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Seasonal variability in environmental tobacco smoke exposure in public housing developments

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Abstract

The risk of tobacco smoking and second-hand smoke (SHS) exposure combined are the leading contributors to disease burden in high-income countries. Recent studies and policies are focusing on reducing exposure to SHS in multiunit housing (MUH), especially public housing. We examined seasonal patterns of SHS levels within indoor common areas located on Boston Housing Authority (BHA) properties. We measured weekly integrated and continuous fine particulate matter (PM_{2.5}) and passive airborne nicotine in six buildings of varying building and occupant characteristics in summer 2012 and winter 2013. The average weekly indoor PM_{2.5} concentration across all six developments was 9.2 µg/m³, higher during winter monitoring period (10.3 µg/m³) compared to summer (8.0 µg/m³). Airborne nicotine concentrations ranged from no detection to about 5000 ng/m³ (mean 310.7 ng/m³). Nicotine levels were significantly higher in the winter compared to summer (620.0 vs. 85.0 ng/m³; 95% CI: 71.7 – 998.2). Smoking related exposures within Boston public housing vary by season, building types, and resident smoking policy. Our results represent exposure disparities which may contribute to health disparities in low-income communities, and highlight the potential importance of efforts to mitigate SHS exposures during winter when outdoor-indoor exchange rates are low and smokers may tend to stay indoors. Our findings support the use of smoke-free policy as an effective tool to eliminate SHS exposure, and protect nonsmokers, especially residents of MUH.

Keywords

Indoor air pollution; Second-hand smoke; Particulate matter; Airborne nicotine; Seasonal patterns; Multi-unit housing

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Introduction

Tobacco smoking and second-hand smoke (SHS) exposure combined are responsible for an estimated 6.3 million annual deaths worldwide (Lim et al., 2012). These two risk factors are the leading contributors to disease burden in high-income countries, including the U.S. (Lim et al., 2012). These estimates include significant morbidity and mortality among non-smoking populations (Scientific Committee on Tobacco and Health (SCOTH), 2004; Surgeon General, 2006). SHS contains fine particles (its largest component by mass) as well as numerous gaseous pollutants that are known carcinogens and toxins (National Toxicology Program, 2011; U.S. Department of Health and Human Services, 2010). Adverse health effects from SHS exposure include heart diseases, cancers (e.g. lung, breast and nasal sinus), asthma and other respiratory illnesses (mostly in children), and birth outcomes (e.g. low birth weight, sudden infant death syndrome (SIDS)) (Jones et al., 2011; National Toxicology Program, 2011; Scientific Committee on Tobacco and Health (SCOTH), 2004; Surgeon General, 2006). Currently in the U.S., tobacco smoking and SHS exposure account for about 20 percent of all deaths each year (Centers for Disease Control and Prevention, 2008; Centers for Disease Control and Prevention, 2013). While there are declines in the prevalence of smoking, nearly 3% of Americans, including about 200,000 children younger than 18 months, are still living with smoking-related diseases. Thus the current smoking-attributable mortality estimate of about half a million is likely to remain high into the future (Centers for Disease Control and Prevention, 2008; Centers for Disease Control and Prevention, 2013; Surgeon General, 2014).

The highest exposure to SHS occurs in homes, followed by workplaces (National Toxicology Program, 2011; Pirkle et al., 2006). Although outdoor fine particles infiltrate many indoor environments, and indoor sources other than smoking (e.g. cooking activities) also contribute to particle concentrations, SHS is often the predominant source of fine particle pollution in many homes (King et al., 2010; National Toxicology Program, 2011). While SHS exposure have declined steadily in the US since the late 1990s, nearly a third of the American population who are nonsmokers, including about 40% of children aged 3–11 years, are still exposed; and disparities in exposure persist across age, sex, race/ethnicity, and income groups (Centers for Disease Control and Prevention, 2013; Pirkle et al., 2006; Surgeon General, 2006). The recent Surgeon General's report on SHS concluded that there is no risk-free level of exposure, and that even short-term exposures can have adverse health consequences for both children and adults (Surgeon General, 2006). Thus, there is a need to understand the determinants of SHS exposure and to design and implement effective mitigation strategies.

In the US, SHS exposure tends to be higher among persons with low incomes, of whom a disproportionate number are among the 80 million Americans living in multi-unit housing (MUH) (Centers for Disease Control and Prevention, 2010; King et al., 2013a; King et al., 2013b; Winickoff et al., 2010). Among low income earners, the risk of SHS-related morbidity and mortality is also elevated (Surgeon General, 2006). Within MUH, sources of exposure may originate from an individual's own unit or from neighboring units through shared air spaces (e.g. ventilation systems, windows, elevator shafts, hallways and leaks in walls) (King et al., 2010; Kraev et al., 2009). However, little is known about the

determinants of exposure variability in these settings. Exposure differentials in MUH may be due to housing, seasonal, and environmental factors. Smoking-related behavior and building characteristics (e.g. design, operations, and age) can all contribute to SHS exposure variability. Air exchange, which is heavily influenced by design, mechanical systems, and human behavior, can also strongly influence SHS exposures. For example, in colder climates and seasons, reduced air exchange during winter (when windows are tightly closed and well insulated) may increase indoor concentrations of fine particulate matter (PM_{2.5}: particulate matter less than 2.5µm in aerodynamic diameter) associated with smoking activity (Zota et al., 2005). These seasonal differences may also coincide with increased amount of times spent indoors, further increasing exposure to non-smoking occupants.

A number of studies have examined SHS exposure, its indicators, or markers in MUH with emphasis on the role of resident smoking policies (King et al., 2010; Kraev et al., 2009; Van Deusen et al., 2009; Wallace, 1996; Wilson et al., 2011), but only a few have assessed seasonal variations in exposure, have analyzed variations within buildings, or have been in public housing units that include a range of varied building and occupant characteristics. In this paper, we report on a study that aimed to examine seasonal patterns of tobacco smoke pollution levels within indoor common areas of MUH located on Boston Housing Authority (BHA) properties. Besides being among the few studies of SHS in low-income MUH developments in urban settings, this study adds important contributions to existing literature. By simultaneously collecting integrated data on fine particle and airborne nicotine concentrations across different seasons and locations, we are able to (i) examine the relationship between both indicators of SHS exposure; (ii) assess how this relationship varies between seasons; and (iii) evaluate the distribution of their concentrations within and between different building types.

Materials and Methods

Overview

While there is no single ideal environmental marker for SHS exposure, atmospheric markers such as nicotine and fine particles have been established as reliable measures of the overall magnitude, duration, and frequency of exposure (Surgeon General, 2006). Airborne nicotine is specific to SHS, though it may exist as a result of older tobacco use events (e.g. off-gassing from nicotine previously deposited on surfaces). SHS also contains a mixture of compounds in fine particulate phase that are suspended and easily inhaled into the lung. These can be inexpensively measured in real-time. Examining PM_{2.5} levels in conjunction with airborne nicotine concentrations provides a valid estimate of SHS exposure.

Study design

We measured weekly PM_{2.5} and passive nicotine concentrations in six buildings at different BHA developments. The study buildings were purposively selected to represent the range of building and occupant characteristics within the portfolio of the housing authority. In each study building, we collected two indoor PM_{2.5} samples in both lower and upper floor hallways along with co-located nicotine samples. We used a combination of integrated gravimetric and continuous real-time monitors to measure PM_{2.5} concentrations. Five to ten

airborne nicotine samples were simultaneously collected per building per monitoring session. The study buildings varied by height (low-, or mid-, vs. high-rise), resident group (elderly/disabled residents vs. families), and resident smoking policies (smoke-free vs. smoking permitted). Concurrent with the indoor monitors, an outdoor PM_{2.5} measurement was conducted at each building using both gravimetric and continuous methods. Field measurements were conducted at each site in summer 2012 (Boston's summer average high and low temperature are 82 °F (27 °C) and 66 °F (27 °C), respectively) and repeated in winter 2013 (Boston's winter average high and low temperature are 22 °F (-6°C) and 16 °F (-8°C), respectively). The air sampling monitors were deployed following consent from both BHA management and building managers.

PM measurement methods

Integrated PM—We collected gravimetric PM_{2.5} samples using Personal Exposure Monitors (Harvard School of Public Health, HSPH, Boston, MA) (Demokritou et al., 2001) with a D₅₀ of 2.5 µm (aerodynamic diameter) at 1.8 liters per minute (lpm) (±10%) and an internal level greased impaction surface. Inside the monitors, Whatman drain discs were used to back-support PTFE filters with ring (Pall Life Sciences, Teflo, 0.2 µm pore size, 37 mm diameter). The monitors were connected by Tygon PVC tubing to electric operated pumps placed in a soundproof box. In four of the study buildings, the monitors were placed in vacant units for security reasons, drawing air from the hallways through Tygon PVC tubing connected through the door peepholes. In the other two buildings, the monitors were placed directly in the hallways. Air flow rates were checked at the beginning and end of each sampling period using a calibrated rotameter.

All filters were weighed before and after sampling on a Mettler Toledo MT5 microbalance maintained at HSPH laboratory, after being conditioned in a temperature- and relative humidity (RH)-controlled environment (20.5 ± 0.2°C, 39 ± 2% RH) for at least a day, and statically discharged via a polonium source. In both pre- and post-weighing, samples were weighed twice; a third weighing was carried out only if the first two masses were more than 5 µg apart. After the third weighing, the average of the two measured masses within 5 µg of each other was used for calculating concentrations. Linearity, zero, and span of the microbalance were checked via a set of class 'S' weights after every batch of 10 samples.

Continuous PM—We used SidePak model AM510 monitors (TSI Inc., Shoreview, MN, USA) for continuous measurement of PM_{2.5} samples. PM_{2.5} concentrations were measured every second, averaged and recorded at 1-minute intervals. The PM monitors were operated at a flow rate of 1.7 lpm using 2.5µ size selective inlets for PM_{2.5}. The PM monitors were calibrated to a zero filter prior to each weekly sampling period to avoid drifts. PM monitors have zero stability of ± 1 µg/m³ over 24 hours and temperature coefficient of 0.5 µg/m³ per °C.

The continuous PM_{2.5} concentrations measured by laser photometry method are known to be inexact in magnitude due to the use of artificial aerosols for factory calibration, which may have different characteristics (e.g. shape, size, density, and refractive index) than those in our study (Levy et al., 2001), and also because measured concentration may be affected

by factors such as temperature and humidity in our study setting (Chakrabarti et al., 2004; Kingham et al., 2006). As such, PM_{2.5} data were adjusted using a correction factor (CF), calculated as the ratio of the co-located integrated (gravimetric) PM_{2.5} measurement to the average of the minute-by-minute continuous measurements over the same time period. We calculated CFs separately for each floor. The median (interquartile range) of the CFs were 0.37 (0.32 – 0.41) for winter, and 0.36 (0.30 – 0.42) for summer monitoring periods. Our CFs are comparable to those reported by other studies in similar settings (King et al., 2010).

Nicotine measurement method

Concurrent with the PM_{2.5} samples, integrated airborne nicotine concentrations were measured using passive monitors (Hammond and Leaderer, 1987). The monitoring devices were distributed in the hallways across multiple floors throughout the study building. The passive diffusion monitors collected vapor-phase nicotine onto a 37-mm polystyrene filter treated with sodium bisulfate held by a sampling cassette. The filters were covered by a diffusion screen allowing air to pass through at a constant flow rate. Filters were analyzed using gas chromatography with a nitrogen/phosphorus detector in a laboratory maintained at the University of California, Berkeley. The analysis methods have a lower limit of detection (LOD) of 2.0 ng/m³. Nearly 93% our samples were above the method LOD value.

Data management and statistical analysis

At the start of each sampling period, date, time, and logging interval on the monitors were set. The minute-by-minute continuous PM_{2.5} concentration data were compiled into a single dataset by matching on the site, date and time stamps, with each record representing a unique site, date, time, and PM concentration.

We provide graphical presentation as well as descriptive statistics for PM and nicotine measurements. We tested mean differences using two sided t-tests within and between the two monitoring seasons using the integrated PM samples; geometric means were assessed using the integrated nicotine samples. Median values were reported for the continuous samples after adjustment using CFs as detailed above. In order to isolate effects related to indoor particles of indoor origin, we tested for relationships between indoor PM_{2.5} and nicotine by season, adjusting for outdoor PM_{2.5}. We also wanted to account for the effect of reduced penetration of outdoor fine particles in winter in our comparison. While ultrafine particles may have penetrations factors near unity (Liu and Nazaroff, 2003; Thatcher et al., 2003), we assumed a penetration factor of 0.5, as suggested in a previous study (Long et al., 2001).

Analyses were done using the open-source statistical package R version 3.0.0 (R Project for Statistical Computing, Vienna, Austria).

Results

Indoor PM concentrations

We collected 20 week-long integrated and 20 continuous PM_{2.5} along with 66 airborne nicotine samples from six MUH buildings located across BHA developments. The average

weekly indoor PM_{2.5} concentration across all six developments was $9.2 \pm 4.7 \mu\text{g}/\text{m}^3$. Average weekly PM_{2.5} levels varied considerably across sites and season, ranging from less than $5 \mu\text{g}/\text{m}^3$ during summer measurements to more than $20 \mu\text{g}/\text{m}^3$ during winter measurements (Table 1). In general, mean (standard deviation) PM_{2.5} samples were higher during the winter monitoring period ($10.3 \pm 6.3 \mu\text{g}/\text{m}^3$) compared to summer ($8.0 \pm 1.9 \mu\text{g}/\text{m}^3$), but the two seasons were not significantly different.

In each study building over the two seasons, indoor PM_{2.5} samples in lower and upper floor hallways were statistically similar, though samples from upper floors were generally higher during winter measurements period (12.4 vs. $10.5 \mu\text{g}/\text{m}^3$) (Table 2). Similarly, while we saw no statistical difference between study buildings types (i.e. by height), PM_{2.5} concentrations were highest in high-rise buildings, followed by mid- and low-rise buildings. While resident group was not a statistically significant factor in determining PM_{2.5} levels within seasons, we observed differences between the two seasons; winter levels in buildings which housed elderly and disabled tenants were roughly twice those observed in individual ‘family’ buildings (14.7 vs. $7.4 \mu\text{g}/\text{m}^3$). Buildings with smoking-allowed policies had higher PM_{2.5} concentrations ($11.8 \mu\text{g}/\text{m}^3$) compared to smoke-free buildings ($6.9 \mu\text{g}/\text{m}^3$) in the winter. Within individual family buildings, those with smoking-allowed policies had slightly higher PM_{2.5} concentrations ($8.0 \mu\text{g}/\text{m}^3$) than family smoke-free buildings ($6.9 \mu\text{g}/\text{m}^3$), and both were lower than levels in elderly/disabled buildings ($14.7 \mu\text{g}/\text{m}^3$) (Figure 1). During summer, levels were highest in family smoking-allowed buildings.

For nearly 20% of the time in the winter, our indoor continuous samples exceeded the outdoor daily U.S. National Ambient Air Quality Standards (NAAQS) of $35 \mu\text{g}/\text{m}^3$ (Table 1). Figure 2 shows boxplots of the minute-to-minute continuous PM_{2.5} concentrations averaged over a one week period, and stratified by time of day and season after standardizing for measurement error. In smoking-permitted buildings, our real-time continuous PM data revealed an increase from around 03:00 PM to 12:00 AM, peaking or stabilizing between 06:00 and 09:00 PM. Median PM_{2.5} levels in smoke-permitted buildings were higher across all six-hour time intervals in both seasons compared to smoke-free buildings; and were pronounced for the winter samples between 6:00 PM and midnight, with median (interquartile range) of $20.8 \mu\text{g}/\text{m}^3$ ($18.2 - 24.4$), followed by midnight to early morning. The least median PM_{2.5} concentration was recorded in the summer between midday and 6:00 PM ($5.6 \mu\text{g}/\text{m}^3$; IQR $5.1 - 6.0$). While outdoor continuous samples also showed some variability, especially for the winter measurement, the indoor median values were much higher by nearly a factor of $\sim 1.5 - 2$.

Indoor nicotine concentrations

Overall, airborne nicotine concentrations ranged from no detection to about $5000 \text{ ng}/\text{m}^3$ (Table 1), with the mean $310.7 \pm 799.3 \text{ ng}/\text{m}^3$. Similar to PM, mean nicotine levels were significantly higher in the winter compared to summer (620.0 vs. $85.0 \text{ ng}/\text{m}^3$; 95% CI: $71.7 - 998.2$). During the winter monitoring period, nicotine concentrations were highest in the high-rise building with geometric mean $438.3 \text{ ng}/\text{m}^3$ compared with either mid- or low-rise or both (geometric mean $< 70 \text{ ng}/\text{m}^3$). Similarly, nicotine concentrations were highly associated with building occupancy (‘family’ vs. ‘elderly/disabled’) in the winter, higher in

buildings that housed elderly and disabled residents compared to ‘family’ buildings ($p < 0.05$) (Table 2). In general, nicotine levels were highest in elderly/disabled buildings (1223.3 ng/m³), followed by family smoking-allowed (322.1 ng/m³) and family smoke-free buildings (45.1 ng/m³) in the winter. However, during the summer monitoring period, family smoking-allowed buildings recorded the highest nicotine concentrations.

Indoor PM-nicotine correlations

The average weekly outdoor PM_{2.5} concentrations during both monitoring seasons were similar; indicating that there were no unusual meteorologic factors outdoors during the measurement periods which could lead to differences in indoor measurements between the two seasons (Table 1). After adjusting for indoor concentrations of ambient particles, the average PM_{2.5} concentrations and average nicotine concentrations measured in the winter were significantly correlated ($r = 0.88$, 95% CI: 55 – 97) (Figure 3). We found weak correlation for summer samples.

Discussion

The aim of this study was to assess seasonal patterns of ETS levels within indoor common areas of MUH that differ by building and occupant characteristics in the BHA, one of the largest public housing authorities in the U.S. The study uniquely combined weekly integrated and continuous PM_{2.5} samples with co-located integrated airborne nicotine concentrations from multiple floors in public housing buildings across two seasons. We observed levels comparable to prior studies that measured PM_{2.5} and nicotine concentrations in similar settings using similar methods across the U.S. (Brown et al., 2008; Hammond and Leaderer, 1987; King et al., 2010; Kraev et al., 2009; Van Deusen et al., 2009; Wallace, 1996). We found evidence of seasonal differences in both fine particle and nicotine concentrations, with higher levels recorded during the winter measurement period. These differences are likely shaped by reduced air exchange (e.g. doors and windows closed in winter) (Zota et al., 2005), and changes in smoking activity patterns, since smoking may be more likely to occur indoors during colder seasons (Kaufman et al., 2011).

Our findings also suggest exposure differentials across building types, occupant groups and resident smoking policies. Notably, particle and nicotine concentrations were elevated indoors in high-rise buildings occupied by elderly disabled individuals. Higher concentrations were also observed for buildings that allowed smoking. The timing of when continuous PM data increased, peaked and stabilized in the smoking-permitted buildings was consistent with King’s study (2010) which found the highest median PM_{2.5} levels between 4:00 PM and 11:59 PM. This time window was shown to correspond to the time period when individuals are typically at home and active (Klepeis et al., 2001).

We found a strong correlation between average indoor PM_{2.5} and average nicotine concentrations measured in the winter; high correlation persisted even after adjusting for potential influence of ambient particles. This association between airborne nicotine and PM_{2.5} is most pronounced at high nicotine concentrations seen in winter, a clear indication of smoking as the source of the PM_{2.5} levels we observed. This evidence is consistent with the low to moderate influence of ambient PM_{2.5} on indoor PM_{2.5} in winter, and the

relatively low ambient levels of PM_{2.5} in Boston (Brown et al., 2008). Thus, indoor PM_{2.5} levels which are significantly above outdoor concentrations are likely driven by indoor sources (Rojas-Bracho et al., 2000), such as smoking. Previous studies found smoking status and the number of cigarettes reported being smoked in the home per day were highly predictive of measured nicotine concentrations in MUH settings (Gehring et al., 2006; Kraev et al., 2009).

While USEPA PM_{2.5} ambient standards were established for ambient air, we clearly see seasonal indoor levels which exceed these health-based standards (for at least some of the time), making PM_{2.5} exposure levels observed in some of these public buildings health-relevant. These common-area measurements are proxies for personal exposures within MUH and highlight the problem of SHS movement between units, supporting the fact that SHS contamination is not limited to homes with active smokers but may also affect non-smokers as suggested elsewhere (King et al., 2010; Kraev et al., 2009). These results represent exposure disparities which may contribute to health disparities in low-income communities, particularly in subsidized MUH that contain multiple vulnerable populations including children and the elderly. Although better ventilation in the summer may in part be able to reduce exposure to indoor pollution from SHS, our results indicate that smoke-free policies are necessary to protect nonsmokers from SHS in MUH all year round. (Pickett et al., 2006; Pirkle et al., 2006; Surgeon General, 2006; U.S. Department of Health and Human Services, 2010). If implemented widely, smoke-free policies may protect the elderly and young populations, as well as individuals with chronic illnesses, who are especially susceptible to the adverse effects of smoking (U.S. Department of Health and Human Services, 2010). In addition, smoke-free policies may result in substantial cost savings due to reduced SHS-related health and property expenses (King et al., 2013b). Our outdoor PM_{2.5} measurements were consistently lower than the NAAQS ambient 24-hour standard of 35 ug/m³, which reinforces the notion that PM-related risk is highest indoors for our study population. NAAQS are meant to provide public health protection from outdoor levels, including protecting the health of vulnerable populations like children and the elderly, but as shown in Table 1, the exceedence of the NAAQS 24-hour and annual standards are quite high among our indoor samples.

There are multiple innovative features and strengths of this study. The study made use of a unique and rich dataset containing both integrated and real-time continuous PM_{2.5} samples as well as co-located and multiple integrated airborne nicotine concentrations from MUH across two seasons. Data from different seasons allowed for assessment of seasonal variations in exposure. Samples from hallways on multiple floors of different study building types permitted us to evaluate variations within buildings and across different building types. Additionally, we were able to examine exposure disparities by resident groups. Most importantly, our study compared concentrations by resident smoking policies to understand the impact of smoke-free versus smoking permitted policies on exposure levels within MUH.

Our study had a number of limitations that are common to many field research studies. First, PM measured with a continuous monitor is subject to error because of the light scattering technique it uses. Thus, we systematically applied a CF to PM data, but the steps involved in

calculating CFs might introduce additional uncertainty. Second, while fine particles are not specific to tobacco smoke as they can be emitted from many other combustion sources, we have observed strong correlation between the two, as were other prior studies (King et al., 2010; Kraev et al., 2009). Lastly, our study represents the experience of a purposive sample of buildings in an urban public housing authority in a single northeastern city, so our findings may not be generalizable to all MUH settings.

Conclusion

Smoking related exposures within Boston public housing vary by season, building types, and resident smoking policy. Our results highlight the potential importance of SHS exposures during winter when outdoor-indoor exchange rates are low and smokers may tend to stay indoors. These findings support calls and efforts to prohibit smoking in all U.S. MUH to protect health, which could result in significant savings in SHS-related healthcare and property expenditures. Such policies have been encouraged by the U.S. Department of Housing and Urban Development as a means to improve health for public housing residents (U.S. Department of Health and Human Services, 2010). Our research also highlights the importance of seasonal variation in longitudinal studies of SHS levels in MUH. While existing ambient PM_{2.5} standards were established for ambient air, we clearly see seasonal indoor levels which exceed these health-based standards for at least some of the time. Elevated exposures during winter may be especially relevant for MUH sub-populations, such as young children and the elderly, who spend a greater portion of their time at home (especially during colder months) and who may be susceptible to SHS-related health effects.

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Practical implications

These findings have important public health implications for smoke-free policies, which can eliminate or reduce exposure to involuntary tobacco smoke, and thus reduce its associated health effects in MUH. Our research also highlights the importance of seasonal variation in longitudinal studies of SHS levels in MUH.

Figure 1. Weekly mean PM_{2.5} and airborne nicotine concentrations by season and building occupancy type.

Figure 2.

Boxplot of the minute-minute continuous PM_{2.5} concentrations averaged over a one week period, and stratified by time of day and season. Continuous samples were standardized for measurement error as detailed in Materials and Methods.

Figure 3.

The relationship between winter PM_{2.5} and airborne nicotine concentrations. PM_{2.5} concentrations were adjusted for indoor concentrations of ambient particles; 100% and 50% of summer and winter outdoor levels were removed from summer and winter indoor levels, respectively. The solid and dashed lines represent regression lines.

Table 1

Summary statistics of nicotine and PM_{2.5} concentrations by season. Outdoor PM_{2.5} annual arithmetic mean for Boston in 2012 was below 10 µg/m³.

	Winter	Summer
Indoor integrated nicotine (ng/m³)		
<i>No. of samples</i>	28	38
<i>Median</i>	82.5	61.4
<i>Mean ± SD</i>	620.0 ± 1169.4	85.0 ± 79.8
<i>Min – Max</i>	0.01 – 4868.0	1.1 – 355.4
<i>Geometric mean</i>	51.0	52.4
Indoor integrated PM_{2.5} (µg/m³)		
<i>No. of samples</i>	10	10
<i>Median</i>	7.7	7.9
<i>Mean ± SD</i>	10.3 ± 6.3	8.0 ± 1.9
<i>Min – Max</i>	5.6 – 24.1	3.8 – 10.9
<i>Geometric mean</i>	9.0	7.8
Indoor continuous PM_{2.5} (µg/m³)		
<i>Range: percent of time above 15 µg/m³</i>	1 – 44	5 – 15
<i>Range: percent of time above 35 µg/m³</i>	0 – 19	1–3
Outdoor integrated PM_{2.5} (µg/m³)		
<i>No. of samples</i>	5	5
<i>Mean</i>	6.2	7.1
<i>Min – Max</i>	5.1 – 8.0	5.0 – 8.8

Table 2

Average indoor integrated PM_{2.5} and nicotine levels, by season, building and occupancy characteristics.

Outdoor PM_{2.5} annual arithmetic mean for Boston in 2012 was below 10 µg/m³.

Indoor integrated PM _{2.5} (µg/m ³)	Winter		Summer	
	No. of samples	Mean	No. of samples	Mean
Resident group				
<i>Elderly/disabled</i>	4	14.7	4	7.9
<i>Individual families</i>	6	7.4	6	8.1
Smoking policies				
<i>Smoke-free</i>	3	6.9	3	7.9
<i>Smoking permitted</i>	7	11.8	7	8.1
Building height				
<i>Low-rise</i>	2	5.9	2	7.9
<i>Mid-rise</i>	4	8.2	4	8.3
<i>High-rise</i>	4	14.7	4	7.7
Floor hallways				
<i>Lower</i>	4	10.5	4	8.5
<i>Upper</i>	4	12.4	4	7.7
Indoor integrated nicotine (ng/m³)				
Resident group				
<i>Elderly/disabled</i>	12	1223.3	14	66.7
<i>Individual families</i>	15	137.4	23	96.2
Smoking policies				
<i>Smoke-free</i>	10	45.1	13	82.0
<i>Smoking permitted</i>	17	958.2	24	86.7
Building height				
<i>Low-rise</i>	4	71.8	6	125.6
<i>Mid-rise</i>	11	161.3	17	85.8
<i>High-rise</i>	12	1223.3	14	66.7
Floor hallways				
<i>Lower</i>	13	872.5	17	64.8
<i>Upper</i>	9	567.9	14	92.2