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The impact of a community–wide woodstove changeout intervention on air quality within two schools

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ABSTRACT

Due to temperature inversions and widespread residential woodstove use, Libby, Montana historically experienced elevated levels of ambient woodsmoke $PM_{2.5}$ throughout the winter months. In an effort to reduce wintertime $PM_{2.5}$, a large community–wide woodstove changeout was conducted between 2005 and 2007, removing nearly 1 200 old polluting stoves from service. To determine the impact of this intervention on indoor air quality, $PM_{2.5}$ sampling was conducted in the gymnasiums of an elementary and middle school before, during, and after the woodstove changeout over a four–year period. Throughout the program, results showed that indoor $PM_{2.5}$ concentrations at the elementary school were moderately high regardless of year or season (mean±sd, 31.9 ± 14.1 $\mu g/m^3$), ranging from 11.0 $\mu g/m^3$ to 79.3 $\mu g/m^3$. At the middle school, the mean was 12.2 ± 11.2 $\mu g/m^3$, with no differences by season. Although there was an overall improvement in ambient air quality (and reduction of woodsmoke–$PM_{2.5}$) when comparing pre– and post–changeout $PM_{2.5}$ concentrations, results suggest that the community–wide woodstove changeout did not have a significant impact on indoor air quality within the gymnasiums over this same time period. These findings are supported by the results of selected chemical markers of woodsmoke measured from indoor PM (including levoglucosan) at both schools, which also demonstrated no significant reductions throughout the four–year sampling program.

Keywords: $PM_{2.5}$, smoke, woodstove, school, gymnasium

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1. Introduction

Studies conducted throughout the United States (Sexton et al., 1984; Fairley, 1990; McDonald et al., 2000; Schauer and Cass, 2000; Fine et al., 2001; Polissar et al., 2001; Maykut et al., 2003; Larson et al., 2004; Chow et al., 2007) and world (McGowan et al., 2002; Luhar et al., 2006; Puxbaum et al., 2007; Alfarra et al., 2007; Szidat et al., 2007; Lanz et al., 2008) have identified woodsmoke as a major component of ambient particulate matter. This is especially true in valley locations located throughout the northern Rocky Mountains of western Montana, where $PM_{2.5}$ from woodstoves have been shown to be the predominant source of $PM_{2.5}$ throughout the winter months (Ward and Lange, 2010).

Libby is a small mountain valley community in northwestern Montana (Lincoln County) with a population of approximately 2 600. Temperature inversions are common throughout the winter months, contributing to elevated levels of ambient $PM_{2.5}$. Before a community–wide woodstove changeout was implemented, the winter $PM_{2.5}$ concentrations were so high that Libby exceeded the annual $PM_{2.5}$ National Ambient Air Quality Standard (NAAQS) of 15 $\mu g/m^3$, resulting in Libby being designated as a nonattainment area for the fine fraction. With the exception of some areas of southern California, Libby was the only $PM_{2.5}$ nonattainment area in the mid and western states prior to the revised 24–hour $PM_{2.5}$ NAAQS in 2007.

Residential woodstoves are a common source of home heating in many areas of the Northern Rocky Mountains, and many other areas throughout the world, and remain a cheap alternative to burning fossil fuels. This is especially true in Libby, where upwards of 80% of the wintertime ambient $PM_{2.5}$ came from residential woodsmoke (Ward et al., 2006). In a 2005 emission inventory conducted by the Montana Department of Environmental Quality, ~2 264 wood burning devices were identified within Libby. This included fireplaces (no insert), pre–certified woodstoves, EPA catalytic certified stoves, EPA non–catalytic stoves, cord wood furnaces, masonry furnaces, and pellet stoves/inserts (Carlin, 2008).

In an effort to lower wintertime ambient $PM_{2.5}$, woodstove emissions were targeted through a community wide woodstove changeout program. From 2005 through 2007, nearly 1 200 old woodstoves were changed out, modified, or surrendered (Eagle and Houck, 2007a; Eagle and Houck, 2007b). Although the woodstove changeout improved ambient air quality in Libby by lowering the woodsmoke–related $PM_{2.5}$ during the winter months (Ward et al., 2010), perhaps a more important question is what impact the changeout had on indoor air quality. Most people spend the majority of their time indoors (Fishbein and Henry, 1991; Jenkins et al., 1992), as much as 95% in some areas. When considering children, they spend at least a third of their total time inside school buildings (ISIAQ, 2001), and up to several hours per week within the school gymnasiums exercising.

To expand our investigation into the effectiveness of the woodstove changeout beyond ambient air quality and residential indoor environments, the overall goal of this sampling program

was to quantify the potential indoor air quality improvements within two Libby–area schools. To this end, we present results from a PM$_{2.5}$ air sampling program conducted in the gymnasiums of an elementary and middle school before, during, and after the woodstove changeout over a four–year period.

2. Experimental

2.1. Woodstove changeout program

From June 2005 through June 2007, a community–wide woodstove changeout was conducted in Libby, Montana. In this context, changeout refers to the removal of older, high–emitting woodstoves and replacement with U.S. EPA–certified woodstoves that meet PM$_{2.5}$ emissions standards of less than 7.5 g/h. The conventional model woodstoves utilize firebox insulation, a longer, hotter gas flow path, and pre–heated combustion air to yield more complete combustion. Other residences chose not to receive a new woodstove, and instead opted for the following heating appliance types: gas stoves/heaters/furnaces, wood inserts, pellet stoves, pellet inserts, pellet furnaces, oil stoves/furnaces, electric heaters, and wood furnaces. At the conclusion of the woodstove changeout program in 2007, nearly 1 200 old woodstoves were changed out, modified, or surrendered in an effort to lower the ambient PM$_{2.5}$ during the winter heating season (Eagle and Houck, 2007a; Eagle and Houck, 2007b). As a result of this community–wide intervention, ambient wintertime PM$_{2.5}$ concentrations were reduced by ~25–30% (Bergauff et al., 2009; Ward et al., 2010).

2.2. Periods of air sampling

Starting in January 2006, PM$_{2.5}$ samples were collected during the winter (January through March), spring (May), and fall (September through October) seasons within two Libby schools throughout the duration of the woodstove changeout. In this school sampling program, please note that the winter of 2005/2006 is considered the baseline winter of the woodstove changeout program, while the winter of 2008/2009 is considered the first winter after the changeout program was completed.

2.3. Sampling sites

PM$_{2.5}$ samples were simultaneously collected within two school gymnasiums throughout the duration of the woodstove changeout, with Figure 1 showing the locations of these schools within Libby. During the periods of sampling (winter 2005/2006

through winter 2008/2009), the elementary school had approximately 400 students in grades pre–kindergarten through 4th grade, while the middle school had approximately 600 students in grades 5–8. Sampling sites within the two school gymnasiums were dictated by practical considerations (i.e., security of sampling equipment and access to power source) and consultation with school administrators. At the elementary school, the PM$_{2.5}$ sampling site was located within the school gymnasium (~300 m^2 in size) approximately 6.1 meters from an exterior door (the only exterior door in the entire gymnasium, which was kept closed the majority of the time), and in proximity to a storage room containing a copying machine. Samples within the elementary school gymnasium were collected at a height of approximately 1.5 meters above the ground.

At the middle school gymnasium (~790 m^2 in size), air samplers were placed on a balcony located approximately 4.6 meters above the gymnasium floor along a side wall approximately 61 meters from the nearest door. In total, there were eight exterior doors in the middle school gymnasium, with doors kept closed the majority of the time. The elementary school was built in 1953, and the middle school was built in 1970. As illustrated in Figure 1, the schools were located approximately 2.5 kilometers from one another.

Both gymnasiums were heated by electricity and mechanically ventilated using Class 2, 40P1, R12347ND air filters. The air–exchange rate for the elementary school gymnasium was 11 900 m^3/h and operated as needed. The air–exchange rate for the middle school gymnasium was 54 400 m^3/h, and operated 05:30 to 16:30 Monday through Friday. Both gymnasiums were heavily used throughout the weekdays and evenings, and sporadically used during the weekends for community recreational activities. Weekday activity patterns were consistent between both school gymnasiums, with ~25–50 students within the gymnasiums at any one time.

For comparison with indoor PM$_{2.5}$ values, ambient PM$_{2.5}$ data were collected on the roof of the Lincoln County Environmental Health Department in downtown Libby on the same days as school sampling. This is the primary PM$_{2.5}$ compliance monitoring site for the city of Libby, and is located approximately 1.6 kilometers from the elementary school, and 3.2 kilometers from the middle school (Figure 1). Daily temperature, wind speed, relative humidity, and precipitation data collected in Libby throughout the time period were obtained from an archived database (WRCC, 2010).
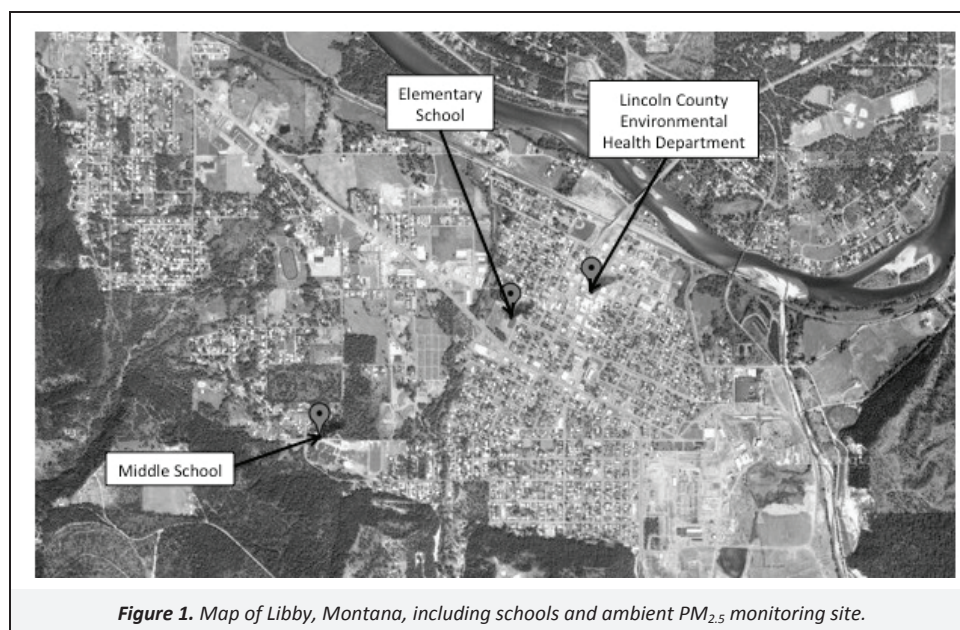

Figure 1. *Map of Libby, Montana, including schools and ambient PM$_{2.5}$ monitoring site.*

2.4. Sampling procedures

At each of the two indoor school sites, 24–hour samples were simultaneously collected approximately once per week (during the weekdays throughout the winter, spring, and fall seasons, respectively) using three individual $PM_{2.5}$ samplers. Two Leland Legacy pumps (SKC Inc., Eighty Four, PA, USA) were used to pull air sample through Personal Environmental Monitors (PEMs). One of the PEMs was fitted with a 37–mm $PM_{2.5}$ Teflon filter to collect information on the indoor $PM_{2.5}$ mass, while a second PEM was fitted with a pre–fired 37–mm quartz filter to quantify $PM_{2.5}$–assoicated levels of Organic Carbon (OC) and Elemental Carbon (EC). A 47–mm quartz filter was also collected during each event using a BGI cyclone (BGI, Inc., Waltham, MA, USA) for subsequent analyses of $PM_{2.5}$–associated chemical markers of woodsmoke such as levoglucosan. The flow rates were set at 10 Liters per Minute (LPM) for the Leland/PEM samplers and ~16.7 LPM for the BGI cyclone.

At the Lincoln County Environmental Health Department, continuous ambient $PM_{2.5}$ was measured with a MetOne BAM–1020 (MetOne Instruments Inc., Grants Pass, OR, USA) to investigate the relationship between ambient and indoor levels of $PM_{2.5}$ at each of the two schools. As the BAM data reported hourly $PM_{2.5}$ measurements, ambient hourly concentrations were averaged to match up with the 24–hour sampling events within each of the schools.

2.5. Analytical procedures

A gravimetric analysis was conducted on the 37–mm Teflon filters, while levels of OC and EC were measured from the 37–mm quartz filters by Thermal Optical Reflectance. Both analyses were conducted by a contracted laboratory (Chester LabNet, Tigard, OR, USA). From the 47–mm quartz filter, chemical markers of woodsmoke (levoglucosan, abietic acid, and dehydroabietic acid) were quantified. These compounds are all known chemical markers of biomass combustion, and were tracked based on their elevated concentrations measured in the ambient environment during a 2003/2004 Libby $PM_{2.5}$ source apportionment program (Ward et al., 2006).

The woodsmoke markers were analyzed at the University of Montana following a method described in Bergauff et al. (2008). This method was adapted from methods reported previously by Schauer et al. (2001) and Simpson et al. (2005). Briefly, half of each filter was spiked with deuterated recovery standards, placed in a vial, and extracted by ultrasonication using ethyl acetate containing 3.6 mM triethylamine. The extract was filtered, reduced in volume to approximately 500 μL and split into two equal fractions. One fraction was evaporated to dryness and derivatized with N–O–bis (trimethylsilyl) trifluoroacetamide, trimethylchloro-silane and trimethylsilylimidazole to convert the sugar anhydrides and the abietic acids to their trimethylsilyl derivatives. The second fraction was treated with a 2:3 mixture of acetic anhydride:triethyl amine to generate the acetate derivatives of the methoxyphenols. Both sample fractions were analyzed by GC/MS on a Hewlett–Packard GC/MSD (GC Model 6890, MSD Model 5973) using an HP–5MS capillary column or equivalent.

2.6. Data analysis and statistical procedures

In an effort to determine the impact of the woodstove changeout on indoor air quality, we have separated the sampling days into winter (wood burning season) and non–winter seasons (the non–burning seasons of fall and spring) for comparison. As mentioned previously, winter samples included sampling days in January through March. Non–winter samples were characterized by samples collected during fall (September through October) and spring (May). Non–detects was assigned a value of ½ the detection

limit for the corresponding analyses. Pearson product correlation coefficients were calculated for indoor $PM_{2.5}$, ambient $PM_{2.5}$, and meteorological measures on the corresponding sample days. For each school, differences in indoor $PM_{2.5}$ concentrations between the first year of the sampling program (winter 2005/2006) and the fourth year (winter 2008/2009) were evaluated by generalized linear models, adjusting for ambient $PM_{2.5}$ and meteorological variables as appropriate. Differences by season (winter versus non–winter) were also evaluated in a similar manner. All analyses were conducted using SAS v9.2 (Cary, NC).

2.7. Quality assurance/quality control (QA/QC)

A comprehensive QA/QC program was employed throughout the sampling program. Using a certified Bios DryCal (SKC Inc., Eighty Four, PA, USA) flow meter, the flow rate on the Leland pump/PEM was measured both before and after each sampling event, while a certified DeltaCal (BGI, Inc., Waltham, MA, USA) was used to measure the flow rates of the cyclone. Teflon and quartz filter field blanks were collected for approximately every 10 samples (10%) to address artifact contamination. Field personnel followed the recommended maintenance and cleaning schedules for the samplers as described in their respective manuals throughout the program.

Filters were always transported in coolers to and from the sampling sites. Clean Teflon and quartz filters were stored in a refrigerator at approximately 2 °C prior to sample collection. Following sample collection, the filter samples were stored in a freezer at –20 °C until analysis. Within the University of Montana laboratory, the QA/QC program for the woodsmoke marker analyses included the analysis of blank filters (one blank filter is analyzed for every 10 samples), spikes, instrument calibration checks, and routine instrument maintenance.

3. Results and Discussion

3.1. Ambient $PM_{2.5}$, indoor $PM_{2.5}$, and meteorology

Ambient characteristics on the school indoor sampling days are presented in Table 1. As anticipated, ambient $PM_{2.5}$ was higher on winter sampling days compared to non–winter sampling days. When we look specifically at the non–winter seasons, there was little difference in ambient $PM_{2.5}$ mean±sd concentrations when comparing fall sampling days, (8.0±5.0 μg/m^3) and spring sampling days (5.7±2.5 μg/m^3, p=0.15). Winter sampling days demonstrated lower temperature, less wind, and higher relative humidity, with no significant difference in precipitation compared to non–winter sampling days. Across all seasons, average ambient temperature was highly correlated with average wind speed (r=0.50, p<0.0001) and maximum wind gust (r=0.46, p<0.0001), and inversely correlated with relative humidity (r=–0.41, p<0.0001).

Ambient $PM_{2.5}$ sampling conducted on days corresponding to the scheduled indoor school sample days did not demonstrate a significant reduction across the winter years. When adjusted for ambient temperature, the difference (and 95% CI) in ambient $PM_{2.5}$ between the first winter 2005/2006 and the winter of 2008/2009 (winter following the completion of the changeout) was –2.0 μg/m^3 (–7.4, +3.4). Table 2 presents the correlations between $PM_{2.5}$ concentrations (including both schools and ambient) and the meteorological variables on the sampling days. Ambient $PM_{2.5}$ was inversely correlated with temperature, wind, and precipitation, while positively correlated with relative humidity. By contrast, indoor $PM_{2.5}$ at the elementary school was positively associated with temperature and wind, and inversely associated with relative humidity and precipitation. Indoor $PM_{2.5}$ at the elementary school was also inversely correlated with ambient $PM_{2.5}$. Results from the middle school showed no correlations between indoor $PM_{2.5}$ and ambient conditions.

***Table 1**. Average (sd) ambient characteristics on school sampling days*

School Year	Sample Days	PM$_{2.5}$ (µg/m^3)	Temp (°C)	Wind Speed (km/h)	Max Wind Gust (km/h)	Relative Humidity (%)	Precipitation (cm)
Winter							
All years	46	18.0 (6.8)	−0.68 (4.5)	0.6 (0.5)	7.8 (4.1)	77.7 (14.6)	0.7 (1.1)
2005/06	15	19.4 (5.7)	−0.03 (3.1)	0.6 (0.5)	8.3 (5.1)	70.1 (15.7)	0.5 (1.1)
2006/07	10	17.2 (7.7)	1.4 (2.6)	0.5 (0.9)	6.8 (4.5)	82.7 (18.0)	0.5 (1.1)
2007/08	11	16.1 (6.9)	−2.2 (6.2)	0.6 (0.4)	8.0 (2.7)	76.5 (9.6)	0.9 (1.3)
2008/09	10	18.8 (7.7)	−2.0 (5.1)	0.6 (0.4)	7.9 (3.7)	85.5 (8.1)	0.8 (1.2)
Non–Winter							
All years	35	6.9 (4.1)	11.2 (4.5)	1.0 (0.6)	11.5 (5.0)	67.2 (13.5)	1.0 (1.3)
2005/06	6	5.8 (1.7)	14.5 (4.4)	1.2 (0.2)	10.7 (1.7)	58.5 (9.1)	0.8 (1.3)
2006/07	12	9.2 (5.8)	10.9 (4.7)	0.9 (0.7)	11.0 (5.7)	66.8 (15.7)	1.1 (1.3)
2007/08	11	5.2 (1.8)	10.4 (3.9)	1.0 (0.6)	12.4 (4.8)	69.3 (14.2)	1.4 (1.3)
2008/09	6	6.2 (3.5)	10.1 (4.7)	0.9 (0.9)	11.8 (7.2)	73.2 (7.7)	0.4 (1.0)

Note: Winter samples included sampling days in January through March. Non-winter samples are characterized by samples collected during fall (September through October) and spring (May).

***Table 2.** Pearson correlation coefficients (p–value) for ambient characteristics and indoor PM$_{2.5}$ mass on indoor sample days*

	Ambient PM$_{2.5}$ (n=79)	Elementary School Indoor PM$_{2.5}$ (n=73)[a]	Middle School Indoor PM$_{2.5}$ (n=69)[b]
Ambient PM$_{2.5}$ (µg/m^3)		−0.252 (0.032)	0.053 (0.663)
Temperature (°C)	−0.687 (<0.001)	0.491 (<0.001)	0.056 (0.643)
Wind Speed (km/h)	−0.512 (<0.001)	0.301 (0.009)	−0.005 (0.969)
Wind Gust (km/h)	−0.542 (<0.001)	0.241 (0.038)	−0.084 (0.489)
Relative Humidity (%)	0.246 (0.029)	−0.426 (<0.001)	0.027 (0.821)
Precipitation (cm)	−0.433 (<0.001)	−0.186 (0.110)	−0.117 (0.332)

[a] n=75 for bivariate comparisons with meteorological variables.
[b] n=71 for bivariate comparisons with meteorological variables.

3.2. Elementary school

Table 3 presents the indoor results from the elementary school. Indoor PM$_{2.5}$ concentrations at the elementary school were moderately high regardless of year or season (mean±sd, 31.9±14.1 µg/m^3), with individual 24–hour sampling results ranging from 11.0 µg/m^3 to 79.3 µg/m^3. The proportion of Total Carbon (TC) comprised of the OC component ranged from 88.8% to 99.7%. Significant differences were observed between the first winter (2005/2006) and the fourth, post–changeout winter (2008/2009), but these differences did not change as expected (i.e. overall reduction across years), nor were the changes in a consistent direction across all years. EC was slightly higher during winter versus non–winter sampling days (p=0.025), but all other results did not demonstrate a strong seasonal response.

3.3. Middle school

Table 4 presents the indoor results for the middle school. Overall, indoor PM$_{2.5}$ concentrations were lower at this school compared to the elementary school. The mean±sd was 12.2±11.2 µg/m^3, and there were no differences by season. Despite the lower overall averages at this site, there were four sampling days with PM$_{2.5}$ concentrations above 30 µg/m^3, with one sample day yielding an average of 83.3 µg/m^3. The proportion of TC comprised from the OC component ranged from 95.0% to 99.9%. Middle school concentrations of PM$_{2.5}$ mass, OC, EC, and abietic acid were actually more elevated during the winter 2008/2009 compared to the initial winter of 2005/2006. Adjusting for year and ambient PM$_{2.5}$, indoor levoglucosan at the middle school was higher in the winter versus non–winter sampling days (p=0.004),

whereas the OC fraction was lower in winter versus non–winter sampling days (p<0.001).

3.4. Comparison with pre–changeout results (Winter 2004/2005)

During January through March 2005, we conducted PM sampling within Libby's elementary and middle schools to establish baseline indoor PM concentrations (in five distinct size fractions) before the start of the woodstove changeout program (Ward et al., 2007). During this winter 2004/2005 baseline study, PM$_{2.5}$ mass averaged 35.6 µg/m^3 at the elementary school and 6.9 µg/m^3 at the middle school. It is important to note that different sampling equipment was used in the baseline program (Sioutas 5–stage impactors) as was a different sampling location for the middle school site (inside the main school compared to inside the gymnasium in this study). Unfortunately, due to these fundamental differences in the sampling programs, we cannot use these baseline data for comparison with the during/post–changeout results presented in this manuscript. Therefore, we have used the winter 2005/2006 results to serve as our baseline winter for comparison with results collected during the winters of 2006/2007, 2007/2008, and 2008/2009.

3.5. Impact of the woodstove changeout on indoor air quality

We have previously reported on an overall successful reduction in wintertime ambient PM$_{2.5}$ concentrations (Ward et al., 2010) as well as indoor residential PM$_{2.5}$ concentrations (Ward et al., 2008; Noonan et al., 2012) as a result of the Libby woodstove changeout program. However, we did not see corresponding improvements in indoor air quality measured within the two

school gymnasiums investigated in this program. In addition to $PM_{2.5}$, we did not measure large reductions in levels of OC and EC when comparing the four winter periods. A more revealing finding is the results of the levoglucosan analyses, which are indicative of woodsmoke–related $PM_{2.5}$. At both the elementary and middle schools, there was no consistent reduction in levoglucosan throughout the four–year winter sampling program. These findings are not consistent with what was observed in the ambient air throughout the duration of the changeout, where there was a 50% reduction in levoglucosan when comparing the winters of 2004/2005 and 2007/2008 (Bergauff et al., 2009). There was a reduction in dehydroabietic acid at each of the schools, but this pattern was not mirrored by the levels of abietic acid, where levels actually increased throughout the program within the schools.

3.6. Comparison between schools, and with other school sampling programs

Throughout the four year program, $PM_{2.5}$ concentrations were much higher at the elementary school compared to the middle school. As noted earlier, there were physical differences between the two sampling sites. At the elementary school, the $PM_{2.5}$ sampling site collected air samples at a height of approximately 1.5 meters above the ground. At the middle school, air samplers were placed on a balcony located approximately 4.6 meters above the gym floor. It is unknown how this difference in sampling heights influenced results (if at all), and it is likely that the difference in measured $PM_{2.5}$ concentrations between schools was likely due to other factors. For example, these factors might include air exchange, ventilation conditions (Rojas–Bracho et al., 2000), and particle deposition (He et al., 2005). Indeed, the unit ventilators within the smaller elementary school gymnasium ran with much less frequency compared to the larger middle school gymnasium, partly explaining the higher levels of $PM_{2.5}$ within the elementary school gymnasium. Other things that may have

influenced concentrations of $PM_{2.5}$ inside the school gymnasiums include building design, number and age of occupants, their activities, and other sources inside the buildings. This includes a copying machine within the elementary gymnasium, as well as greater frequency/quantities of tracked–in dirt within the elementary school gymnasium compared to the middle school gymnasium.

The $PM_{2.5}$ levels reported in this Libby study can be compared to findings reported in other school sampling programs. In a study conducted in three elementary schools in Central and Southeast Ohio, $PM_{2.5}$ concentrations ranged from ~15–18 µg/m³ (John et al., 2007). In Munich, indoor air quality within 64 schools was evaluated. The median indoor $PM_{2.5}$ concentration during the winter was 37.0 µg/m³ (range of 4.3 to 73.1 µg/m³), with the median in summer of 22.1 µg/m³ (range of 9.8 and 55.1 µg/m³) (Fromme et al., 2006). In another study conducted in Germany, samples were collected within two classrooms for a period of six weeks. Median indoor $PM_{2.5}$ concentrations were 37.4 µg/m³ (Fromme et al., 2008). $PM_{2.5}$ concentrations measured in these programs were consistent with the concentrations measured throughout the Libby school sampling program.

Large indoor sources of $PM_{2.5}$ within the schools and infiltration of ambient particles to the indoor environment can lead to elevated indoor $PM_{2.5}$ concentrations. This partly explains the results measured in other school studies conducted across the world in places such as Delhi City, India (Goyal and Khare, 2009), Tehran, Iran (Halek et al., 2009), Antwerp, Belgium (Stranger et al., 2008), Istanbul, Turkey (Ekmekcioglu and Keskin, 2007), and Athens, Greece (Diapouli et al., 2008). Each of these studies reported much higher indoor school $PM_{2.5}$ levels compared to what was measured in this program.

Table 3. Mean (sd) for indoor $PM_{2.5}$ mass (µg/m³), and $PM_{2.5}$–associated OC/EC (µg/m³) and selected organics (ng/m³) at the elementary school by season and year

	Winter Sampling					Non–winter Sampling					Season[c]
	2005/06 (n=15)	2006/07 (n=7)	2007/08 (n=11)	2008/09 (n=10)	Year[a] p-value	2005/06 (n=6)	2006/07 (n=9)	2007/08 (n=11)	2008/09 (n=6)	Year[b] p-value	p–value (n=73)
$PM_{2.5}$	27.1 (5.8)	26.6 (16.4)	21.5 (8.5)	30.5 (11.3)	<0.001	25.6 (4.3)	52.7 (12.8)	36.1 (12.9)	39.2 (15.7)	0.012	0.378
Organic carbon	13.9 (1.3)	12.1 (1.3)	12.0 (1.3)	17.5 (4.2)	<0.001	11.6 (0.9)	16.7 (4.1)	13.2 (2.4)	16.2 (3.4)	0.002	0.160
Elemental carbon	0.4 (0.12)	0.5 (0.24)	0.4 (0.23)	1.0 (0.65)	<0.001	0.2 (0.04)	0.4 (0.14)	0.2 (0.11)	0.5 (0.22)	<0.001	0.025
Levoglucosan	264 (116)	163 (128)	706 (285)	410 (145)	0.551	107 (86.3)	74.2 (90.7)	422 (207)	167 (28.3)	0.871	0.709
Dehydroabietic acid	134 (35.9)	74.5 (21.4)	162 (39.6)	50.4 (22.9)	<0.001	86.3 (13.7)	79.0 (31.5)	109 (18.5)	12.5 (5.6)	<0.001	0.595
Abietic acid	8.4 (3.6)	4.5 (4.7)	20 (13)	73 (41)	<0.001	2.1 (1.0)	1.6 (4.0)	90 (157)	25 (9.6)	0.944	0.088

[a] *Least squares mean model for first versus last year differences during winter sampling, adjusted for ambient $PM_{2.5}$, temperature and relative humidity. Observation numbers may vary from crude data due to missing ambient data.*
[b] *Least squares mean model for first versus last year differences during non-winter sampling, adjusted for ambient $PM_{2.5}$, temperature and relative humidity. Observation numbers may vary from crude data due to missing ambient data.*
[c] *Least squares mean model for winter versus non–winter differences, adjusted for year, ambient $PM_{2.5}$, temperature and relative humidity.*

Table 4. Mean (sd) for indoor $PM_{2.5}$ mass (µg/m³), and $PM_{2.5}$–associated OC/EC (µg/m³) and selected organics (ng/m³) at the middle school by season and year

	Winter Sampling					Non–winter Sampling					Season[c]
	2005/06 (n=12)	2006/07 (n=8)	2007/08 (n=10)	2008/09 (n=9)	Year[a] p–value	2005/06 (n=5)	2006/07 (n=11)	2007/08 (n=10)	2008/09 (n=6)	Year[b] p-value	p–value (n=69)
$PM_{2.5}$	7.7 (1.9)	15.2 (6.9)	6.5 (2.0)	20.4 (24.1)	0.022	10.1 (2.1)	19.8 (10.9)	9.9 (4.8)	5.6 (3.3)	0.342	0.778
Organic carbon	8.3 (1.5)	7.3 (0.7)	6.8 (0.9)	9.8 (1.4)	0.002	7.4 (1.0)	9.4 (2.3)	7.5 (1.6)	8.9 (1.3)	0.175	<0.001
Elemental carbon	0.2 (0.11)	0.2 (0.08)	0.1 (0.11)	0.3 (0.16)	0.009	0.1 (0.04)	0.2 (0.14)	0.0 (0.06)	0.1 (0.11)	0.809	0.086
Levoglucosan	573 (223)	341 (185)	598 (120)	405 (120)	0.068	225 (359)	130 (83.3)	412 (37.2)	149 (116)	0.385	0.004
Dehydroabietic acid	163 (46.2)	113 (71.9)	132 (13.6)	27.4 (9.3)	<0.001	94.7 (24.2)	67.4 (13.4)	129 (21.7)	10.6 (3.0)	<0.001	0.163
Abietic acid	7.1 (2.3)	6.9 (7.9)	11.3 (4.2)	37.6 (26.9)	<0.001	1.8 (1.6)	1.6 (1.6)	14.7 (7.0)	0.3 (0.0)	0.500	0.739

[a] *Least squares mean model for first versus last year differences during winter sampling, adjusted for ambient $PM_{2.5}$. Observation numbers may vary from crude data due to missing ambient data.*
[b] *Least squares mean model for first versus last year differences during non-winter sampling, adjusted for ambient $PM_{2.5}$. Observation numbers may vary from crude data due to missing ambient data.*
[c] *Least squares mean model for winter versus non–winter differences, adjusted for year and ambient $PM_{2.5}$.*

3.7. Indoor PM$_{2.5}$ correlations with ambient PM$_{2.5}$

Some of the school sampling studies have reported a strong correlation between ambient and indoor levels of PM$_{2.5}$. In a school study conducted in biomass–smoke impacted Christchurch, New Zealand, results showed a close relationship between the fine fraction of indoor and outdoor particles (Kingham et al., 2008). In a study conducted in Prague (Czech Republic) from November 2005 to August 2006, 24–hour indoor concentrations of PM$_{2.5}$ averaged 24.03 µg/m^3 in the studied gymnasium. In addition, these levels were closely correlated to ambient levels, suggesting a high outdoor–to–indoor penetration rate (Branis et al., 2009). We did not observe a positive correlation between the ambient and indoor PM$_{2.5}$ concentrations at the two schools in this study – even with wood burning residences in proximity to the schools (especially the elementary school as shown in Figure 1). Indeed, one of our schools demonstrated a strong inverse correlation between indoor and ambient PM$_{2.5}$ concentrations.

4. Conclusions

Results from the Libby woodstove changeout program have shown that targeting woodstoves can have a positive impact on ambient air quality throughout the winter months. Between 2005 and 2007, nearly 1 200 older model woodstoves were replaced with newer, cleaner burning models in an effort to reduce winter-time ambient PM$_{2.5}$ concentrations. A secondary (but perhaps more important) benefit was discovered from sampling conducted within wood burning residences in the area, where modest improvements to indoor air quality were measured (Ward et al., 2008; Noonan et al., 2012). These residential improvements were likely due to the replacement of woodstoves (and chimney packages) within each of the residences, directly influencing indoor air quality. To determine the impact of this community–wide intervention on indoor air quality within the local schools, PM$_{2.5}$ sampling was conducted in the gymnasiums of an elementary and middle school throughout the woodstove changeout over a four–year period.

Results from this study suggest that the changeout did not result in a measurable improvement on school indoor air quality. Overall, none of the chemical markers of woodsmoke (levoglucosan or resin acids), OC/EC, or PM$_{2.5}$ measures at either school showed a pattern that would be consistent with the timing of the woodstove intervention program. Even though compliance monitoring of PM$_{2.5}$ showed a ~20% reduction in ambient wintertime PM$_{2.5}$ as a result of the community–wide changeout over the same time period (Bergauff et al., 2009), crude ambient PM$_{2.5}$ measured on our scheduled winter indoor sample days was only 0.6 µg/m^3 lower in the fourth, post–changeout winter (2008/2009) compared to the 2005/2006 winter. Comparing these years after adjusting for ambient temperature indicated no significant differences.

These ambient results on the scheduled indoor sampling days could help explain why there was not an improvement in indoor air quality at the conclusion of the woodstove changeout. However, we did not observe a positive correlation between the ambient and indoor PM$_{2.5}$ concentrations at the two schools in this study – even with wood burning residences in proximity to the schools. This suggests the presence of indoor sources of PM$_{2.5}$ within the schools throughout the duration of the changeout. It is also possible that concentrations were dependent on the presence and/or absence of students in the gymnasiums, though this activity was not tracked in this study. Looking ahead to future studies, additional sampling can be used within these school gymnasiums to evaluate the effectiveness of indoor interventions, such as frequent mopping of the gym floors, as well as increased ventilation strategies.

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