoor monitor.
c QMEc was not available for side-by-side monitoring at S1.
d Instrument nozzles were replaced prior to work at S5.

Table B.2. Calibration parameters for CO2 monitors.

<table>
<thead>
<tr>
<th>Date</th>
<th>LI-COR 820 a</th>
<th>Q-Trak (T85a) a</th>
<th>Q-Trak (T85b) a</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Slope</td>
<td>Offset (ppm)</td>
<td>Slope</td>
</tr>
<tr>
<td>May 2008</td>
<td>1.03</td>
<td>-5.1</td>
<td>0.94</td>
</tr>
<tr>
<td>June 2008</td>
<td>1.06</td>
<td>3.5</td>
<td>0.92</td>
</tr>
<tr>
<td>3 July 2008</td>
<td>—</td>
<td>—</td>
<td>0.90</td>
</tr>
<tr>
<td>Sept 2008</td>
<td>1.00</td>
<td>0.2</td>
<td>—</td>
</tr>
<tr>
<td>15 Oct 2008</td>
<td>0.99</td>
<td>2.2</td>
<td>0.90</td>
</tr>
<tr>
<td>11 Nov 2008</td>
<td>0.99</td>
<td>5.2</td>
<td>0.92</td>
</tr>
<tr>
<td>26 Nov 2008</td>
<td>1.00</td>
<td>6.5</td>
<td>1.00</td>
</tr>
<tr>
<td>Average</td>
<td>1.01</td>
<td>2.1</td>
<td>0.94</td>
</tr>
</tbody>
</table>

a LI-COR 820 was used indoors at all sites. Outdoor CO2 data were collected by T85b at S1 and S6, and T85a at S4 and S5. The alternate Q-Trak was used to collect supplementary indoor data. T85b was the only Q-Trak available during monitoring at S3.
Table B.3. Calibration parameters for ozone monitors

<table>
<thead>
<tr>
<th>Date</th>
<th>Instrument 2B2a</th>
<th>Instrument 2B2b</th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Slope</td>
<td>Offset (ppb)</td>
<td>Slope</td>
<td>Offset (ppb)</td>
</tr>
<tr>
<td>22 May 2008</td>
<td>—</td>
<td>-1.8</td>
<td>—</td>
<td>-0.8</td>
</tr>
<tr>
<td>17 June 2008</td>
<td>—</td>
<td>-0.1</td>
<td>—</td>
<td>-0.5</td>
</tr>
<tr>
<td>3 July 2008</td>
<td>—</td>
<td>-1.7</td>
<td>—</td>
<td>-1.7</td>
</tr>
<tr>
<td>15 July 2008</td>
<td>0.98</td>
<td>-1.7</td>
<td>1.00</td>
<td>-1.7</td>
</tr>
<tr>
<td>25 September 2008</td>
<td>—</td>
<td>-1.5</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td>15 October 2008</td>
<td>—</td>
<td>-1.5</td>
<td>—</td>
<td>-0.9</td>
</tr>
<tr>
<td>11 November 2008</td>
<td>—</td>
<td>-1.8</td>
<td>—</td>
<td>0.1</td>
</tr>
<tr>
<td>26 November 2008</td>
<td>—</td>
<td>-0.7</td>
<td>—</td>
<td>0.4</td>
</tr>
<tr>
<td><strong>Average</strong></td>
<td>—</td>
<td><strong>-1.4</strong></td>
<td>—</td>
<td><strong>-0.7</strong></td>
</tr>
</tbody>
</table>

*aAt S2-S6, instrument 2B2a was outdoors and 2B2b was indoors. At S1, instruments were in the reverse locations.

Table B.4. Calibration parameters for nitric oxide monitors.

<table>
<thead>
<tr>
<th>Date</th>
<th>Instrument 2B4a</th>
<th>Instrument 2B4b</th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Slope</td>
<td>Offset (ppb)</td>
<td>Slope</td>
<td>Offset (ppb)</td>
</tr>
<tr>
<td>May 2008</td>
<td>1.07</td>
<td>-6.0</td>
<td>1.05</td>
<td>7.4</td>
</tr>
<tr>
<td>June 2008</td>
<td>—</td>
<td>-6.8</td>
<td>—</td>
<td>8.2</td>
</tr>
<tr>
<td>July 2008</td>
<td>1.06</td>
<td>-7.7</td>
<td>1.05</td>
<td>4.7</td>
</tr>
<tr>
<td>September 2008</td>
<td>0.91</td>
<td>-4.4</td>
<td>0.94</td>
<td>0.7</td>
</tr>
<tr>
<td>October 2008</td>
<td>—</td>
<td>-6.4</td>
<td>—</td>
<td>-5.0</td>
</tr>
<tr>
<td>November 2008</td>
<td>0.89</td>
<td>-14.7</td>
<td>0.90</td>
<td>-1.6</td>
</tr>
<tr>
<td>November 2008</td>
<td>—</td>
<td>—</td>
<td>0.90</td>
<td>-4.0</td>
</tr>
<tr>
<td><strong>Average</strong></td>
<td><strong>0.98</strong></td>
<td><strong>-7.7</strong></td>
<td><strong>0.97</strong></td>
<td><strong>1.5</strong></td>
</tr>
</tbody>
</table>

*aAt all sites, instrument 2B4a was outdoors and 2B4b was indoors.*
Appendix C: Time-series figures for school sites S1-S6

This appendix provides time-series figures for the school days monitored at each site. Included in each set of plots are time-series figures at 1-minute resolution for CO₂, particle number (PN), temperature (T), number of open doors and windows, and number of students present. Also included is a 1-hour resolved time-series plot of the centrally monitored wind speed. The calculated air-exchange rate (AER) and indoor proportion of outdoor particles (IPOP) are also shown, and are presented as constant over the period for which they were calculated (See Chapter 3 for more details regarding the calculation of the air-exchange rate and indoor proportion of outdoor particles). Data are shown for the period that is nominally considered a school day: 8:00 to 16:00. Students were generally present for a subset of this time, as illustrated by the student occupancy plot, but teachers sometimes arrived before 8:00 or remained in the classroom beyond 16:00. To better interpret the time-series figure for the number of open doors and windows, the total number of doors and windows in each classroom is specified in the figure captions. Windows are only included if they were opened at any point during the monitoring period. Details regarding the layout and location of the classrooms are provided in Table 2.1, and a floor plan of each classroom is presented in Appendix A.
Figure C.1. Time-series figures for June 3rd, 2008 at S1. This classroom had 2 doors. The windows were never opened during the monitored period.
Figure C.2. Time-series figures for June 4th, 2008 at S1. This classroom had 2 doors. The windows were never opened during the monitored period.
Figure C.3. Time-series figures for June 5th, 2008 at S1. This classroom had 2 doors. The windows were never opened during the monitored period.
Figure C.4. Time-series figures for June 6th, 2008 at S1. This classroom had 2 doors. The windows were never opened during the monitored period. A pancake cooking PN source event occurred in the morning on this day. The PN concentration reached a peak of 149,000 cm$^{-3}$ as a result of this event (peak is truncated in the figure).
Figure C.5. Time-series for October 6th, 2008 at S2. This classroom had one door and two windows that were opened at some point during the monitored period.
Figure C.6. Time-series figures for October 13\textsuperscript{th}, 2008 at S2. This classroom had one door and two windows that were opened at some point during the monitored period.
Figure C.7. Time-series figures for October 14th, 2008 at S2. This classroom had one door and two windows that were opened at some point during the monitored period.
Figure C.8. Time-series figures for October 20$^{th}$, 2008 at S3. This classroom had two sets of double doors, but only 3 doors were opened during the monitored period. The windows were inoperable. A birthday candle PN source event occurred in the afternoon on this day.
Figure C.9. Time-series figures for October 21\textsuperscript{st}, 2008 at S3. This classroom had two sets of double doors, but only 3 doors were opened during the monitored period. The windows were inoperable. No outdoor CO\textsubscript{2} data were collected on this day. No air-exchange rate or IPOP determinations were made from data collected on this day.
Figure C.10. Time-series figures for October 22\textsuperscript{nd}, 2008 at S3. This classroom had two sets of double doors, but only 3 doors were opened during the monitored period. The windows were inoperable. A central heater PN source event occurred in the morning on this day. No outdoor CO\textsubscript{2} data was collected until \textasciitilde14:30.
Figure C.11. Time-series figures for November 4th, 2008 at S4. This classroom had one door and three windows that were opened at some point during the monitored period.
Figure C.12. Time-series figures for November 5th, 2008 at S4. This classroom had one door and three windows that were opened at some point during the monitored period.
Figure C.13. Time-series figures for November 6th, 2008 at S4. This classroom had one door and three windows that were opened at some point during the monitored period.
Figure C.14. Time-series figures for November 18th, 2008 at S5. This classroom had two doors. The windows were inoperable.
Figure C.15. Time-series figures for November 19th, 2008 at S5. This classroom had two doors. The windows were inoperable.
Figure C.16. Time-series figures for November 20th, 2008 at S5. This classroom had two doors. The windows were inoperable. A central heater PN source event occurred in the morning on this day.
Figure C.17. Time-series figures for December 2nd, 2008 at S6. This classroom had one door. The windows were never opened during the monitored period.
Figure C.18. Time-series figures for December 8th, 2008 at S6. This classroom had one door. The windows were never opened during the monitored period.
Appendix D: Floor plans of the four investigated Beijing apartments

This appendix presents floor plans for the four investigated apartments.
Figure D.1. Floor plan of Beijing apartment A1. Dimensions were measured by a researcher using a tape measure.
Figure D.2. Floor plan of Beijing apartment A2. Dimensions were measured by a researcher using a tape measure.
Figure D.3. Floor plan of Beijing apartment A3. Area measurements were acquired from official construction plans.
Figure D.4. Floor plan of Beijing apartment A4. Dimensions were measured by a researcher using a tape measure.