

Control of wildfire-sourced PM_{2.5} in an office setting using a commercially available portable air cleaner

Dylan A. Stauffer, Daniel A. Autenrieth, Julie F. Hart & Stella Capoccia

To cite this article: Dylan A. Stauffer, Daniel A. Autenrieth, Julie F. Hart & Stella Capoccia (2020) Control of wildfire-sourced PM_{2.5} in an office setting using a commercially available portable air cleaner, Journal of Occupational and Environmental Hygiene, 17:4, 109-120, DOI: [10.1080/15459624.2020.1722314](https://doi.org/10.1080/15459624.2020.1722314)

To link to this article: <https://doi.org/10.1080/15459624.2020.1722314>



Published online: 11 Mar 2020.



Submit your article to this journal [↗](#)



Article views: 488



View related articles [↗](#)



View Crossmark data [↗](#)



Citing articles: 1 View citing articles [↗](#)



SHORT REPORT

Control of wildfire-sourced PM_{2.5} in an office setting using a commercially available portable air cleaner

Dylan A. Stauffer^a, Daniel A. Autenrieth^a, Julie F. Hart^a, and Stella Capoccia^b

^aSchool of Mines and Engineering, Safety, Health and Industrial Hygiene Department, Montana Technological University, Butte, Montana; ^bCollege of Letters, Sciences, and Professional Studies, Department of Biological Sciences, Montana Technological University, Butte, Montana

ABSTRACT

A steady increase in wildfire event severity and season length has led to greater potential for exposure to fine particulate matter associated with wildfire smoke. Research has found fine particulate matter to be correlated with a myriad of health ailments and thus effective strategies for controlling exposures are needed. In this study, a correction factor associated with wildfire-sourced fine particulate matter was established for a TSI SidePak AM520 by conducting sampling with a co-located MetOne BAM 1020. Portable air cleaner efficacy was assessed by simultaneously measuring PM_{2.5} mass concentrations in two identical offices with the inclusion of a portable air cleaner in one. The relationship between indoor and outdoor PM_{2.5} mass concentrations was assessed by comparing concentrations recorded in an office to those recorded at the nearest National Ambient Air Quality Standards monitoring station. Results revealed that a portable air cleaner reduced indoor fine particulate matter within an office by 73% and 92% during working and non-working hours, respectively, and that a strong significant correlation ($\rho = .91$, $p = 0.00$) existed between indoor and outdoor fine particulate matter mass concentration measurements. A direct relationship between indoor and outdoor PM_{2.5} mass concentrations was observed during this study, suggesting that elevated fine particulate matter concentrations due to wildfire smoke could be a concern in the indoor work environment; however the current study determined that the use of a portable air cleaner can substantially decrease fine particulate matter concentrations even in an active office setting.

KEYWORDS

Indoor air quality; occupational exposure; PM_{2.5}; SidePak personal aerosol monitor; wildfire smoke

Introduction

Research demonstrates an upward trend in the length of annual wildfire seasons (Jolly et al. 2015) as well as the severity of wildfire events (Flannigan et al. 2013), particularly in the western region of the United States (Abatzoglou & Williams 2016; Management 2018; Westerling 2016). A rise in wildfire activity increases the potential for dispersion of wildfire emissions into the ambient environment and ultimately increases the potential for exposure to wildfire smoke. The chemical constituents of wildfire smoke can vary depending on factors such as the type of organic material burned (Heilman et al. 2014; Liu et al. 2014; Urbanski 2014) and the distance a smoke plume has traveled (Alonso-Blanco et al. 2018; May et al. 2013; Zu et al. 2016); however, research has consistently shown wildfire smoke to be comprised primarily of fine particulate matter (Makkonen et al. 2010; Vicente et al. 2013),

which is defined as particles 2.5 micrometers (μm) and less in aerodynamic diameter (PM_{2.5}) (Park & Allaby 2017). Fine particulate matter stays suspended in air substantially longer than larger particulate matter, which makes it especially harmful to human health because it can be inhaled deep into the lung alveoli where it can enter pulmonary circulation and potentially systemic circulation (Wu et al. 2018).

It has been well established that combustion-related PM_{2.5} induces cardiopulmonary disorders (Brook et al. 2010; Crouse et al. 2012), general systemic inflammation, and respiratory inflammation (Wu et al. 2018). In addition to respiratory morbidity, research has found a strong correlation between wildfire smoke and all-cause mortality with an estimated 339,000 deaths attributed to varying levels of exposure to wildfire smoke globally (Analitis et al. 2012; Johnston et al. 2012). Because of the effect wildfire-sourced PM_{2.5} has on ambient particle concentrations,

it has been the general practice for public health agencies to recommend staying indoors during these events, and newer guidelines suggest also using an indoor portable air cleaner (PAC) (Elliott 2014; Maguet et al. 2018; MTDPHHS 2017; OHA 2014; USEPA 2016; WADOH 2015). A PAC is a device designed to remove airborne PM within a single room by circulating the air through some type of filter media. Previous studies have revealed positive results regarding the efficacy of PACs at reducing indoor wood-sourced PM_{2.5} concentrations in homes due to both wood stove smoke (Hart et al. 2011; McNamara et al. 2017; Wheeler et al. 2014) and wildfire-sourced smoke (Barn et al. 2008; Henderson et al. 2005). It is important to note that these studies were conducted in residential environments. Many individuals spend 40 hr a week or more in an office, and in contrast to a residential environment, an office setting may differ substantially due to variables of which occupants may have little control over including open doors and windows, HVAC system operation, and building occupancy. Each of these variables could markedly impact both the infiltration rate of PM_{2.5} and the effectiveness of a PAC at reducing indoor PM_{2.5} especially when ambient concentrations are elevated due to wildfire smoke. Limited information is available regarding methods of measuring and mitigating exposure to wildfire-sourced PM in an office setting, presenting a gap in research.

The present study aims to address this gap by fulfilling the following goals: (1) to establish a correction factor associated with wildfire-sourced PM_{2.5} for a direct reading instrument used throughout this study; (2) to evaluate the efficacy of a commercially available PAC at reducing wildfire-sourced PM_{2.5} concentrations in an active office setting; and (3) to assess the relationship between outdoor and indoor PM_{2.5} concentrations during a wildfire event.

Methods

Phase I: Correction factor comparison

Data collection

A TSI SidePak Personal Aerosol Monitor (AM520) (TSI Inc, Shoreview, MN, USA) that uses a light-scattering laser photometer to estimate particle mass concentration and is factory calibrated in accordance with ISO 12103-1, using A1 Arizona test dust (TSI 2016), was used throughout this study. A correction factor was established and applied to the AM520 mass concentration data to account for differences in aerosol characteristics between the A1 test dust used for

calibration and wildfire PM. All sampling conducted throughout this study was completed during the months of August and September, the height of the Pacific Northwest wildfire season, 2018. Two AM520 instruments that were factory calibrated within one year of data collection, along with a MetOne Instruments Beta Ray Attenuation Monitor 1020 (BAM 1020) (MetOne Instruments Inc., Grants Pass, OR) equipped with the specified accessories required to meet U.S. EPA Federal Equivalent Method (FEM) designation for continuous PM_{2.5} monitoring, were used throughout data collection. An AM520 was placed on the roof of the local county's National Ambient Air Quality Standards (NAAQS) air station within 1.5 m (60 in) of the BAM 1020 PM_{2.5} monitor inlet. To ensure the PM_{2.5} being sampled was primarily associated with wildfire smoke, sampling was not conducted unless smoke was visibly impacting the airshed and ambient PM_{2.5} concentrations exceeded or were trending toward an exceedance of a 12.0 µg/m³ threshold which was based on the EPA Air Quality Index "moderate" health effects category (12.1–35.4 µg/m³) (MTDEQ 2019). When this condition was met, an AM520 was operated simultaneously with the BAM 1020 from 8:00 AM to 6:00 PM and 8:00 PM to 6:00 AM. When successive sampling was performed, the AM520 that was previously in use was replaced by the second AM520 during the 2-hr window between sampling sessions.

Each AM520 was equipped with a PM_{2.5} impactor. The AM520 instruments were cleaned, operated, and calibrated per the manufacturer's instructions before and after each sampling session. The instruments were calibrated to a target flowrate of 1.7 L/min with a TSI Primary Calibrator model 4146 (Shoreview, MN). Meteorological data was recorded during pre- and post- calibration, and at the start and end of each sampling period using a Kestrel 4000 Pocket Weather Meter (Boothwyn, PA). The AM520(s) were programmed to log measured PM_{2.5} concentrations at 1-min intervals.

Data analysis

The timeframe in which the BAM 1020 was actively sampling during each hour was from minute xx:03 to xx:45; therefore, PM_{2.5} mass concentrations from the AM520(s) were compiled into 1-hr averages using the identical 42-min sampling period (Met One Instruments 2016). The paired, co-located 1-hr average PM_{2.5} mass concentrations and the log-transformed 1-hr average concentrations were not normally distributed. Thus, they were compared using

Spearman's rank-order correlation ($H_0: \rho = 0$ and $H_a: \rho \neq 0$). The relationship between the two measurement methods was assessed with the BAM 1020 data being treated as the reference concentration and the AM520 data being treated as the unknown concentration. An inverse regression equation based on this linear relationship was developed as one of the potential correction factors. A second correction factor was developed according to the manufacturer's instructions, and a third correction factor was adopted from previous research involving a TSI DustTrak light-scattering instrument and wood smoke-specific $PM_{2.5}$ (McNamara et al. 2011). Using each of the three correction methods, the mean, corrected $PM_{2.5}$ concentrations from the AM520(s) and the mean, $PM_{2.5}$ concentrations from the BAM 1020 were compared using a Welch's ANOVA ($H_0 =$ all means equal) due to the data lacking homogeneity of variance. The mean concentrations from each group were then compared using a post-hoc Games-Howell multiple comparisons test. Based on objective statistical analysis and subjective review of the data distributions, the correction factor that provided predicted concentrations that most closely resembled that of the reference concentrations was applied to all $PM_{2.5}$ mass concentrations collected via the TSI SidePak AM520(s) during later phases of this research.

Phase 2: Office $PM_{2.5}$ comparison

Data collection

In the second phase, the effectiveness of a portable air cleaner at reducing indoor $PM_{2.5}$ concentrations associated with wildfire smoke in an office setting was evaluated, and the infiltration of this smoke into an office setting was assessed. A 3M Filtrete Ultra Clean Air Purifier FAP02-RS equipped with a pleated fabric filter impregnated with electrostatically charged fibers that attract and capture airborne PM as small as $0.1 \mu\text{m}$ at 99.9% efficiency and is intended for room sizes up to 14.8 m^2 (160 ft^2) (3M 2006) was used during this phase. The Clean Air Delivery Rates for this unit are $253 \text{ m}^3/\text{hr}$ ($149 \text{ ft}^3/\text{min}$) for pollen, $217 \text{ m}^3/\text{hr}$ ($128 \text{ ft}^3/\text{min}$) for dust, and $175 \text{ m}^3/\text{hr}$ ($103 \text{ ft}^3/\text{min}$) for tobacco smoke (AHAM 2011). Two 12.2 m^2 (132 ft^2) single-occupancy university campus offices consisting of a single door, single window, and comparable furnishings were used. An AM520 was placed on the desk in both office environments 0.81 m (32 in) from the floor. The 3M Filtrete PAC, equipped with a newly replaced FAPF02 filter, was placed on the floor in one of the two offices approximately 2.1 m (84 in)

from the AM520. Placement of the PAC and the AM520(s) with respect to the single door, single window, and the occupant's seat was approximately equivalent within each office. The PAC was operated on the high setting and was only run for the duration of each sampling session. The PAC was rotated between the two offices each day. Both AM520 instruments, one in each office, ran simultaneously for two 8-hr sampling sessions per 24-hr period. Measurement sessions were conducted during both working and non-working hours. The daytime office sampling session was conducted from 8:00 AM to 4:00 PM in an active office setting in which both offices were occupied, the doors remained open, and visitors came and went as per usual. The nighttime office session was conducted during non-working hours from 10:00 PM to 6:00 AM when the offices were not occupied, and the office doors remained shut for the duration of the sampling period. The windows in each office remained closed during both the working and non-working sampling sessions.

To estimate $PM_{2.5}$ infiltration, the mass concentration data from the AM520 located in the office without the PAC was compared with paired hourly $PM_{2.5}$ readings from the BAM 1020 used during Phase I of this study at the local NAAQS air station located approximately 3 miles from campus. The AM520 sampling and calibration methods followed those described in Data Collection under Phase I of this narrative.

Data analysis

The correction factor chosen in Phase I was applied to all $PM_{2.5}$ concentrations measured by the AM520(s). To assess the effectiveness of the PAC, the corrected day and night $PM_{2.5}$ concentrations from each AM520 were categorized as "w/PAC" or "w/o PAC" based on the presence of the PAC. The $PM_{2.5}$ concentrations were compiled into 1-hr averages, and the differences of the sessions were compared using a paired, two-sample, one-sided, Wilcoxon Signed Rank test ($H_0: \mu_{(w/o \text{ PAC})} - \mu_{(w/PAC)} \leq 0$ and $H_a: \mu_{(w/o \text{ PAC})} - \mu_{(w/PAC)} > 0$) due to the paired differences lacking normality.

To estimate infiltration, the $PM_{2.5}$ mass concentrations collected from the AM520 in the office without the PAC were compiled into 1-hr averages using the identical 42-min sampling period as the BAM 1020. The paired, 1-hr average $PM_{2.5}$ mass concentrations and log-transformed concentrations were not normally distributed and were therefore compared using the Spearman rank-order correlation test ($H_0: \rho = 0$

Table 1. Summarized statistics of PM_{2.5} mass concentrations (µg/m³) measured using a MetOne BAM 1020 and both uncorrected and corrected PM_{2.5} mass concentrations (µg/m³) measured using a TSI SidePak AM520 after applying one of three correction factors (CF 1-3). All PM_{2.5} concentrations were recorded during periods in which the ambient airshed was impacted by wildfire smoke (n = 157).

	BAM 1020 Uncorrected	SidePak AM520			
		Uncorrected	CF-1 Regression Equation	CF-2 Manufacturer Instructed	CF-3 Established DustTrak CF (McNamara et al. 2011)
Mean (±SD)	22.80 (±14.21)	167.10 (±90.66)	22.80 (±15.72)	22.80 (±12.37)	101.27 (±54.95)
Median	19.70	146.50	19.23	19.99	88.79
Range	5.40 – 76.60	14.64 – 442.19	–3.63 – 70.49	1.99 – 60.33	8.87 – 267.99
Coef. Var.	62.33	54.26	68.94	54.26	54.26

and H_a: $\rho \neq 0$). A significance level of 0.05 was used for all tests, and all statistical analyses were performed using Minitab Statistical Software version 18 (State College, PA, USA).

Results

Phase I: Correction factor comparison

One-hour average PM_{2.5} concentrations ranging from 14.64–442.19 µg/m³ and 5.40–76.60 µg/m³ were recorded by the AM520(s) and BAM 1020, respectively. A total of 168 co-located 1-hr averaged concentrations were collected. Paired concentrations that were below the BAM 1020 1-hr measurement cycle Limit of Detection (LOD) of 4.8 µg/m³ were not included in the analysis, resulting in an effective sample size of n = 157. Using each of the three correction factor methods, descriptive statistics of the corrected and raw PM_{2.5} concentrations are presented in Table 1. Mean (±SD) PM_{2.5} concentrations of 167.10 (±90.66) µg/m³ and 22.80 (±14.21) µg/m³ were measured using the AM520(s) and BAM 1020, respectively. The median PM_{2.5} concentration measured by the AM520(s) was 146.5 µg/m³ and 19.70 µg/m³ for the BAM 1020. The concentrations measured using both instruments were not normally distributed, even after log-transformation. A moderately strong ($\rho = .84$) and statistically significant ($p = 0.00$) Spearman correlation was observed between the uncorrected AM520 and BAM 1020 PM_{2.5} concentrations. This monotonic relationship is shown in Figure 1.

The inverse regression equation derived from the line of best fit represented in Figure 1 was used as CF-1 shown in Table 2. Per the manufacturer's instructions (TSI 2016), the ratio of an average gravimetric concentration over an average photometric concentration can be used to develop a correction factor for the AM520. An averaged reference concentration from an FEM-designated instrument was used in place of the average gravimetric concentration to develop CF-2 in Table 2. Previous research involving

a TSI DustTrak revealed that this light scattering instrument, which is similar to the TSI SidePak AM520, reports wood smoke-specific PM_{2.5} concentrations 1.65 times higher than filter-based methods; therefore, the third CF compared in this study was the inverse of this ratio shown as CF-3 in Table 2 (McNamara et al. 2011).

After each correction factor was applied to the raw AM520 PM_{2.5} concentrations, the mean concentrations were compared using a Welch's ANOVA. There was a statistically significant difference between at least one of the means tested ($F = 173.91$, $p = 0.00$). Post-hoc Games-Howell analysis results are provided in Table 3. A mean difference of 0.00 was found for both CF-1 and CF-2 as compared to the reference concentration, and these differences were not statistically significant (adj. $p = 1.00$). The 95% confidence interval of the mean difference was -4.62 – 4.62 and -4.10 – 4.10 for CF-1 and CF-2, respectively. Box and whisker plots of the BAM 1020 PM_{2.5} concentrations and the corrected AM520 concentrations are shown in Figure 2 (see additional histogram plots in Figure S1 of the online supplemental material). These plots demonstrate similarity between the data sets where CF-1 and CF-2 were applied; however, as shown in the CF-1 boxplot in Figure 2 and under the range column of Table 1, CF-1 produced negative corrected concentrations. Although CF-2 may skew PM_{2.5} concentrations near the upper and lower limits as demonstrated by the narrower interquartile range shown in Table 1, PM_{2.5} concentrations corrected using CF-2 proved to be statistically similar to the BAM 1020 PM_{2.5} concentrations, did not contain any negative concentrations, and the difference in means testing produced a narrower 95% confidence interval. Additional comparison of correction methods one and two was conducted using a Bland-Altman analysis plot and revealed both a narrower limit of agreement and mean difference 95% confidence interval for CF-2, which is shown in Figure 3. Due to these factors it was determined that CF-2, based on the manufacturer's instructed ratio

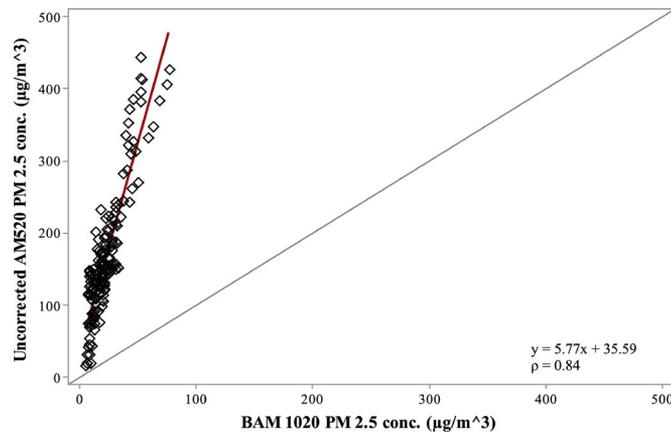


Figure 1. The y-axis represents uncorrected PM_{2.5} concentrations (µg/m³) measured using a TSI SidePak AM520 vs. PM_{2.5} concentrations (µg/m³) measured using a MetOne BAM 1020 which are shown on the x-axis. Each ◇ symbol represents a paired 1-hr average PM_{2.5} concentration recorded during periods in which the ambient airshed was impacted by wildfire smoke (n = 157). The line of best fit equation for the AM520 vs. BAM 1020 and Spearman rho are displayed in the bottom right-hand corner.

Table 2. Potential correction factors considered for wildfire-sourced PM_{2.5} concentration (µg/m³) measurements recorded using a TSI SidePak AM520 where y represents the concentration to which the correction factor is applied, and x represents the estimated true concentration.

Correction Factor	Equation
CF-1: Inverse Regression Equation	$x = \frac{(y - 35.59)}{5.77}$
CF-2: Manufacturer Instructed	$x = y \left(\frac{\text{avg. ref.}}{\text{avg. photo.}} \right)$ $x = y \left(\frac{22.8}{167.1} \right)$
CF-3: Established DustTrak (McNamara et al. 2011)	$x = y \left(\frac{1}{1.65} \right)$

Table 3. Results of Games-Howell simultaneous tests for differences of mean PM_{2.5} concentrations (µg/m³) measured by a BAM 1020 against both uncorrected and corrected mean PM_{2.5} concentrations (µg/m³) measured by a TSI SidePak AM520 (n = 157). The correction factors applied to the TSI SidePak AM520 include the Inverse Regression Equation (CF-1), the Manufacturer Instructed (CF-2), and the DustTrak Recommended (CF-3).

	Difference of Means	95% CI	Adjusted p-value
BAM vs CF-1	0.00	(-4.62, 4.62)	1.000
BAM vs CF-2	0.00	(-4.10, 4.10)	1.000
BAM vs CF-3	78.47	(65.98, 90.96)	0.000
BAM vs Uncorrected	144.30	(124.10, 164.50)	0.000

method of developing a correction factor for the TSI SidePak AM520, provided the best representation of the reference PM_{2.5} concentrations collected by the BAM 1020. A scatterplot of the corrected AM520 PM_{2.5} concentrations versus the BAM 1020 reference PM_{2.5} concentrations is provided in Figure 4. The association remained moderately strong (ρ = 0.84) and statistically significant (p < 0.001).

Phase II: Office PM_{2.5} comparison

Six sampling sessions were completed during the day and eight sampling sessions were completed during the night during wildfire smoke events, resulting in a

1-hr averaged PM_{2.5} concentration sample size of n = 48 and n = 64, respectively. Mean PM_{2.5} concentrations (µg/m³) were significantly reduced by the PAC as shown in Table 4 and illustrated in Figure 5. During the day, a mean (±SD) concentration of 11.09 (±9.70) µg/m³ was measured in the office without the PAC, and a mean concentration of 2.95 (±2.39) µg/m³ was measured in the office with the PAC. Overall, mean concentrations were found to be lower during the night sessions, with a mean concentration of 6.55 (±7.10) µg/m³ without the PAC and 0.50 (±0.39) µg/m³ with the PAC. Mean concentration percent change reductions of 73 and 92% were observed during the day and night, respectively. A paired, two-sample, one-sided, Wilcoxon Signed-Rank test run on the differences demonstrated there was a statistically significant mean difference between the “w/PAC” and “w/o PAC” pairs during both the day (Wilcoxon Test Statistic = 1176, p < 0.001) and night (Wilcoxon Test Statistic = 2080, p < 0.001) sessions.

To estimate infiltration, PM_{2.5} concentrations measured by the AM520(s) located in the office without the PAC were compiled into 1-hr averages that chronologically matched the 42-min sampling time-frame of the BAM 1020. Measured PM_{2.5} concentrations that were below the BAM 1020 LOD (4.8 µg/m³) were not included in the analysis, resulting in an

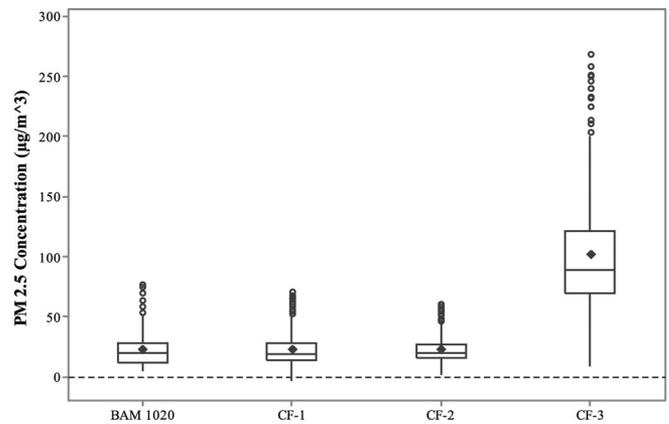


Figure 2. Box and whisker plot showing $PM_{2.5}$ concentrations ($\mu\text{g}/\text{m}^3$) recorded by a MetOne BAM 1020 and corrected $PM_{2.5}$ concentrations ($\mu\text{g}/\text{m}^3$) recorded by a TSI SidePak AM520 after applying the Inverse Regression Equation (CF-1), the Manufacturer Instructed (CF-2), and the Established DustTrak CF (McNamara et al. 2011) (CF-3) correction factors. All $PM_{2.5}$ concentrations were recorded during periods in which the ambient airshed was impacted by wildfire smoke ($n=157$). (The box represents the 25th–75th percentiles; the centerline represents the median, the diamond represents the mean, and the circles represent outliers which are defined as concentrations that were 1.5 times the interquartile range above the third quartile.)

effective paired sample size of $n=83$. Descriptive statistics of the indoor and outdoor $PM_{2.5}$ concentrations are presented in Table 5. One-hour average $PM_{2.5}$ concentrations ranging from 4.90–64.90 $\mu\text{g}/\text{m}^3$ were measured outdoors, and corrected $PM_{2.5}$ concentrations ranging from 2.25–46.19 $\mu\text{g}/\text{m}^3$ were measured indoors. Mean (\pm SD) $PM_{2.5}$ concentrations of 17.47 (\pm 13.07) and 10.44 (\pm 9.07) $\mu\text{g}/\text{m}^3$ were observed outdoors and indoors, respectively. A Spearman correlation test of the TSI SidePak AM520(s) and the BAM 1020 $PM_{2.5}$ concentrations demonstrated a strong, significant association ($\rho=0.91$, $p<0.001$). This relationship is shown in Figure 6.

Discussion

The present study found that when wildfire smoke impacted the ambient airshed, the TSI SidePak AM520 overestimated $PM_{2.5}$ concentrations by a factor of nearly seven while maintaining a moderately strong positive linear correlation ($R^2=0.82$) when compared to a co-located U.S. EPA FEM which is consistent with other studies ($R^2=0.98$ – 0.86) (McNamara et al. 2011; Wang et al. 2016; Yanosky et al. 2002; Zhu et al. 2011). To improve accuracy and compensate for laser photometer instruments overestimating $PM_{2.5}$ concentrations, various methods of developing and applying a correction factor have been published for a similar instrument, the TSI DustTrak. While a statistically significant difference was found when a previously developed wood smoke-specific correction factor of 0.61 (1/1.65) (McNamara et al. 2011) was applied to the concentrations measured during this study, no statistically significant difference was observed when an inverse regression (Yanosky et al.

2002) correction method was used. It was determined that a correction factor of 0.14, developed using the manufacturer's instructed ratio method, provided the best representation of the reference concentrations which suggests that correction factors are not only source specific but instrument specific as well, even when the instruments use similar measurement technology.

Phase II of this study determined that when the ambient airshed was impacted by wildfire smoke, a PAC resulted in a percent change reduction in mean indoor $PM_{2.5}$ concentrations of 73 and 92% during working hours and non-working hours, respectively. The lower percent reduction observed during working hours was expected due to the potential introduction of new PM via the open door, the resuspension of PM due to occupancy (Chatoutsidou et al. 2015), and the reduced efficiency of a PAC at controlling sporadic peaks as opposed to background PM concentrations (Cheng et al. 2016). The percent change reductions of mean indoor $PM_{2.5}$ concentrations due to a PAC observed in this study are slightly higher than those reported by previous research, which found a mean percent reduction range of 40–76% (Allen et al. 2011; Chen et al. 2015; Hart et al. 2011; Kajbafzadeh et al. 2015). Two additional studies that evaluated PAC efficacy in residences during wildfire events reported percent reductions of 63–88% (Henderson et al. 2005) and 65% (Barn et al. 2008), which are more similar to those observed in the current study; however, each of these studies utilized an equation to determine PAC efficacy rather than reporting mean percent change reduction. The higher percent reductions observed in these two studies and in the current study could be attributed to the

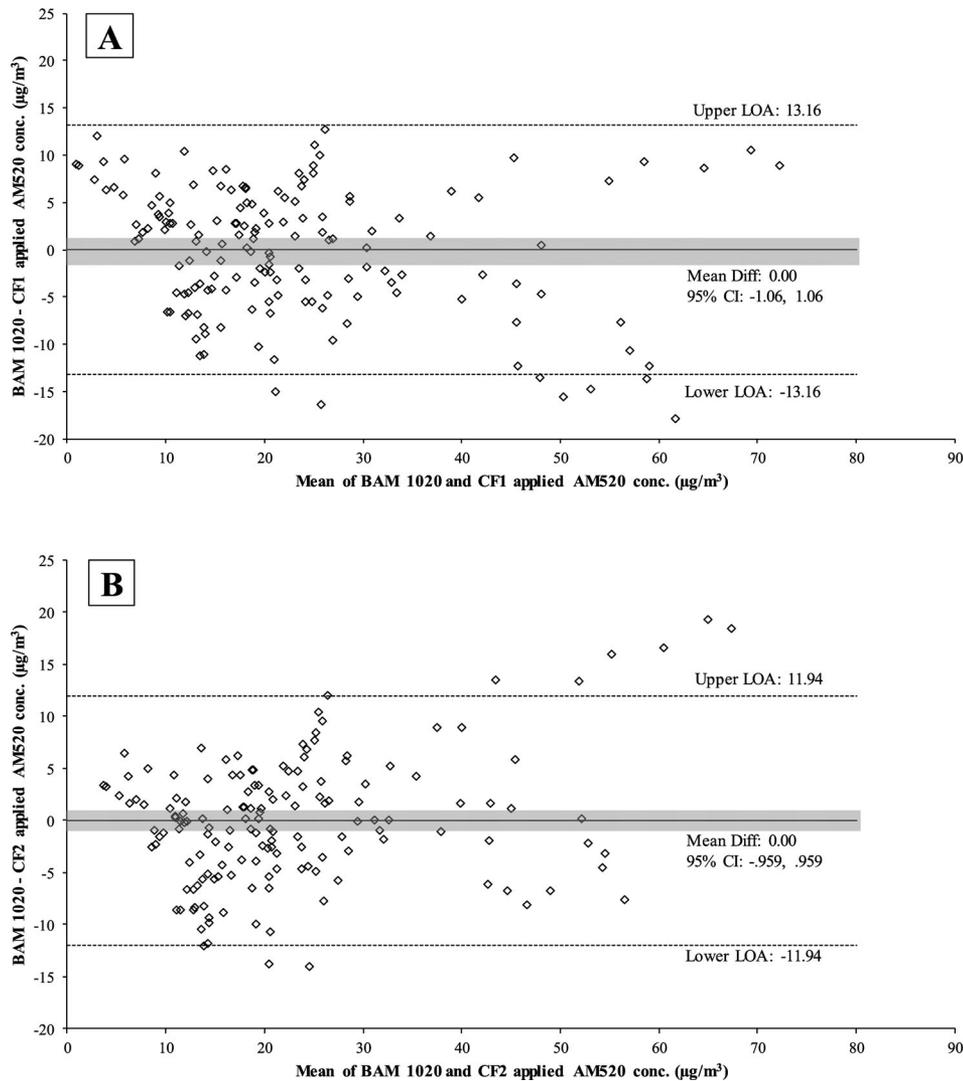


Figure 3. Bland and Altman plots of the differences between $\text{PM}_{2.5}$ concentrations measured by the BAM 1020 and the AMS520 (y-axis) vs. the mean of the two instrument measurements (x-axis). Corrected AMS520 concentrations using the Inverse Regression Method (CF-1) are shown in plot A, and corrected concentrations using the Manufacturer's Instructed Method (CF-2) are shown in plot B. All $\text{PM}_{2.5}$ concentrations were recorded during periods in which the ambient airshed was impacted by wildfire smoke ($n = 157$). The solid line represents the mean difference, the shaded area represents the 95% confidence interval of the mean difference, and the dotted lines represent the upper and lower Limits of Agreement.

elevated $\text{PM}_{2.5}$ concentrations that occur during wildfire events, which may introduce a higher potential for $\text{PM}_{2.5}$ reduction via PAC use. Regardless of why higher percent reductions were observed in this study, these results provide strong evidence suggesting that the use of a relatively inexpensive PAC may be an effective means of reducing office $\text{PM}_{2.5}$ exposures during a wildfire event.

The results of this study also demonstrated a strong, significant correlation ($\rho = 0.91$, $p < 0.001$) between $\text{PM}_{2.5}$ concentrations measured within the offices and $\text{PM}_{2.5}$ concentrations measured at the nearest NAAQS monitoring station. This relationship is consistent with previously mentioned residential studies which claimed that elevated ambient $\text{PM}_{2.5}$

concentrations due to wildfire smoke can significantly impact indoor air quality (Henderson et al. 2005; Kirk et al. 2018). These results suggest that, although ambient $\text{PM}_{2.5}$ measurements should not be used to predict indoor $\text{PM}_{2.5}$ concentrations during a wildfire event, higher ambient $\text{PM}_{2.5}$ concentration measurements may be indicative of higher indoor concentrations in an office building raising the concern that $\text{PM}_{2.5}$ associated with wildfire smoke could be an occupational hazard under these conditions.

Limitations

It is important to note that the relationship of outdoor and indoor $\text{PM}_{2.5}$ concentrations determined in

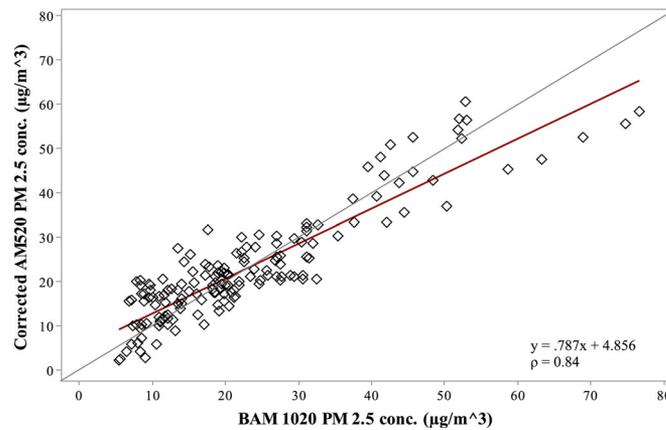


Figure 4. The y-axis represents corrected PM_{2.5} concentrations ($\mu\text{g}/\text{m}^3$) measured using a TSI SidePak AM520 vs. PM_{2.5} concentrations ($\mu\text{g}/\text{m}^3$) measured using a MetOne BAM 1020 which are shown on the x-axis. Each \diamond symbol represents a paired 1-hr average PM_{2.5} concentration recorded during periods in which the ambient airshed was impacted by wildfire smoke ($n = 157$). The line of best fit equation for the AM520 vs. BAM 1020 and Spearman rho are displayed in the bottom right-hand corner.

Table 4. Summarized statistics of corrected PM_{2.5} concentrations ($\mu\text{g}/\text{m}^3$) measured using a TSI SidePak AM520 in offices with and without a Portable Air Cleaner during working (Day) and non-working (Night) hours. All PM_{2.5} concentrations were recorded during periods in which the ambient airshed was impacted by wildfire smoke.

	Day		Night	
Mean % reduction	-73%		-92%	
<i>p</i> -value	0.00		0.00	
Wilcoxon Test Stat.	1176		2080	
Sample size	48		64	
PAC presence	Yes	No	Yes	No
Mean (\pm SD)	2.95 (\pm 2.39)	11.09 (\pm 9.70)	0.50 (\pm 0.39)	6.55 (\pm 7.10)
Median	2.17	6.49	0.37	3.33
Range	0.77–10.88	3.23–46.27	0.25–1.76	1.12–24.58

this study was assessed using simultaneously measured PM_{2.5} concentrations recorded approximately 3 miles apart. Ideally, this relationship should be assessed by collecting simultaneously recorded PM_{2.5} measurements both within and directly outside of the structure of interest, as several previously referenced studies, which also determined structure-specific infiltration factors, have done (Barn et al. 2008; Chen & Zhao 2011; Kearney et al. 2014; Macneill et al. 2014). No components of infiltration were assessed during this study including variables that could affect infiltration such as weather patterns and HVAC systems.

It is assumed that the PM_{2.5} aerosol measured in this study was primarily derived from wildfires; however, it is important to note that source apportionment was not performed. A source apportionment study conducted within this region revealed that when the airshed is impacted by wildfire smoke, 81% of the measured ambient PM_{2.5} was derived from wood combustion (Ward & Smith 2005). As described in the methods, efforts were made to ensure the PM_{2.5} being sampled was primarily associated with wildfire smoke. For practical reasons, the correction factor was not developed using a FEM inside of the offices which

could be considered a limitation due to a potential difference in size distributions between the ambient and infiltrated aerosols. A final aspect of this study that could be considered a limitation was that only one relatively short wildfire season affecting a single building, in one geographic location was evaluated.

Conclusion

This study demonstrated that when an airshed is impacted by wildfire smoke, the TSI SidePak AM520 overestimates ambient PM_{2.5} concentrations by a factor of nearly seven when compared to a U.S. EPA FEM; however, the application of a source-specific correction factor developed according to the manufacturer's instructed ratio method can compensate for this overestimation and greatly improve the level of agreement with a reference standard. This finding confirms the importance of applying a correction factor when using a light-scattering instrument, and that correction factors may be both source and instrument specific. In addition, during a wildfire season, this study demonstrated that the use of a PAC in an office setting can reduce PM_{2.5} concentrations by 73%

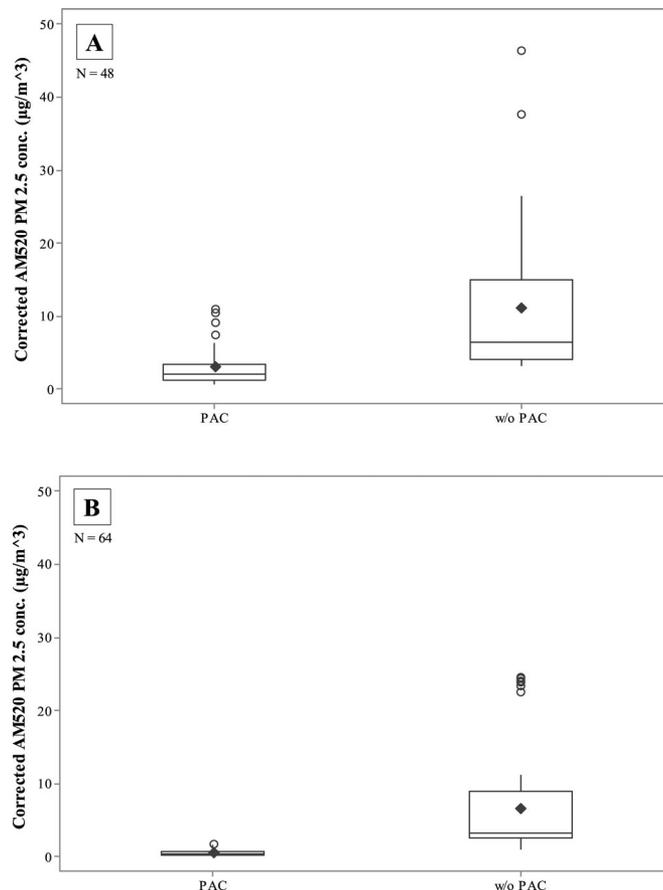


Figure 5. Box and whisker plot showing corrected $PM_{2.5}$ concentrations ($\mu\text{g}/\text{m}^3$) recorded by a TSI SidePak AM520 in offices with and without a Portable Air Cleaner during working (A) and non-working (B) hours. All $PM_{2.5}$ concentrations were recorded during periods in which the ambient airshed was impacted by wildfire smoke. (The box represents the 25th–75th percentiles; the centerline represents the median; the diamond represents the mean and the circles represent outliers which are defined as concentrations that fell 1.5 times the interquartile range above the third quartile.).

Table 5. Summarized statistics of corrected $PM_{2.5}$ mass concentrations ($\mu\text{g}/\text{m}^3$) measured in an office using a TSI SidePak AM520 (Indoor) and ambient $PM_{2.5}$ mass concentrations ($\mu\text{g}/\text{m}^3$) measured using a MetOne BAM 1020 (Outdoor) during periods in which the ambient airshed was impacted by wildfire smoke ($n = 83$).

	Mean (\pm SD)	Median	Range	Spearman rho	p – value
Indoor	10.44 (\pm 9.07)	11.80	2.25–46.19	0.91	0.00
Outdoor	17.47 (\pm 13.07)	6.33	4.90–64.90		

during active daytime hours and by 92% during stationary nighttime hours, and that outdoor $PM_{2.5}$ concentration fluctuations can directly influence indoor $PM_{2.5}$ concentrations. Even though this study was conducted during a relatively short wildfire season, a high correlation was found between $PM_{2.5}$ mass concentrations reported by the local NAAQS monitoring station and those measured within offices. This suggests that simply staying indoors may not provide adequate protection from wildfire-sourced $PM_{2.5}$. The current study presents new evidence suggesting that elevated ambient $PM_{2.5}$ concentrations could be a

concern in the indoor work environment, and that the use of a commercially available PAC in an office setting may be an effective control for protecting occupational health during the wildfire season.

While this study presents promising results using only a single PAC in a single building environment, further investigation into the efficacy of different PACs, which use utilize different filtering mechanisms and flow rates, is warranted to determine which type(s) work best at controlling $PM_{2.5}$ concentrations. Further, the use of different office, building, and HVAC system configurations is needed to determine whether or not PACs work similarly in different office environments. The use of personal exposure monitoring for building occupants working in spaces with and without PACs would clarify the effect that PACs can have on occupational exposures and inform whether occupational exposure limits are exceeded indoors during wildfires, and if so, whether PACs can adequately reduce exposures below these limits. The use of very low-cost homemade PACs is another

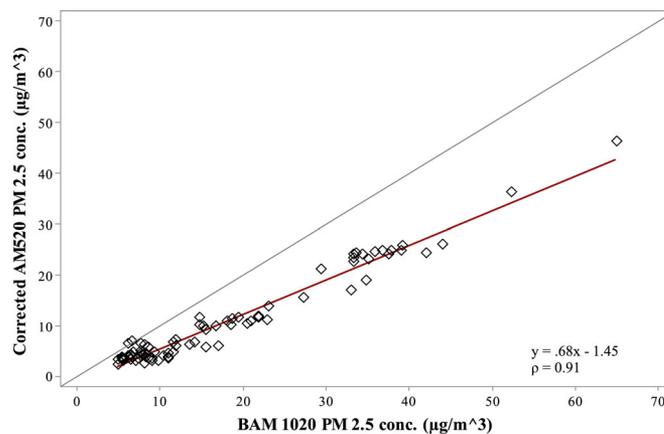


Figure 6. The y-axis represents corrected PM_{2.5} concentrations ($\mu\text{g}/\text{m}^3$) measured in an office using a TSI SidePak AM520 (Indoor) vs. ambient PM_{2.5} concentrations ($\mu\text{g}/\text{m}^3$) measured using a MetOne BAM 1020 (Outdoor) which are shown on the x-axis. Each \diamond symbol represents a paired 1-hr average PM_{2.5} concentration recorded during periods in which the ambient airshed was impacted by wildfire smoke ($n=83$). The line of best fit equation for the AM520 vs. BAM 1020 and Spearman rho are displayed in the bottom right-hand corner.

relevant area that needs additional study. With consideration to the volatile components of wildfire-derived PM, it may also be beneficial to investigate the efficacy of PACs in locations more immediate to active wildfire events, including potentially the evaluation of other hazardous components of wildfire smoke.

Acknowledgments

The authors greatly appreciate the cooperation from the Butte-Silver Bow Health Department for allowing access to the National Ambient Air Quality Standards monitoring station for data collection. Support for this project was provided in part by the Montana Tech NIOSH Training Project Grant (T03 OH008630). The contents of this publication are solely the responsibility of the authors and do not represent the official views of the Centers for Disease Control and Prevention or the National Institute for Occupational Safety and Health Department of Health and Human Services.

References

- 3M. 2006. Filtrete ultra clean air purifier FAP02-RS: Manual. St. Paul, MN: 3M.
- Abatzoglou JT, Williams AP. 2016. Impact of anthropogenic climate change on wildfire across western US forests. *Proc Natl Acad Sci USA*. 113(42):11770–11775. <http://www.pnas.org/lookup/doi/10.1073/pnas.1607171113>.
- AHAM 2011. CADR Ratings Guide for Air Purifiers - Air Purifier Guide. Air Purifier Guide. [accessed 2019 May 28]. <http://www.airpurifierguide.org/faq/cadr#cadr-air-purifiers>.
- Allen RW, Carlsten C, Karlen B, Leckie S, Van Eeden S, Vedal S, Wong I, Brauer M. 2011. An air filter intervention study of endothelial function among healthy adults in a wood-smoke-impacted community. *Am J Respir Crit Care Med*. 183(9):1222–1230. [accessed 2019 Jun 12] <http://www.ats-journals.org/doi/abs/10.1164/rccm.201010-1572OC>.
- Alonso-Blanco E, Castro A, Calvo AI, Pont V, Mallet M, Fraile R. 2018. Wildfire smoke plumes transport under a subsidence inversion: Climate and health implications in a distant urban area. *Sci Total Environ*. [accessed 2019 Mar 5] 619–620:988–1002. doi:10.1016/j.scitotenv.2017.11.142
- Analitits A, Georgiadis I, Katsouyanni K. 2012. Forest fires are associated with elevated mortality in a dense urban setting. *Occup Environ Med*. 69(3):158–162. [accessed 2019 Mar 1] <http://oem.bmj.com/lookup/doi/10.1136/oem.2010.064238>.
- Barn P, Larson T, Noullett M, Kennedy S, Copes R, Brauer M. 2008. Infiltration of forest fire and residential wood smoke: An evaluation of air cleaner effectiveness. *J Expo Sci Environ Epidemiol*. 18(5):503–511. [accessed 2019 Jan 27] www.nature.com/jes. doi:10.1038/sj.jes.7500640
- Brook RD, Rajagopalan S, Pope CA, Brook JR, Bhatnagar A, Diez-Roux A V, Holguin F, Hong Y, Luepker R V, Mittleman MA, et al. 2010. Particulate matter air pollution and cardiovascular disease: An update to the scientific statement from the American Heart Association. *Circulation*. 121(21):2331–2378. <https://www.ahajournals.org/doi/10.1161/CIR.0b013e3181d8bec1>.
- Chatoutsidou SE, Ondráček J, Tesar O, Tørseth K, Ždímal V, Lazaridis M. 2015. Indoor/outdoor particulate matter number and mass concentration in modern offices. *Build Environ*. [accessed 2019 Jan 27] 92:462–474. <https://www.sciencedirect-com.mtproxy.lib.umt.edu:3443/science/article/pii/S0360132315300020>.
- Chen C, Zhao B. 2011. Review of relationship between indoor and outdoor particles: I/O ratio, infiltration factor and penetration factor. *Atmos Environ*. 45(2):275–288. [accessed 2019 Feb 13]. doi:10.1016/j.atmosenv.2010.09.048
- Chen R, Zhao A, Chen H, Zhao Z, Cai J, Wang C, Yang C, Li H, Xu X, Ha S, et al. 2015. Cardiopulmonary benefits of reducing indoor particles of outdoor origin: A randomized, double-blind crossover trial of air purifiers. *J Am Coll*

- Cardiol. 65(21):2279–2287. [accessed 2019 Jun 12] <https://www.sciencedirect.com/science/article/pii/S0735109715016708>. doi:10.1016/j.jacc.2015.03.553
- Cheng KC, Park HK, Tetteh AO, Zheng D, Ouellette NT, Nadeau KC, Hildemann LM. 2016. Mixing and sink effects of air purifiers on indoor PM_{2.5} concentrations: A pilot study of eight residential homes in Fresno, California. *Aerosol Sci Technol*. 50(8):835–845. [accessed 2019 Feb 14] <https://www.tandfonline.com/doi/full/10.1080/02786826.2016.1197375>. doi:10.1080/02786826.2016.1197375
- Crouse DL, Peters PA, van Donkelaar A, Goldberg MS, Villeneuve PJ, Brion O, Khan S, Atari DO, Jerrett M, Pope CA, et al. 2012. Risk of nonaccidental and cardiovascular mortality in relation to long-term exposure to low concentrations of fine particulate matter: A Canadian national-level cohort study. *Environ Health Perspect*. 120(5):708–714. [accessed 2019 May 28] <http://dx.doi.org/10.1289/ehp.1104049>
- Elliott C. 2014. Guidance for BC public health decision makers during wildfire smoke events. www.bccdc.ca.
- Flannigan M, Cantin AS, De Groot WJ, Wotton M, Newbery A, Gowman LM. 2013. Global wildland fire season severity in the 21st century. *For Ecol Manage*. 294:54–61. [accessed 2019 Feb 24] <https://www.sciencedirect.com/mtproxy.lib.umt.edu:3443/science/article/pii/S0378112712006196>.
- Hart JF, Ward TJ, Spear TM, Rossi RJ, Holland NN, Loushin BG. 2011. Evaluating the effectiveness of a commercial portable air purifier in homes with wood burning stoves: A preliminary study. *J Environ Public Health*. [accessed 2019 Jan 27] 2011:324809. <http://www.ncbi.nlm.nih.gov/pubmed/21331283>. doi:10.1155/2011/324809
- Heilman WE, Liu Y, Urbanski S, Kovalev V, Mickler R. 2014. Wildland fire emissions, carbon, and climate: Plume rise, atmospheric transport, and chemistry processes. *For Ecol Manage*. 317:70–79. [accessed 2019 Mar 5] doi:10.1016/j.foreco.2013.02.001
- Henderson DE, Milford JB, Miller SL. 2005. Prescribed burns and wildfires in Colorado: Impacts of mitigation measures on indoor air particulate matter. *J Air Waste Manag Assoc*. 55(10):1516–1526. https://search-proquest-com.mtproxy.lib.umt.edu:3443/docview/68803806?accountid=28080&rfr_id=info%3Axri%2Fsid%3Aprimo. doi:10.1080/10473289.2005.10464746
- Johnston FH, Henderson SB, Chen Y, Randerson JT, Marlier M, DeFries RS, Kinney P, Bowman D, Brauer M. 2012. Estimated global mortality attributable to smoke from landscape fires. *Environ Health Perspect*. 120(5):695–701. [accessed 2019 Jan 27] <https://ehp.niehs.nih.gov/doi/10.1289/ehp.1104422>.
- Jolly WM, Cochrane MA, Freeborn PH, Holden ZA, Brown TJ, Williamson GJ, Bowman D. 2015. Climate-induced variations in global wildfire danger from 1979 to 2013. *Nat Commun*. 6(1):7537. [accessed 2019 Jan 27] <http://www.ncbi.nlm.nih.gov/pubmed/26172867>. doi:10.1038/ncomms8537
- Kajbafzadeh M, Brauer M, Karlen B, Carlsten C, Van Eeden S, Allen RW. 2015. The impacts of traffic-related and woodsmoke particulate matter on measures of cardiovascular health: A HEPA filter intervention study. *Occup Environ Med*. 72(6):394–400. [accessed 2019 Jun 12] www.clinicaltrials.gov. doi:10.1136/oemed-2014-102696
- Kearney J, Wallace L, MacNeill M, Héroux ME, Kindzierski W, Wheeler A. 2014. Residential infiltration of fine and ultrafine particles in Edmonton. *Atmos Environ*. [accessed 2019 Jan 27] 94:793–805. <https://www.sciencedirect.com/mtproxy.lib.umt.edu:3443/science/article/pii/S1352231014003677>.
- Kirk W. M, Fuchs M, Huangfu Y, Lima N, O’Keeffe P, Lin B, Jobson T, Pressley S, Walden V, Cook D, et al. 2018. Indoor air quality and wildfire smoke impacts in the Pacific Northwest. *Sci Technol Built Environ*. 24(2):149–159. [accessed 2019 Jan 27] <https://www.tandfonline.com/doi/full/10.1080/23744731.2017.1393256>. doi:10.1080/23744731.2017.1393256
- Liu Y, Goodrick S, Heilman W. 2014. Wildland fire emissions, carbon, and climate: Wildfire-climate interactions. *For Ecol Manage*. 317:80–96. [accessed 2019 Jul 1] doi:10.1016/j.foreco.2013.02.020
- Macneill M, Kearney J, Wallace L, Gibson M, Héroux ME, Kuchta J, Guernsey JR, Wheeler AJ. 2014. Quantifying the contribution of ambient and indoor-generated fine particles to indoor air in residential environments. *Indoor Air*. 24(4):362–375. [accessed 2019 Jan 27] <http://doi.wiley.com/10.1111/ina.12084>.
- Maguet S, Chairotto J, Health M, Corriveau A, Du Preez M, Columbia B, Canadian AE, Cross R, Eyamie J, Canada H, et al. 2018. Public health responses to wildfire smoke events. [accessed 2019 Feb 22]. [http://www.nccch.ca/sites/default/files/Responding to Wildfire Smoke Events EN.pdf](http://www.nccch.ca/sites/default/files/Responding%20to%20Wildfire%20Smoke%20Events%20EN.pdf).
- Makkonen U, Hellén H, Anttila P, Ferm M. 2010. Size distribution and chemical composition of airborne particles in south-eastern Finland during different seasons and wildfire episodes in 2006. *Sci Total Environ*. 408(3):644–651. [accessed 2019 Jan 27] <https://www.sciencedirect.com/mtproxy.lib.umt.edu:3443/science/article/pii/S0048969709010316>. doi:10.1016/j.scitotenv.2009.10.050
- Management UFS and DB of L. 2018. 2017 Pacific Northwest fire narrative. <https://permanent.access.gpo.gov/gpo90663/fseprd572820.pdf>.
- May AA, Levin EJT, Hennigan CJ, Riipinen I, Lee T, Collett JL, Jimenez JL, Kreidenweis SM, Robinson AL. 2013. Gas-particle partitioning of primary organic aerosol emissions: 3. Biomass burning. *J Geophys Res Atmos*. 118(19):11327–11338. [accessed 2019 Mar 2] <http://doi.wiley.com/10.1002/jgrd.50828>.
- McNamara ML, Noonan CW, Ward TJ. 2011. Correction factor for continuous monitoring of wood smoke fine particulate matter. *Aerosol Air Qual Res*. [accessed 2019 Jan 27] 11(3):316–323. <http://www.aaqr.org/doi/10.4209/aaqr.2010.08.0072>.
- McNamara ML, Thornburg J, Semmens EO, Ward TJ, Noonan CW. 2017. Reducing indoor air pollutants with air filtration units in wood stove homes. *Sci Total Environ*. 592:488–494. [accessed 2019 Jan 27] <https://www.sciencedirect.com/mtproxy.lib.umt.edu:3443/science/article/pii/S004896971730623X>. doi:10.1016/j.scitotenv.2017.03.111
- Met One Instruments. 2016. BAM1020 Particulate Monitor Operation Manual BAM1020-9800 Rev W. (Revision W): 1–108.

- MTDEQ. 2019. Breakpoints for Particulate Concentrations. Mont DEQ Website. [accessed 2019 Oct 9]. <http://deq.mt.gov/Air/SF/breakpointsrevised>.
- MTDPHHS. 2017. Wildfire smoke and your health DPHHS recommendations. [accessed 2019 May 23]. www.today-sair.mt.gov.
- OHA. 2014. Wildfire Smoke and Your Health. 6
- Park C, Allaby M. 2017. A dictionary of environment and conservation. [accessed 2019 Apr 6]. <http://oxfordreference.com/view/10.1093/acref/9780191826320.001.0001/acref-9780191826320>.
- TSI. 2016. SidePak Personal Aerosol Monitor AM520: Manual. TSI Man. (Revision A):1–77. <http://www.tsi.com/SIDEPAK-Personal-Aerosol-Monitor-AM510/>.
- Urbanski S. 2014. Wildland fire emissions, carbon, and climate: Emission factors. For Ecol Manage. [accessed 2019 Jul 1] 317:51–60.
- USEPA 2016. Wildfire smoke: A guide for public health officials. [accessed 2019 Feb 22]. https://www3.epa.gov/airnow/wildfire_may2016.pdf.
- Vicente A, Alves C, Calvo AI, Fernandes AP, Nunes T, Monteiro C, Almeida SM, Pio C. 2013. Emission factors and detailed chemical composition of smoke particles from the 2010 wildfire season. Atmos Environ. 71: 295–303. [accessed 2019 Feb 14] <https://www.sciencedirect.com.mtproxy.lib.umt.edu:3443/science/article/pii/S1352231013000873>. doi:10.1016/j.atmosenv.2013.01.062
- WADOH 2015. Improving ventilation and indoor air quality during wildfire smoke events recommendations for schools and buildings with mechanical ventilation. [accessed 2019 May 23]. www.doh.wa.gov/SchoolEnvironmentContact.
- Wang Z, Calderón L, Patton AP, Sorensen Allacci MA, Senick J, Wener R, Andrews CJ, Mainelis G. 2016. Comparison of real-time instruments and gravimetric method when measuring particulate matter in a residential building. J Air Waste Manag Assoc. 66(11): 1109–1120. [accessed 2019 Jan 27] <https://www.tandfonline.com/doi/full/10.1080/10962247.2016.1201022>.
- Ward TJ, Smith GC. 2005. The 2000/2001 Missoula Valley PM 2.5 chemical mass balance study, including the 2000 wildfire season - Seasonal source apportionment. Atmos Environ. 39(4):709–717. [accessed 2019 Jan 27] doi:10.1016/j.atmosenv.2004.10.042
- Westerling ALR. 2016. Increasing western US forest wildfire activity: Sensitivity to changes in the timing of spring. Philos Trans R Soc B Biol Sci. 371(1696). [accessed 2019 Jan 27] <http://www.ncbi.nlm.nih.gov/pubmed/27216510>.
- Wheeler AJ, Gibson MD, MacNeill M, Ward TJ, Wallace LA, Kuchta J, Seaboyer M, Dabek-Zlotorzynska E, Guernsey JR, Stieb DM. 2014. Impacts of air cleaners on indoor air quality in residences impacted by wood smoke. Environ Sci Technol. 48(20):12157–12163. doi:10.1021/es503144h
- Wu W, Jin Y, Carlsten C. 2018. Inflammatory health effects of indoor and outdoor particulate matter. J Allergy Clin Immunol. 141(3):833–844. [accessed 2019 Jan 27] <https://www.sciencedirect.com.mtproxy.lib.umt.edu:3443/science/article/pii/S0091674918300290>.
- Yanosky JD, Williams PL, MacIntosh DL. 2002. A comparison of two direct-reading aerosol monitors with the federal reference method for PM2.5 in indoor air. Atmos Environ. 36(1):107–113. [accessed 2019 Jan 27] <https://www.sciencedirect.com.mtproxy.lib.umt.edu:3443/science/article/pii/S1352231001004228>.
- Zhu Y, Smith TJ, Davis ME, Levy JI, Herrick R, Jiang H. 2011. Comparing gravimetric and real-time sampling of pm2.5 concentrations inside truck cabins. J Occup Environ Hyg. 8(11):662–672. [accessed 2019 Jan 27]. doi: 10.1080/15459624.2011.617234
- Zu K, Tao G, Long C, Goodman J, Valberg P. 2016. Long-range fine particulate matter from the 2002 Quebec forest fires and daily mortality in Greater Boston and New York City. Air Qual Atmos Health. 9(3):213–221. [accessed 2019 Mar 2] <http://link.springer.com/10.1007/s11869-015-0332-9>.