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Infiltration of forest fire and residential wood smoke: an evaluation of air cleaner effectiveness

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Communities impacted by fine-particle air pollution (particles with an aerodynamic diameter less than 2.5 μm; $PM_{2.5}$) from forest fires and residential wood burning require effective, evidence-based exposure-reduction strategies. Public health recommendations during smoke episodes typically include advising community members to remain indoors and the use of air cleaners, yet little information is available on the effectiveness of these measures. Our study attempted to address the following objectives: to measure indoor infiltration factor (F_{inf}) of $PM_{2.5}$ from forest fires/wood smoke, to determine the effectiveness of high-efficiency particulate air (HEPA) filter air cleaners in reducing indoor $PM_{2.5}$, and to analyze the home determinants of F_{inf} and air cleaner effectiveness (ACE). We collected indoor/outdoor 1-min $PM_{2.5}$ averages and 48-h outdoor $PM_{2.5}$ filter samples for 21 winter and 17 summer homes impacted by wood burning and forest fire smoke, respectively, during 2004–2005. A portable HEPA filter air cleaner was operated indoors with the filter removed for one of two sampling days. Particle F_{inf} and ACE were calculated for each home using a recursive model. We found mean $F_{inf} \pm$ SD was 0.27 ± 0.18 and 0.61 ± 0.27 in winter ($n = 19$) and summer ($n = 13$), respectively, for days when HEPA filters were not used. Lower $F_{inf} \pm$ SD values of 0.10 ± 0.08 and 0.19 ± 0.20 were found on corresponding days when HEPA filters were in place. Mean $\pm$ SD ACE ([F_{inf} without filter–F_{inf} with filter]/F_{inf} without filter) in winter and summer were $55 \pm 38\%$ and $65 \pm 35\%$, respectively. Number of windows and season predicted F_{inf} ($P < 0.001$). No significant predictors of ACE were identified. Our findings show that remaining indoors combined with use of air cleaner can effectively reduce $PM_{2.5}$ exposure during forest fires and residential wood burning.
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Introduction

Smoke from forest fires and residential wood burning is a significant source of $PM_{2.5}$. Over one-third of total Canadian concentrations of particles with an aerodynamic diameter less than 2.5 μm ($PM_{2.5}$) are attributed to forest fire emissions (Rittmaster et al., 2006) while over one quarter of total $PM_{2.5}$ concentrations are attributed to residential wood burning (Environment Canada, 2006). In the US, wood is burned regularly in approximately 30 million homes and residential wood combustion is responsible for 9% of national space heating energy requirements (Houck et al., 1998). Forest fire frequency is expected to increase as a result of climate change (Flannigan et al., 2000) while recent emphasis on renewable energy sources raises the possibility of increased wood burning to supply household heating needs (Fischer and Schrattenholzer, 2001).

While forest fires and wood burning emit a complex mixture of pollutants, the release of fine particles is of particular concern due to their high emission rate, potential to cause adverse health effects, and their potential for long-range transport (Sapkota et al., 2005). In a recent review, Naeher et al. (2007) summarized the evidence of adverse health effects in communities exposed to forest fire and residential wood smoke. Exposure to forest fire smoke has been associated with increased respiratory symptoms (Aditama, 2000; Kunzli et al., 2006), increased COPD and asthma-related emergency room visits (Duclos et al., 1990), increased physician visits (Moore et al., 2006), and increased medication use (Kunzli et al., 2006). Similarly, exposure to wood smoke has been associated with declines in lung function (Koenig et al., 1993), increased respiratory symptoms in children (Larson and Koenig, 1994; Norris et al.,

1. Abbreviations: a_1, coefficient representing penetration of outdoor particles; a_2, coefficient representing decay of indoor particles; ACE, air cleaner efficiency; F_{inf}, infiltration factor; HEPA, high-efficiency particulate air; pDR, personal DataRAM; $PM_{2.5}$, particles with an aerodynamic diameter less than 2.5 μm; RH, relative humidity

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1999) and increased emergency room visits in both children (Larson and Koenig, 1994) and the general public (Norris et al., 2000).

A common public-health recommendation issued by health authorities during air-pollution episodes includes remaining indoors during times of high smoke levels, as it is believed that this will reduce residents' exposure to $PM_{2.5}$ levels (Emmanuel, 2000; US EPA, 2003). While the home is believed to provide a protective barrier against particulate air pollution, few studies have investigated infiltration of particles during episodes of high air pollution, such as those created by nearby forest fires. Studies investigating infiltration suggest that for smaller particles, including $PM_{2.5}$, penetration via building surfaces, such as open doors, windows or through building cracks can easily occur (Lai, 2002), raising questions as to the effectiveness of such a health recommendation. In addition to staying indoors, public health recommendations also include using air cleaners during episodes of forest fire and residential wood burning smoke (Emmanuel, 2000; US EPA, 2003). Aside from the lack of knowledge about outdoor $PM_{2.5}$ infiltration into homes, little information is available on the effectiveness of air cleaners in reducing indoor levels of $PM_{2.5}$ in homes. Studies that have evaluated the use of air cleaners during forest fire events in either reducing exposure or mitigating health effects have lacked accompanying exposure measurements (Mott et al., 2002), appropriate controls to assess the effectiveness of such an intervention (Henderson et al., 2005) or have been restricted to small numbers (<5) of homes (Henderson et al., 2005).

Measurements of infiltration, the fraction of outdoor particles that penetrate and remain suspended in indoor air, based upon simultaneous continuous indoor and outdoor monitoring provide a useful approach to estimate exposure among affected residents (Allen et al., 2003; Wu et al., 2006). Infiltration calculations include the penetration and decay rates of particles. While penetration of particles is not expected to be altered by air cleaner use, their decay rate is expected to increase. Air cleaners expel filtered air and particles remain in the air for a shorter period of time. This increases the decay rate and decreases the overall infiltration rate.

Accordingly, we measured infiltration and air cleaner effectiveness directly in homes in British Columbia affected by either forest fire smoke (during the summers of 2004 and 2005) or residential wood smoke (during winter 2004). Our sampling method allowed us to account for the dynamic nature of indoor and outdoor PM concentrations in occupied homes during these biomass combustion events.

Materials and methods

Study Design

We conducted indoor and outdoor $PM_{2.5}$ sampling in 21 homes affected by residential wood smoke in the winter (2004) and in 17 homes affected by forest fire smoke in the summer (2004–2005). Wintertime sampling was conducted in Prince George, a Northern Canadian community with high levels of PM, resulting in part from residential wood smoke (British Columbia Lung Association, 2006). In summer, sampling was conducted throughout Southern British Columbia in communities impacted by forest fire smoke. Selection of communities was based on information gathered from fire maps, wind direction, and satellite imagery. Where available, $PM_{2.5}$ monitoring data collected by the British Columbia Ministry of Environment were also used. These resources were available online and were monitored regularly to ensure that communities affected by forest fire smoke were identified as quickly as possible. Homes were recruited from volunteers with pre-existing respiratory disease and supplemented with homes of healthy volunteers as needed. Given the short duration time period required to recruit homes and collect measurements during fire events, no attempts were made to recruit a representative sample of homes. Recruitment methods included contacting local lung health support groups, the British Columbia Lung Association, and regional health authorities, as well as the distribution of introductory study letters, stories in community newspapers as well as airing a brief story on the local radio station of one community. The only exclusion criteria were that all volunteers who offered their homes had to be nonsmokers and no smoking could occur within the home during sampling. Although our purpose was to assess indoor infiltration of wood smoke from outdoors, residents in homes with wood stoves were also asked to refrain from operating them during the sampling period as this may have contributed to elevated particle levels indoors and affected our ability to estimate infiltration. This study was approved by the UBC Behavioural Research Ethics Committee (B03-0602).

Each home was sampled for a minimum of 48 h, during which time a portable high-efficiency particulate air (HEPA) filter room air cleaner (18150, Honeywell, Morristown, NJ, USA) was introduced into the main bedroom of the home as this room was considered to be occupied for the largest portion of time relative to other rooms. To investigate the relationship between infiltration and air cleaner operation within each home, the HEPA filter was installed in the air cleaner for only one of two sampling days. Filter installation was assigned randomly to either the first or second day and residents were blinded to filter status. The air cleaner's Clean Air Delivery Rate (CADR), describing airflow and particle removal efficiency was 150 for tobacco smoke, as specified by the manufacturer. On the basis of this CADR, the air cleaner removes approximately 89% of the particles in a room size of $9 \times 12\,\mathrm{ft}^2$, 74% of the particles in a room size of $12 \times 18\,\mathrm{ft}^2$, and 51% of the particles in a room size of $18 \times 24\,\mathrm{ft}^2$ (Office of Air and Radiation, 2006). All sampled rooms met the maximum room size of $15 \times 15\,\mathrm{ft}^2$ as specified by the manufacturer.

One-minute averages of $PM_{2.5}$ were monitored during the 48-h sampling period with a photometer (personal DataRAM (pDR)-1000, Thermo Andersen, Smyrna, GA, USA) located opposite the air cleaner in the main bedroom of the home. Additionally, 1 min average CO_2 concentrations were measured (Q-Trak 8551, TSI Incorporated, USA). During each sampling period, residents were asked to fully vacate the home for a minimum of 2 h. An average air-exchange rate (h^{-1}) was calculated from the decrease in CO_2 during this time. Indoor and outdoor relative humidity (RH) was measured (HOBO, H08-032-08, Onset Computer Corp., Pocasset, MA, USA; Q-Trak), to help assess pDR data quality, as instrument response has been shown to be affected at high RH (Quintana et al., 2000; Wu et al., 2005). To measure levoglucosan, a wood smoke tracer, as well as to calibrate the pDR response to a gravimetric measure, a 48-h average $PM_{2.5}$ filter sample was collected outside each home using a Harvard impactor (HI) measuring particles with an aerodynamic diameter less than 2.5 μm (Air Diagnostics and Engineering Inc., Harrison, ME, USA) at a flow rate of 4 l/min. The HI detection limit, defined as three times the SD of the field blanks divided by the mean sample volume was 4.2 $\mu g/m^3$. The lower limit of the pDR measurement range provided by the manufacturer was 1 $\mu g/m^3$. For quality control, both pDRs were collocated both indoors and outdoors for 10 min before, at the midpoint (when the HEPA filter status was changed) and after each 48-h sampling period.

Subject activity within the home was measured with a time activity log divided into half hour intervals to record information such as particle-generating activities and whether windows were open. This time interval was chosen to correspond to the time interval used for calculation of particle infiltration factors (F_{inf}). These time-activity data were used to identify indoor-generated $PM_{2.5}$ peaks as measured by the pDR. A similar protocol was followed for both summer and winter sampling with slight modifications to the winter protocol, specifically, a 60-h sampling duration and heating of outdoor pDRs to ensure reliable operation at cold temperatures.

Data Analysis

Any homes with <24 h of sampling for at least one of the 2 days, were excluded from analysis. Samples in which quality control measurements indicated poor agreement between the two pDR monitors were also removed prior to analysis. Specifically, samples with relative differences (average difference between the indoor and outdoor monitor reading for each test period/mean indoor and outdoor concentration over the test period) >1.0 and correlations <0.40 (identified as outliers based on examination of the distribution of correlations) between the indoor and outdoor monitors for each test period were removed prior to further analysis.

Particle F_{inf} were calculated separately for days when the HEPA filter was in place and when it was not in place. Prior to infiltration calculations, censoring algorithms (Allen et al., 2003) were used to separate indoor peaks into those resulting from indoor generated $PM_{2.5}$ and those resulting from infiltration of outdoor $PM_{2.5}$. Specifically, we identified peaks for 30 min averages of every half hour concentration value collected. Any increase in the indoor concentration $>50\%$ of the concentration from the previous 30 min average (Eq. (1)), without a subsequent increase in the outdoor concentration was considered to be of indoor origin. The use of continuous monitoring and censoring algorithms allowed us to collect data when the home was occupied and provides time-resolved data on concentration fluctuations during smoke episodes caused by both forest fire and wood burning.

Outdoor peaks were identified when a minimum 10% increase from the previous 30 min outdoor average was seen (Eq. (2)):

$$\text{indoor generated peak} = \frac{(C_{in})_t - (C_{in})_{t-1}}{(C_{in})_t} > 50\% \quad (1)$$

$$\text{outdoor peak} = \frac{(C_{out})_t - (C_{out})_{t-1}}{(C_{out})_t} > 10\% \quad (2)$$

Additionally, indoor and outdoor concentrations were plotted in a simple line graph to allow for visual inspection of these data to ensure that indoor generated peaks were removed. Although the rising edges of identified indoor peaks were censored, decaying portions were retained as they provided information on particle decay rates (Allen et al., 2003).

Multiple linear regression was used to estimate values for the coefficient representing penetration of outdoor particles (a_1) (the penetration parameter) and the coefficient representing decay of indoor particles (a_2) (the decay parameter by a recursive mass balance model) (Switzer and Ott, 1992) (Eq. (3)). The S_{in} term represents the indoor generation component, which was removed in the previous censoring step. The F_{inf} (Eq. (4)) and 95% confidence intervals (Eq. (5)) were then calculated for each home for each sampling day:

$$C_{in} = a_1(C_{out})_t + a_2(C_{in})_{t-1} + S_{in} \quad (3)$$

$$F_{inf} = \frac{a_1}{(1-a_2)} \quad (4)$$

$$\text{standard error} = \frac{\sqrt{((\mathrm{SE}\,a_1)^2 \times n \times (\frac{1}{(1-a_2)}))^2 + ((\frac{a_1}{(1-a_2)^2})^2 \times (\mathrm{SE}\,a_2)^2 \times n)}}{\sqrt{n} \times 1.96} \quad (5)$$

F_{inf} factors were then used to calculate air cleaner effectiveness for each home:

$$\mathrm{ACE} = \frac{F_{inf}(\text{no filter}) - F_{inf}(\text{filter})}{F_{inf}(\text{no filter})} \times 100 \quad (6)$$

All analysis was conducted using Microsoft EXCEL.

Levoglucosan Analysis
To verify the presence of biomass combustion during the study periods, all filters were analyzed for levoglucosan, a product of cellulose combustion, that is frequently used as a tracer for wood smoke (Nolte et al., 2001). Filters were analyzed by a modified GC/MS method (Simpson et al., 2004) in which the main modifications included the use of a deuterium-labeled d7-levoglucosan instead of tri-isopropyl benzene as an internal standard and a shorter ultrasonication time of 30 min compared to 1 h. The method had a limit of detection of 0.1 μg/filter, corresponding to 0.01 $\mu g/m^3$ levoglucosan for a 24-h sample, assuming a mean sample volume of 12.5 m^3.

Modeling
The following housing characteristics were considered potentially significant in explaining the variability seen in F_{inf} and air cleaner effectiveness (ACE) values: age of home, square footage of home, number of windows, percentage of carpeting in home, type of stove, use of range hood, use of air conditioning, type of heating system, use of fireplaces, and use of windows. Analysis was performed separately on summer and winter data and due to the small sample size, was also combined with the creation of a "season" variable. These variables were all included in the initial stages of modeling and then removed sequentially if variables were highly correlated or indicated univariate regressions with $P>0.25$. All analysis was conducted using Stata statistical software (Stata 9, StataCorp LP, Texas, USA).

Sensitivity Analysis
Previous research has raised issues about the use of the pDR for ambient air sampling due to impacts of high relative humidity (RH) and the occurrence of baseline drift in instrument response (Wu et al., 2005). Specifically, the pDR has been found to provide high readings under conditions of RH > 85% (Quintana et al., 2000; Wu et al., 2005). PM$_{2.5}$ concentration data points collected at > 85% RH were removed and F_{inf} and ACE were re-calculated and compared to original data to investigate the effect of high RH on monitor performance. Negative drift occurs when the pDR underestimates the true concentration of PM due to zeroing errors and can be identified when the time weighed average concentration (TWAC$_{man}$) calculated manually from the data output is not equal to the TWAC$_{pDR}$ recorded by the instrument (Wu et al., 2005). Since such data cannot be corrected due to the uncertainty of the specific data points experiencing negative drift, such data were removed from the analysis (Wu et al., 2005). As with high-RH data, F_{inf} and ACE values were calculated with and without baseline drift excluded data to investigate the effect of negative drift on monitor performance.

Results

Valid samples were collected for 19 homes in winter and 13 homes in summer. Date from two homes were removed due to low correlations (<0.40) and high relative differences (>1.0) between monitors during pre-mid and mid-post test periods. Four days of data with <24 h of sampling were also removed. Collocated pDR and HI$_{2.5}$ data showed strong relationships with R^2 values of 0.85 and 0.78 for winter and summer data, respectively. Outdoor PM$_{2.5}$ concentrations are summarized in Table 1.

Levoglucosan Analysis
Mean levoglucosan concentrations of 0.40 ± 0.30 and $0.10 \pm 0.10\ \mu g/m^3$ were found for winter and summer sampling periods, respectively. These values indicate that higher levels of PM$_{2.5}$ resulting from wood burning in the winter were collected than PM$_{2.5}$ resulting from forest fires in the summer. Regression analysis showed that similar relationships between PM$_{2.5}$ and levoglucosan concentration

Table 1. Summary of 30 min outdoor HI-corrected[a] PM$_{2.5}$ concentrations measured on days with and without filter placed in air cleaner for summer, winter and both seasons combined.

Season	HEPA filter	N	Indoor PM$_{2.5}$ concentration ($\mu g/m^3$)			Outdoor PM$_{2.5}$ concentration ($\mu g/m^3$)		
			Mean ± SD	GM	Range	Mean ± SD	GM	Range
Summer	Yes	429	4.9 ± 1.6	4.7	3.3–11.7	11.4 ± 10.0	9.5	3.5–90.8
	No	574	8.2 ± 5.0	7.3	3.3–69.3	10.6 ± 6.8	9.5	3.3–54.9
Winter	Yes	1103	3.9 ± 8.6	2.5	<1.0–186.8	18.7 ± 19.4	10.7	<4.2–189.2
	No	988	5.8 ± 7.0	4.1	<1.0–74.9	16.2 ± 14.2	11.6	<4.2–88.2
Both	Yes	1532	4.2 ± 7.3	3.0	<1.0–186.8	16.6 ± 17.5	10.8	<4.2–189.2
	No	1562	6.7 ± 20.7	5.1	<1.0–74.9	14.3 ± 12.3	10.7	<4.2–88.2

GM = geometric mean.
[a]pDR data corrected using regression relationships between summer and winter pDR and HI collocated data. N refers to number of 30-min averages. The HI detection limit, defined as three times the SD of the field blanks divided by the mean sample volume was 4.2 $\mu g/m^3$. The lower limit of the pDR measurement range provided by the manufacturer was 1 $\mu g/m^3$.

existed for both seasons, with slopes of 0.33 and 0.25 for winter ($R^2 = 0.35$) and summer ($R^2 = 0.40$), respectively, for HI data. Similar regression relationships were found between pDR and levoglucosan measurements.

Infiltration

Mean infiltration during summer for days without HEPA filters in place was 0.61, indicating that a substantial proportion of outdoor particles remained suspended indoors. During winter, when windows were not opened as frequently, the mean infiltration was 0.28, indicating a substantial reduction in the concentration of outdoor generated particles indoors, relative to outdoors. Significantly, higher infiltration was measured for the summer season ($P < 0.05$) with some values exceeding 1.0 for days when filters were not in place (Figure 1). With the removal of all particles of indoor origin, F_{inf} values > 1 represent $> 100\%$ outdoor particle penetration. As this is not physically possible, F_{inf} values > 1.0 indicate the inability of our algorithm to fully remove indoor generated particles or alternatively may result from imprecision in the modeling.

Figure 1 shows an example plot of indoor and outdoor PM concentrations for one of the study residences on days with and without the HEPA filter in place. In both seasons, as well as for data pooled across seasons, significantly lower mean F_{inf} were measured when filters were in place ($P < 0.05$, Table 2). In summer, the use of air cleaners resulted in an overall mean infiltration of 19% while in winter on average only 10% of outdoor generated particles remained suspended indoors. In addition, infiltration was lower on days when filters were in place for 23 out of 26 homes (Figure 2).

Air Cleaner Efficiency

No significant differences in ACE were found between summer and winter ($P > 0.10$). In 3 of 26 homes, we estimated negative efficiencies as a result of no calculated reduction in infiltration on the day in which filters were in place. These three homes all had relatively low F_{inf} (< 0.4).

Modeling

Data on categorical (Table 3) and continuous variables were collected. The age of sampled homes ranged from 5 to 60 years, floor area ranged from 640 to 4330 ft^2, air exchange rates ranged from 0.40 to 0.69 h^{-1}, the number of windows ranged from 4 to 21, and the degree of carpeting ranged from 0 to 93% of the entire household. Only air exchange rate significantly differed between homes sampled in summer (mean, SD: 0.26, 0.20) and winter (mean, SD: 0.14, 0.06) seasons. No housing characteristics could significantly explain the variability seen in F_{inf} values for the winter or summer seasons. When seasonal data were combined, increasing number of windows and the summer season were significantly related to increased infiltration ($R^2 = 0.41$, $P < 0.0001$). This result is consistent with F_{inf} calculations for our work, which showed lower infiltration for homes sampled in winter for both filter and no filter days. An increase in infiltration with the number of windows in a home is reasonable as more windows may lead to greater air exchange rates due to leakage. No variables were significantly associated with ACE for winter, summer or combined seasonal data suggesting that the use of air cleaners may be effective in most homes, regardless of the characteristics of the home.

Sensitivity Analysis

For summer samples, 52% of these data consisted of at least one 1-min average data point collected at an RH > 85%. For affected homes, an average of 24% of these data were removed in this sensitivity analysis. For winter, 38% of these data consisted of at least one 1-min average data point

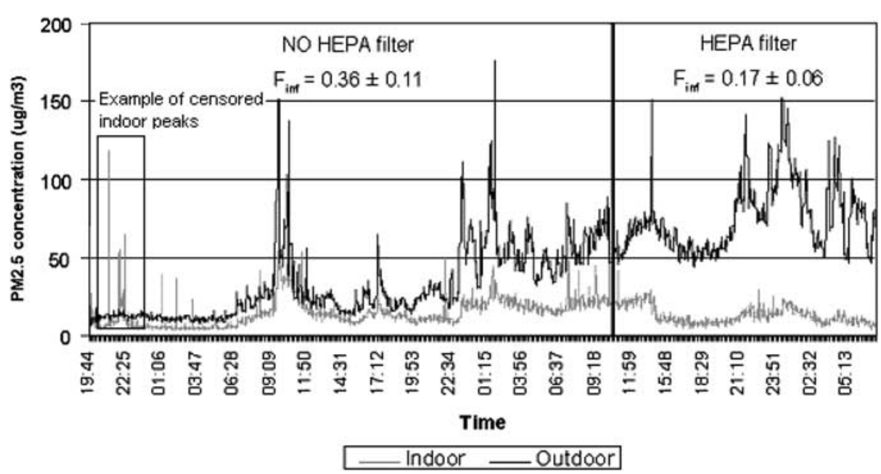

Figure 1. Indoor and outdoor particles with an aerodynamic diameter less than 2.5 micrometers ($PM_{2.5}$) sampling conducted at a home for 2 days and calculated infiltration (F_{inf}) values for both days (with and without high-efficiency particulate air (HEPA) filter). Personal DataRAM (pDR) concentrations are not corrected based on colocated HI sampling and gravimetric analysis.

Table 2. Summary of infiltration factors (F_{inf}) (unitless) and air cleaner efficiency (ACE) (%) values calculated for homes sampled in winter and summer comparing days with and without HEPA filter placed in air cleaner.

Season	HEPA filter	No. of samples	F_{inf}		ACE (%)	
			Mean (SD)	Range	Mean (SD)	Range
Summer	Yes	10	0.19 (0.20)	0.01–0.61	64.5 (35.0)	−3.2 to 98.9
	No	13	0.61 (0.27)	0.30–1.10		
Winter	Yes	19	0.10 (0.08)	0.01–0.30	54.5 (37.6)	−25 to 98.5
	No	16	0.28 (0.18)	0.10–0.68		
Both	Yes	29	0.13 (0.14)	0.01–0.61	57.7 (36.3)	−25 to 98.9
	No	29	0.42 (0.27)	0.10–1.10		

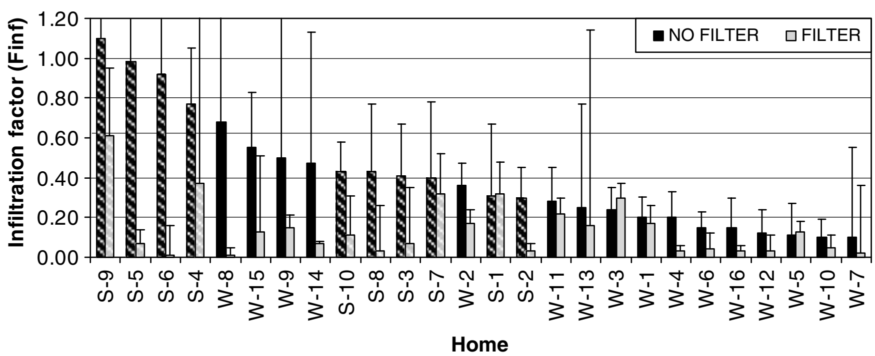

Figure 2. Infiltration factors (mean + SE F_{inf}) calculated for each home in summer (dashed bars) and winter (solid bars) for filter and no filter days.

Table 3. Summary of categorical housing variables included in F_{inf} and ACE modeling.

Variable	Category	Winter (*N*)	Summer (*N*)
Season	Winter/summer	16	13
Stove type	Electric	15	3
	Wood + gas	1	10
Range hood use	Never	6	3
	Sometimes	8	4
	Always	2	5
Air conditioning use	No	16	9
	Yes	0	4
Heating system	Gas + wood + furnace	14	10
	Gas + electric	2	3
Fireplace use	Never	13	13
	Sometimes	3	0
Windows open	Never	12	0
	Sometimes	2	9
	Always	2	4

N refers to the number of homes in each category.

collected at RH > 85% with a mean value of 29% of these data removed for this analysis. There were no significant differences in F_{inf} values calculated without excluding points with RH > 85%. After applying this exclusion, mean differences in F_{inf} were 0.04 and 0.02 for summer filter and no filter days, respectively. For winter, differences of 0.01 and 0.02 were found for filter and no filter days. Additionally, there were no significant differences in ACE values calculated after high RH data were removed, with mean differences of 12.2% and 2.0% for winter and summer samples, respectively. These results indicate that high-RH data did not significantly affect the main study findings.

A total of 2 days in the summer (10% of data) and 2 days in the winter (6% of data) displayed baseline drift. For summer data, baseline drift values of 2 and 5 $\mu g/m^3$ were found, while the average baseline drift for winter data was 2 $\mu g/m^3$. Small differences existed between the two data sets. The F_{inf} values calculated for "all data" and "baseline drift removed data" were 0.19 and 0.21, respectively, for filter days, and 0.60 and 0.63, respectively, for no filter days. Even smaller differences were found for winter sampling. Overall no significant differences in F_{inf} values were calculated after baseline drift data were censored and ACE values for either season affected. These results indicate that baseline drift was not an important factor in pDR operation for this data set.

Discussion

Average F_{inf} values of 0.61 ± 0.27 and 0.27 ± 0.18 and were found in homes sampled in summer and winter periods, respectively. These F_{inf} values suggest that for summer, the home does not provide optimum protection against exposure to outdoor-generated $PM_{2.5}$ exposure. Although, in the winter, approximately 30% of outdoor $PM_{2.5}$ infiltrated indoors, in the summer this was true of over 60% of particles generated from forest fires. High variability in infiltration was observed across homes in both seasons indicating that for some homes very little protection may in fact be offered to residents remaining indoors during periods of high outdoor-PM levels.

Lower variability and lower mean F_{inf} were observed in winter homes in a community impacted by residential wood smoke, suggesting a more protective effect of remaining indoors during this season. Two other studies using the recursive model to calculate infiltration of $PM_{2.5}$ into homes found somewhat higher infiltration rates than in our study. As in our work, Allen et al. (2003) calculated a higher F_{inf} for the nonheating season (March to September), with a mean value of 0.79 ± 0.18 compared to a mean value of 0.53 ± 0.61 for the heating season (October to February) in Seattle homes. Wu et al. (2006) used the recursive model to measure infiltration of $PM_{2.5}$ from agricultural burning into homes. This study, conducted in 23 homes in Washington from September to November, reported a mean F_{inf} value of 0.62 ± 0.16.

Our observations of somewhat lower F_{inf} values may be explained by differences in housing characteristics associated with local climatological factors resulting from geographical differences. All winter sampling was conducted in a northern community where temperatures typically reach $-12°C$ in the winter. Consequently, homes in this cold climate are more thoroughly insulated than in more temperate locations. For the homes sampled during summer, lower F_{inf} values may be indicative of residents closing windows in an attempt to reduce smoke entering the home during forest fire events. The lower infiltration measured in this study relative to others was also supported by our estimation of air-exchange rates that are at the low end of distributions computed for large residential samples in the US (Murray and Burmaster, 1995; Pandian et al., 1998; Chan et al., 2005).

In both seasons, infiltration was lowered with air cleaner use. F_{inf} of 0.10 ± 0.20 and 0.19 ± 0.20 were found for days when the air cleaner was run with the HEPA filter for winter and summer sampling periods, respectively. Infiltration was decreased with air cleaner use for 9 out of 10 homes sampled in the summer with F_{inf} values ranging from 0.01 to 0.61. Infiltration was decreased in 14 out of 16 homes sampled in the winter with F_{inf} values ranging from 0.01 to 0.30. Air cleaner efficiency calculations showed that air cleaners were effective at decreasing infiltration for nearly all homes during both winter and summer sampling periods. Mean ACE values of $65 \pm 35\%$ and $55 \pm 38\%$ were calculated for summer and winter sampling periods, respectively, and no significant difference was found between the seasons. This indicates that, on average, air cleaners were effective in both seasons, and therefore their use can be recommended in winter and summer.

It is unclear why the air cleaner was not effective in three homes where F_{inf} was not lowered with air cleaner use. Since the conditions of the home were not controlled for, a change in air exchange, or other factors including measurement error or unrecorded (by residents) changes in air cleaner operation (being turned off or set to a lower setting for a portion of the sampling period), may have contributed to these findings.

As this study only evaluated HEPA filter air cleaners it was not possible to extrapolate these findings to air cleaners that operate via different techniques. To maximize the potential exposure reduction provided by HEPA filter room air cleaners during smoke episodes, specific air cleaners should be appropriately matched to room sizes, as specified by manufacturers, and placed where occupants spend the majority of their time.

No previous studies have assessed ACE by measuring decreases in F_{inf} and therefore may not have fully accounted for the impact of indoor generated PM or considered the dynamic nature of indoor and outdoor PM concentrations in occupied homes during biomass combustion events. Henderson et al. (2005) investigated indoor $PM_{2.5}$ concentrations in four homes with and without air cleaners. It was found that indoor concentrations were lowered by 63–88% in unoccupied homes with three electrostatic air cleaners compared to unoccupied homes with no air cleaners during forest fire smoke events. Although operation of air cleaners was shown to decrease indoor $PM_{2.5}$ concentrations, a similar efficiency would not be expected when the home was occupied and indoor sources increased PM within the home. Mott et al. (2002) also reported that air cleaners were useful during periods of high PM due to forest fires. No exposure measurements were made in the study, but use of HEPA filter air cleaners was associated with decreased reporting of respiratory symptoms by affected residents. Longer use of the air cleaner was also associated with decreased symptom reporting.

Strengths, Limitations, and Recommendations

A major strength of this work is that all sampling was conducted in homes during forest fire and residential wood burning smoke events and not in unoccupied homes or in laboratory settings. This was possible due to the continuous $PM_{2.5}$ concentration sampling allowing for the calculation of infiltration in homes after accounting for indoor-generated PM. Due to the complex composition of PM, it is not possible to attribute any sample of particulate matter solely to one source, even if sampling is conducted during periods of wood or forest fire smoke. Collection of filter samples and analysis of levoglucosan, however, allowed us to verify that a large portion of collected $PM_{2.5}$ was due to summer forest fires or winter wood smoke. Although we had targeted a larger number of homes for sampling during summer, the sporadic nature of forest fires made sampling logistically difficult. To our knowledge, these measurements constitute the largest number of indoor samples collected during forest fire events, although homes and subjects were enumerated voluntarily and likely not representative of the general housing stock or general population. Further, due to the typically short duration of forest fire events, a shorter sampling time of 48–60 h was used to estimate infiltration compared to other studies of infiltration in which sampling times ranged from 24 h to 12-month periods (Abt et al., 2000; Long et al., 2001; Wallace et al., 2005). For wintertime sampling, because residential wood burning is more predictable and consistent, a longer sampling time would be possible.

Although this work, as well as the study conducted by Henderson et al. (2005) have shown that indoor $PM_{2.5}$ concentrations as well as infiltration decrease with the use of an air cleaner, questions as to their overall effectiveness in reducing health impacts still remain. Short-term elevations in $PM_{2.5}$ concentrations have been associated with adverse health impacts including myocardial infarction (Peters et al., 2001; Zanobetti and Schwartz, 2005), and, therefore, simultaneous assessment of infiltration and health impacts could greatly advance our work, although application to unpredictable forest fire events would be extremely difficult.

Conclusions

Our findings regarding infiltration and air cleaner effectiveness have important policy implications. For summer, high F_{inf} values indicated that remaining indoors was unlikely to be very protective of exposure to outdoor PM generated in vegetation fires. In winter, infiltration was found to be low, especially when compared to values found in the literature, and indicates that, in colder climates, remaining indoors offers substantial protection in winter for most homes. No variables could significantly explain variability in F_{inf} values for winter and summer which limited the ability to make general public health statements on remaining indoors during smoke events. Air cleaners were found to be effective in both summer and winter across a wide range of housing characteristics, as indicated by the lack of variables able to significantly explain variability in ACE values. This is an important finding for community members concerned about their exposure during times of high $PM_{2.5}$ concentration as our results indicate that the use of HEPA filter air cleaners can dramatically reduce indoor concentrations across different homes.

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References

Abt E., Suh H.H., Catalano P., and Koutrakis P. Relative contribution of outdoor and indoor particle sources to indoor concentrations. *Environ Sci Tech* 2000: 34: 3579–3587.

Aditama T.Y. Impact of haze from forest fire to respiratory health: Indonesian experience. *Respirology* 2000: 5: 169–174.

Allen R., Larson T., Sheppard L., Wallace L., and Lui L.-J.S. Use of real-time light scattering data to estimate the contribution of infiltrated and indoor-generated particles to indoor air. *Environ Sci Tech* 2003: 37: 3484–3492.

British Columbia Lung Association. BC state of the air report, 2006. Accessed at http://www.bc.lung.ca/pdf/bcstate_of_the_air_report2006.pdf January 15 2007.

Chan W.R., Nazaroff W.W., Price P.N., Sohn M.D., and Gadgil A.J. Analyzing a database of residential air leakage in the United States. *Atmos Environ* 2005: 39: 3445–3455.

Duclos P., Sanderson L.M., and Lipsett M. The 1987 forest fire disaster in California: assessment of emergency room visits. *Arch Environ Health* 1990: 45: 53–58.

Emmanuel S.C. Impact to lung health of haze from forest fires: the Singapore experience. *Respirology* 2000: 5: 175–182.

Environment Canada. Main emission sources, particulate matter, 2006. Accessed at http://www.ec.gc.ca/cleanair-airpur/Main_Emission_Sources-WS9A24D6D1-1_En.html September 19 2006.

Fischer G., and Schrattenholzer L. Global bioenergy potentials through 2050. *Biomass Bioenergy* 2001: 20: 151–159.

Flannigan M.D., Stocks B.J., and Wotton B.M. Climate change and forest fires. *Sci Total Environ* 2000: 262: 221–229.

Henderson D.E., Milford J.B., and Miller S.L. Prescribed burns and wildfires in Colorado: impacts of mitigation measures on indoor air particulate matter. *J Air Waste Manag Assoc* 2005: 55: 1516–1526.

Houck J.E., Tiegs P.E., McCrillis R.C., Keithley C., and Crouch J. Air emissions from residential heating: the wood heating option put into environmental perspective. *Proceedings of US EPA and Air and Waste Management Association Conference: Emission Inventory: Living in a Global Environment*, New Orleans, LA. December, 1998 Air and Waste Management Association, Pittsburgh, PA, 1998: 1: 373–384.

Koenig J.Q., Larson T.V., Hanley Q.S., Rebolledo V., Dumler K., Checkoway H., et al. Pulmonary function changes in children associated with fine particulate matter. *Environ Res* 1993: 63: 26–38.

Kunzli N., Avol E., Wu J., Gauderman W.J., Rappaport E., and Millstein J. Health effects of the 2003 southern California wildfire on children. *Am J Respir Crit Care Med* 2006: 174: 1221–1228.

Lai A.C. Particle deposition indoors: a review. *Indoor Air* 2002: 12: 211–214.

Larson T.V., and Koenig J.Q. Wood smoke: emissions and non-cancer respiratory effects. *Annu Rev Public Health* 1994: 15: 133–156.

Long C.M., Suh H.H., Catalano P.J., and Koutrakis P. Using time- and size-resolved particulate data to quantify indoor penetration and deposition behavior. *Environ Sci Tech* 2001: 35: 2089–2099.

Moore D., Copes R., Fisk R., Joy R., Chan K., and Brauer M. Population health effects of air quality changes due to forest fires in British Columbia in 2003: estimates from physician-visit billing data. *Can J Public Health* 2006: 97: 105–108.

Mott J.A., Meyer P., Mannino D., Redd S.C., Smith E.M., Gotway-Crawford C., et al. Wildland forest fire smoke: health effects and intervention evaluation, Hoopa, California, 1999. *West J Med* 2002: 176: 157–162.

Murray D.M., and Burmaster D.E. Residential air exchange rates in the United States: empirical and estimated parametric distributions by season and climatic region. *Risk Anal* 1995: 15: 459–465.

Naeher L.P., Brauer M., Lipsett M., Zelikoff J.T., Simpson C.D., Koenig J.Q., et al. Woodsmoke health effects: a review. *Inhalation Toxicol* 2007: 19: 67–106.

Nolte C.G., Schauer J.J., Cass G.R., and Simoneit B.R. Highly polar organic compounds present in wood smoke and in the ambient atmosphere. *Environ Sci Tech* 2001: 35: 1912–1919.

Norris G., Larson T., Koenig J., Claiborn C., Sheppard L., and Finn D. Asthma aggravation, combustion, and stagnant air. *Thorax* 2000: 55: 466–470.

Norris G., YoungPong S.N., Koenig J.Q., Larson T.V., Sheppard L., and Stout J.W. An association between fine particles and asthma emergency department visits for children in Seattle. *Environ Health Perspect* 1999: 107: 489–493.

Office of Air and Radiation. United States Environmental Protection Agency, 2006. Residential air cleaning devices: a summary of available information. Accessed at http://www.epa.gov/iaq/pubs/residair.html#standard%20for%20 portables November 04 2006.

Pandian M.D., Behar J.V., Ott W.R., Wallace L.A., Wilson A.L., Colome S.D., and Koontz M.D. Correcting errors in the nationwide data base of residential air exchange rates. *J Exp Anal Environ Epidemiol* 1998: 8: 577–586.

Peters A., Dockery D.W., Muller J.E., and Mittleman M.A. Increased particulate air pollution and the triggering of myocardial infarction. *Circulation* 2001: 103: 2810–2815.

Quintana P.J., Samimi B.S., Kleinman M.T., Liu L.J., Soto K., Warner G.Y., et al. Evaluation of a real-time passive personal particle monitor in fixed site residential indoor and ambient measurements. *J Expo Anal Environ Epidemiol* 2000: 10: 437–445.

Rittmaster R., Adamowicz W.L., Amiro B., and Pelletier R.T. Economic analysis of health effects from forest fires. *Can J For Res* 2006: 36: 868–877.

Sapkota A., Symons J.M., Kleissl J., Wang L., Parlange M.B., and Ondov J. Impact of the 2002 Canadian forest fires on particulate matter air quality in Baltimore city. *Environ Sci Tech* 2005: 39: 24–32.

Simpson C.D., Dills R.L., Katz B.S., and Kalman D.A. Determination of levoglucosan in atmospheric fine particulate matter. *J Air Waste Manag Assoc* 2004: 54: 689–694.

Switzer P., and Ott W. Derivation of an indoor air averaging time model from the mass balance equation for the case of independent source inputs and fixed air exchange rates. *J Exp Anal Environ Epidemiol* 1992: 2(suppl): S113–S135.

United States Environmental Protection Agency. How smoke from fires can affect your health, 2003. Accessed at http://depts.washington.edu/wildfire/resources/ EPA2003.pdf June 23 2006.

Wallace L., and Williams R. Use of personal-indoor-outdoor sulfur concentrations to estimate the infiltration factor and outdoor exposure factor for individual homes and persons. *Environ Sci Tech* 2005: 39: 1707–1714.

Wu C.F., Delfino R.J., Floro J.N., Samimi B.S., Quintana P.J., Kleinman M.T., et al. Evaluation and quality control of personal nephelometers in indoor, outdoor and personal environments. *J Exp Anal Environ Epidemiol* 2005: 15: 99–110.

Wu C.H., Jimenez J., Claiborn C., Gould T., Simpson C.D., Larson T., Lui L.-J.S., et al. Agricultural burning smoke in eastern Washington: Part II. Exposure assessment. *Atmos Environ* 2006: 40: 5379–5392.

Zanobetti A., and Schwartz J. The effect of particulate air pollution on emergency admissions for myocardial infarction: a multi-city case-crossover analysis. *Environ Health Perspect* 2005: 113: 978–982.