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Correction factor for continuous monitoring of wood smoke fine particulate matter

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Abstract

The US Environmental Protection Agency (EPA) has designated a handful of instruments as Federal Reference or Federal Equivalency Methods (FRM and FEM, respectively) for the monitoring of fine particulate matter ($PM_{2.5}$). More commonly used for indoor exposure assessment studies are optical scanning devices such as the DustTrak (TSI) due to the their portability and affordability. It is recommended by the manufacturer of these instruments that a "correction factor" be applied when assessing source-specific conditions. In this study, DustTraks were collocated with multiple samplers in various environments in an effort to establish an indoor, wood smoke-source specific correction factor. The DustTrak was found to report $PM_{2.5}$ levels on average 1.6 times higher than a filter based method in two indoor sampling programs. The DustTrak also reported indoor $PM_{2.5}$ concentrations 1.7 times higher than a FRM sampler during a regional forest fire event. These real-world scenarios give a correction factor within a reasonable range of the results of a controlled laboratory experiment in which DustTraks reported $PM_{2.5}$ approximately 2 times higher than a FEM. Our indoor wood smoke-specific correction factor of 1.65 will allow for DustTraks to be confidently used in quantifying $PM_{2.5}$ exposures within indoor environments predominantly impacted by wood smoke.

Keywords

$PM_{2.5}$; Indoor Air Pollution; Wood Burning

INTRODUCTION

Wood smoke is a complex mixture, with many components having well-documented adverse human health effects. Commonly detected pollutants in wood smoke include fine particulate matter with an aerodynamic diameter of 2.5 microns or less ($PM_{2.5}$), carbon monoxide, and nitrogen oxides, all of which are regulated in the ambient air by the EPA. Among these regulated pollutants, a comprehensive review by Naeher *et al.* (2007) concluded that $PM_{2.5}$ serves as the best exposure metric for wood smoke, and tends to be among the most elevated pollutants during exposure events for comparison with existing air quality standards.

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Worldwide, wood smoke is a significant source of ambient $PM_{2.5}$ in rural areas where wood is burned for domestic heating or cooking, and has been reported to account for up to 90% of ambient $PM_{2.5}$ levels in these areas (Epton *et al.* 2008; McGowan *et al.* 2002; Giles 2010). In the United States, wood stove biomass combustion is a common source of residential heating and has been identified as a major source of ambient $PM_{2.5}$ in the Northern Rocky Mountains of western Montana (Ward and Lange 2010; Ward *et al.* 2006; Schumpert *et al.* 2006; Noonan and Ward 2007). There are also significant wood smoke exposures that can occur in the indoor environment. Indoor air quality studies conducted in homes with wood stoves have shown elevated levels of $PM_{2.5}$ from wood smoke, with both short and long-term indoor levels frequently much higher than those occurring in the ambient environment. For example, in a residential study conducted in Libby, MT within 16 homes with older model wood stoves, the average (sd) concentration (as measured by DustTraks) was 51.2 (32.0) $\mu g/m^3$, with average spike levels of 434 (419) $\mu g/m^3$ (Ward *et al.* 2008).

Due to current regulatory standards and the health concerns regarding $PM_{2.5}$ exposure, conducting accurate measurements of $PM_{2.5}$ is critical. For monitoring the ambient environment, there are Federal Reference Method (FRM) samplers that have been evaluated and approved by EPA. These filter-based instruments (including the BGI PQ200 and the RAAS2.5 single and multi-day sampler) report a cumulative average over a 24-hour period. However, these FRMs do not provide valuable continuous data, such as changes in hourly $PM_{2.5}$ levels. These data are critical when investigating health effects of $PM_{2.5}$, as acute exposures to elevated levels of $PM_{2.5}$ have been linked to multiple adverse health outcomes (Barrett *et al.* 2006). To measure $PM_{2.5}$ variability in the ambient environment, Federal Equivalency Method (FEM) samplers such as the MetOne BAM-1020 can provide hourly $PM_{2.5}$ monitoring, thereby allowing for a more comprehensive exposure assessment.

People spend the majority of their time indoors (Klepeis *et al.* 2001), stressing the need for accurate indoor measurements of hazardous pollutants, including $PM_{2.5}$. The use of EPA FRM or FEM samplers to monitor residential indoor air for health-based research studies is impractical, as such instruments that are large, loud, and expensive. Instead, researchers commonly use more portable and affordable instruments such as the TSI DustTrak, which uses real-time photometric technology to monitor $PM_{2.5}$. Optical mass measurements (such as those made by the DustTrak) are dependent upon particle size and material properties, therefore custom calibrations are needed to improve the measurement accuracy when evaluating specific sources of combustion (i.e. diesel exhaust, biomass combustion, cooking, etc.). Optical samplers have also been shown to produce different results when collocated with filter-based sampling methods. For example, DustTrak measurements were shown to over-report ambient urban $PM_{2.5}$ levels by up to a factor of 3 when compared to the MetOne BAM-1020 (Chung *et al.* 2001).

In this study, our goal was to develop a correction factor that can be applied to $PM_{2.5}$ data generated when using a DustTrak in wood smoke exposure studies. Specifically, this correction factor can be applied to DustTrak data collected from wood smoke-dominated indoor environments such as from within homes using wood stoves or in homes impacted by nearby forest fires. To this end, we discuss in this manuscript the results of several studies.

First we report on studies that were conducted in a controlled laboratory setting, where DustTraks were compared to an FEM sampler (MetOne BAM-1020) during the burning of a wood stove. We then present the results of two indoor studies: where a DustTrak was compared with two different filter based methods within a university building during the forest fire season of summer 2007; and where a DustTrak was collocated with a filter-based method within multiple homes containing wood stoves. Finally, and for comparison with the indoor studies, we present the results of an ambient wintertime sampling study, where a DustTrak was collocated with a BAM within a community heavily impacted by residential wood smoke.

METHODS

Instruments Used

TSI DustTrak, Model 8520 and 8530 (TSI Inc., Shoreview, MN, USA)—The DustTrak is a real-time optical scattering instrument that measures particulate matter in the air flow by the extent of forward scattering of an infra-red diode laser beam. They are factory calibrated to the respirable fraction of standard ISO 12103-1, A1 test dust (formerly Arizona Test Dust). DustTraks were zeroed and the impaction plate was cleaned and greased/oiled as necessary prior to each sampling event. In developing the DustTrak correction factors, both the 8520 and 8530 models were utilized.

MetOne BAM-1020 (MetOne Instruments Inc., Grants Pass, OR, USA)—The BAM is a beta attenuation monitor that uses a small ^{14}C (carbon-14) element that emits a constant source of beta rays through filter tape. A vacuum pump pulls a measured and controlled amount of air through the filter tape, loading it with ambient $PM_{2.5}$. The attenuation of the beta ray signal is used to determine the $PM_{2.5}$ mass on the filter tape, and the volumetric concentration in ambient air is calculated. Specified settings and accessories were utilized to meet the US EPA Federal Equivalent Method designation for continuous $PM_{2.5}$ monitoring. The active sampling occurs for 42 minutes (from time xx:08 to time xx: 50 every hour), with the remaining 18 minutes designated for tape movement and reading of the filter tape before and after the sampling period. For this program, a BAM was used within our controlled laboratory wood smoke studies, as well as in the ambient trials conducted within a wood smoke impacted community.

Leland Legacy Pump / PEM (SKC Inc., Eighty Four, PA, USA)—A Leland Legacy was used to collect indoor $PM_{2.5}$ on preweighed Teflon filters housed in a Personal Environmental Monitor (PEM). Air was sampled at 10 L/min for the duration of the 24-hour events, with flow rates verified before and after each sampling event. Gravimetric analyses were conducted by a contracted environmental laboratory (Chester LabNet, Tigard, OR, USA). The Leland/PEM systems (collocated with a DustTrak) were used during the indoor sampling programs described in this study.

PQ200 (BGI, Waltham, MA, USA)—The PQ200 is a filter-based sampler that collects $PM_{2.5}$ at 16.7 L/min typically over 24-hour periods. It is an FRM sampler, and is widely used by many states for ambient compliance monitoring for comparison with National

Ambient Air Quality Standards (NAAQS). In this study, a certified PQ200 was collocated with a Leland/PEM and DustTrak to determine the accuracy of their mass measurements indoors during a forest fire smoke event. Chester LabNet (Tigard, OR, USA) provided pre-weighed Teflon filters for this program, and conducted final weights following sampling.

Instrument Comparisons

Controlled Laboratory Wood Stove Burns - DustTraks and MetOne BAM (FEM)—Eight controlled laboratory wood stove burns were performed within an enclosed room during June and July of 2009 to determine the accuracy of the DustTraks collocated with an FEM. In an effort to simulate the loading/stoking of a wood stove in a residential environment (and the exposures one would get by living within a wood stove home), wood stoves were loaded with 50 g of local softwood (Douglas Fir, Larch, and Ponderosa Pine) approximately every 15 minutes. The same batch of wood was used for all burn events. TSI DustTraks (DustTrak 8520 and 8530; ranging from 1-9 units per burn) were collocated with a BAM during each of the combustion trials within the same room as the wood stove. For this experiment we compared indoor $PM_{2.5}$ readings resulting from the loading/stoking activities rather than direct ambient emissions from the stack.

During each burn trial, the DustTraks recorded 60-second $PM_{2.5}$ averages while the BAM recorded hourly (42 minutes of active sampling) $PM_{2.5}$ concentrations. Due to the potential for high spikes of $PM_{2.5}$ occurring during the time in which the BAM was not actively sampling, only the DustTrak data from the 42 minutes/hour that the BAM was actively sampling were averaged to obtain the hourly DustTrak values for comparison with the BAM values.

Indoor Forest Fire Smoke Sampling – DustTrak, Leland/PEM, and BGI PQ200 (FRM)—During the summer of 2007, a DustTrak and Leland/PEM were collocated with a certified (flow, pressure, and ambient/filter temperature sensors) BGI PQ200 within a laboratory at The University of Montana (Missoula). Four 24-hour samples were conducted within the laboratory prior to the start of the 2007 forest fire season. When smoke from a forest fire (located approximately 15 miles west of Missoula) impacted the ambient air of the community, three 24-hour samples (including a DustTrak, Leland/PEM, and BGI PQ200 FRM sampler) were collected within the laboratory to determine the impact of the smoke on the indoor environment.

As our indoor residential study (described below) compared the $PM_{2.5}$ concentrations measured by a DustTrak with those measured by a Leland/PEM. This small indoor sampling program provided information on the comparability of the Leland/PEM and FRM filter based methods. In addition, this study illustrated how these three instruments compared during a different type of smoke challenge (i.e. wood smoke from forest fires compared to smoke from residential wood stoves) within the indoor environment.

Indoor/Residential Air Sampling – DustTrak and Leland/PEM—During the winters of 2007/2008 and 2008/2009, 43 residential air sampling events were conducted in homes with wood stoves in Libby, Montana. The air quality within each of these wood stove homes was primarily influenced by wood smoke from the wood stoves through both loading and

stoking events (Ward *et al.* 2008). Using a Leland Legacy pump/PEM, 24-hour samples were collected on Teflon filters. TSI DustTraks (model 8520) were also collocated within the homes during each of the 24-hour sampling events for comparison.

Ambient Air Sampling – DustTrak and MetOne BAM (FEM)—During the winter of 2006/2007, a TSI DustTrak (model 8520) was collocated with a MetOne BAM-1020 on the roof of the Lincoln County Health Department in Libby, Montana throughout seven 24-hour sampling events. Libby is a small community located in northwest Montana that frequently experiences elevated ambient levels of wood smoke-$PM_{2.5}$ throughout the winter months. Following the same rationale as in our indoor controlled laboratory experiment, only the DustTrak data from the 42 minutes/hour that the BAM was actively sampling were averaged to obtain the hourly DustTrak values for comparison with the BAM values.

Data Analysis

A Pearson's correlation was used to determine the linear correlation of $PM_{2.5}$ values reported by DustTraks and the BAM. To determine significance between correction factors based on grouped concentrations of $PM_{2.5}$, a one-way analysis of variance (ANOVA) was used. Correction factors were determined by the formula: correction factor for *x* instrument versus *y* instrument = *x* value for time *t*/*y* value for time *t*. Data are reported as mean ± standard deviation.

RESULTS AND DISCUSSION

Controlled Laboratory Wood stove Burns – DustTraks and MetOne BAM (FEM)

Throughout simultaneous sampling of wood smoke-impacted indoor air during eight controlled laboratory wood stove burns, the DustTraks and BAM showed a high correlation ($r = 0.9873$, 95% CI: 0.8684 - 0.9263, Fig. 1). The measured data included multiple data points from before, during, and after burn events, representing a wide range of concentrations. It should be noted that a flow error consistently occurred with the BAM when concentrations of $PM_{2.5}$ exceeded roughly 300 $\mu g/m^3$, therefore all data presented in Figure 1 are below 300 $\mu g/m^3$ as measured by the BAM.

To obtain a correction factor for the DustTraks in this scenario, hourly DustTrak averages (from the same 42 minutes/hour the BAM was actively sampling) were compared to hourly BAM values. The formula (Correction factor = hourly DustTrak value/hourly BAM value) yielded an average correction factor value of 2.18 ± 1.22 (Fig. 2). Low concentrations of $PM_{2.5}$ (< 40 $\mu g/m^3$) yielded greater variation in correction factor values than at higher concentrations of $PM_{2.5}$. Nevertheless, there is no significant difference found between the average correction factor values at arbitrary ranges of 0-40, 41-60, 61-100, and >100 $\mu g/m^3$ $PM_{2.5}$ (One-way ANOVA, $p = 0.34$, Fig.2).

Indoor Forest Fire Smoke Sampling – DustTrak, Leland/PEM, and BGI PQ200 (FRM)

In total, there were seven 24-hour sampling events that were conducted in this study. Four events were carried out prior to the start of the forest fire season (June 13, 2007 – August 3, 2007), while three sampling events were conducted during the forest fire season (August 8–

16, 2007). For the first four events, indoor concentrations were very low, with the FRM mass measurements between 1.5 and 5.5 μg/m^3. The PQ200 FRM sampler had a marginal correlation (r) with the other samplers (FRM/Leland PEM=0.6613, 95% CI: -0.8228 to 0.9920; FRM/DustTrak=0.8953, 95% CI: -0.4720 to 0.9978), while there was poor agreement between the DustTrak and Leland/PEM at these extremely low concentrations (r=0.3294, 95% CI: -0.9244 to 0.9802). At lower non-forest fire baseline concentrations the Leland/PEM system over-reported concentrations and the DustTrak under-reported concentrations compared to the FRM values (see Table 1).

At elevated indoor concentrations during forest fire events, there were strong correlations between the three instruments. Filter based mass results as determined by the FRM and Leland/PEM were very similar with one another (r=1.0), and both the Leland/PEM and the FRM correlated highly with the DustTrak (r=1.0 in each case). When comparing the DustTrak values versus both the FRM and Leland/PEM, DustTrak values were 1.7 and 1.6 times higher, respectively. In summary, findings from this small sampling program show that at low concentrations (<5 μg/m^3), the agreement between the three sampling methods can be variable. At more elevated concentrations due to ambient smoke from nearby forest fires there is good agreement between the filter-based methods, but the DustTrak reported concentrations 1.59-1.70 (correction factor) times higher than the filter based methods. The elevated indoor concentrations observed during these forest fire events were consistent with levels seen within wood stove homes.

Indoor/Residential Air Sampling – DustTrak and Leland/PEM

DustTrak and Leland/PEM units were collocated in residences using wood stoves during 43 (24-hour) indoor sampling events. Each of these homes had significant sources of $PM_{2.5}$ from wood stoves with the major causes being the loading and stoking of the wood stoves when in operation. Throughout this wintertime program, the average $PM_{2.5}$ concentration (sd) determined by the Leland/PEM (Teflon filter) was 30.7 (± 34.7) μg/m^3, while the continuous DustTrak measurements from the same sampling events was 43.3 (± 60.2) μg/m^3 (see Figure 3). Using the formula (correction factor= 24-hour DustTrak average concentration/24-hour gravimetric filter concentration), an average correction factor or of 1.60 ± 1.05 was calculated.

Ambient Air Sampling – DustTrak and MetOne BAM-1020 (FEM)

Seven 24-hour ambient sampling events resulted in an average correction factor (correction factor= hourly DustTrak value/hourly BAM value) of 1.43 ± 0.61 (see Figure 4). The average $PM_{2.5}$ concentration (sd) reported by the BAM during the seven events was 24.6 (± 8.0) μg/m^3, while the DustTrak reported an average concentration of 38.3 (± 21.2) μg/m^3.

This manuscript reports on a range of DustTrak correction factors that were developed under a variety of sampling scenarios. Table 2 summarizes these factors that can be utilized in specific exposure assessment scenarios, primarily in wood smoke impacted indoor and ambient environments. To simulate significantly elevated wood smoke exposures that can occur in the indoor environment, we conducted multiple wood stove burning simulations in a controlled laboratory setting. From these studies, the DustTraks were found to correlate (r

= 0.9873, 95% CI: 0.9807 to 0.9916) with the US EPA FEM BAM, yet over-reported $PM_{2.5}$ values by a factor of ~2.2 compared to the FEM sampler. Although the correction factors varied (see Figure 2), there was no significant difference in correction factor values between BAM reported $PM_{2.5}$ concentrations 0-40, 41-60, 61-100 and >100 $\mu g/m^3$.

The average wood smoke levels measured during the controlled trials are only quasi-representative of exposures typically encountered in wood stove homes (through loading and stoking activities). Wood stove homes would have additional PM sources as well as different venting and air movement than the laboratory simulations. Therefore, we feel that the correction factors developed during the indoor programs (summer 2007 and residential studies) likely are more representative of "typical" in-home wood smoke exposures. In these studies, a value of 1.6 and 1.7, respectively, were calculated. The results of the summer 2007 study also suggests that using the filter based Leland/PEM collocated to the DustTrak can provide an accurate correction factor in the absence of a collocated EPA approved FRM or FEM sampler.

With the exception of the laboratory simulations there was a narrow range of correction factors that were calculated, depending on the sampling scenario. One explanation for the observed differences was the sources of the $PM_{2.5}$. For the Libby ambient study, 82% of the $PM_{2.5}$ in the ambient air during the winter months was apportioned to smoke from residential wood stoves (Ward *et al.* 2006). For the controlled laboratory study, the measured $PM_{2.5}$ was exclusively from wood stove burning activities. The particles measured during the indoor programs had source contributions other than (and in addition to) wood smoke. For the residential study, this included cooking and cleaning emissions among others, as well as infiltration from outdoor sources.

The difference of aged versus fresh particles should also be considered. For the ambient correction factor development, sampling was conducted during the winter months when temperatures are often below freezing. The particles that were measured had been aged (from source to monitor), with many of the volatile components of the wood smoke likely condensed onto the particles (i.e. augmenting the particle sizes) before being measured. In contrast, the particles measured during the controlled laboratory setting were "fresh" particles, meaning that they were released from the wood stove during the loading/stoking of the wood stove and detected by the monitors less than a few minutes later. It is unknown if optical scanning devices, such as the DustTrak, measure aged vs. fresh particles differently than filter-based instruments. The $PM_{2.5}$ measured within the homes and during summer 2007 were likely a mixture of fresh and aged particles, therefore it is possible that this may be a source of the variation in these studies. In addition, these samplers collected measurements under indoor conditions (i.e. 70-75 °F). Meteorological conditions such as temperature, humidity, and wind speed have been shown to be important determining factors of ambient $PM_{2.5}$ (Hien et al. 2002), which may also reasonably explain the differences between the indoor- and ambient-based correction factors. These unavoidable differences in sampling conditions likely contributed to the discrepancy between the different correction factors.

Our wood smoke-specific findings are consistent with the conclusions of other sampler comparison studies. DustTraks have been shown to over-report PM levels in comparison to gravimetric or filter-based methods in several environments. Kingham *et al.* (2006) found that in a wood smoke-dominated ambient airshed, DustTraks over-recorded PM_{10} (particulate matter with <10 μm aerodynamic diameter) relative to a tapered element oscillating microbalance (TEOM) by a factor of 2.73. In an undisturbed indoor environment with no sources of wood smoke present, PM_{10} was found to be over-recorded by DustTraks by a correction factor of 2.2 when compared to a tripod mounted PQ100 (BGI Inc.) gravimetric sampler (Heal *et al.* 2000). Similarly, Yanosky *et al.* (2002) determined that a DustTrak measuring 24-hour $PM_{2.5}$ levels for indoor air not impacted by wood smoke correlated well with a FRM filter-based sampler, but required a correction factor of ~2.6. In an ambient environment that is frequently impacted by wood smoke in the wintertime (Schauer and Cass 2000), DustTraks were found to overestimate $PM_{2.5}$ by a correction factor of 3 compared to reference filter-based samples (Chung *et al.* 2001). These studies further illustrate the variety in source specific correction factors for both indoor and ambient environments.

One of the main limitations of this study was the small sample size for both the summer 2007 and ambient studies. However, the resulting correction factors (1.43, 1.59 and 1.70) generated from these programs are in close agreement with the correction factor developed as part of the indoor/residential air study (1.60) that had a substantially larger sample size (n=43). For the controlled wood smoke study, our largest obstacle was the frequency of flow errors with the BAM when concentrations of $PM_{2.5}$ exceeded roughly 300 $\mu g/m^3$. These errors occurred because the filter tape would become so thick with particles that the flow rate could not be maintained, resulting in the sample being flagged. Although elevated (sustained) concentrations above 300 $\mu g/m^3$ can occur within the residential environment, these levels are infrequent and short lived in developed countries. This is supported by a study conducted within 16 wood stove homes in Libby, Montana using the DustTrak, where the median spike concentration was 266 $\mu g/m^3$ (mean of 434 ± 419 $\mu g/m^3$) (Ward et al. 2008). Although there is the possibility that the DustTrak correction factors presented here are not representative at $PM_{2.5}$ concentrations above 300 $\mu g/m^3$, our field studies have shown that it can be applied for the majority of the measurements conducted within the wood stove homes. These findings may not be generalizable to homes in developing countries that use biomass combustion for cooking or heating.

CONCLUSIONS

The development of a $PM_{2.5}$ correction factor for wood smoke-dominated indoor environments has lasting implications in exposure assessment and epidemiological research. $PM_{2.5}$ has been linked to numerous health outcomes including, but not limited to, cardiovascular diseases and respiratory diseases such as bronchitis, asthma, and pneumonia (Naeher *et al.* 2007; Robin 1996; Neupane *et al.* 2009). While most studies correlate ambient $PM_{2.5}$ levels to health outcomes, it has been documented that most people spend >90% of their time indoors (Klepeis *et al.* 2001). As fuel prices continue to rise, the use of wood as a source of heat will likely continue to gain popularity, thereby subjecting more people to both acute and chronic wood smoke $PM_{2.5}$ exposures. As wood stoves are already a predominant

source of home heating in many rural areas across the northern Rocky Mountains (and other areas across the US and world where wood heating is prevalent), understanding the relationship between indoor wood smoke exposures and human health is critical.

Due to their portability and ease of use, light-scattering instruments are more appropriate for use in indoor air sampling studies. As it is impractical and expensive to allocate a FRM or FEM sampler to measure indoor residential $PM_{2.5}$ levels, the development of a $PM_{2.5}$ indoor wood smoke correction factor establishes confidence in studies that employ DustTraks for this purpose. We have developed a wood smoke-specific correction factor (1.65) to be applied to DustTrak measurements conducted in indoor settings. This correction factor can primarily be utilized to correctly quantify $PM_{2.5}$ exposures for those living in homes with wood stoves, which can help us further understand connections between wood smoke $PM_{2.5}$ exposure and adverse health outcomes.

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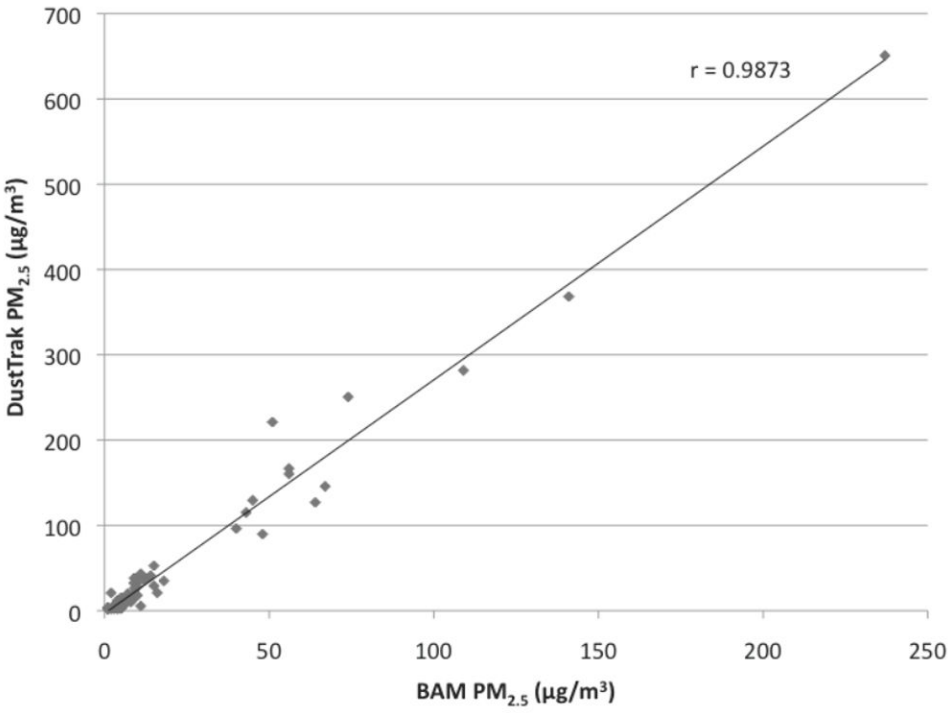

Fig. 1.

Correlation of DustTraks and BAM. Average hourly DustTrak $PM_{2.5}$ values vs hourly BAM values during controlled laboratory woodstove burns of Summer 2009. BAM values greater than 300 µg/m^3 and corresponding DustTrak value for that hour are excluded due to frequency of flow errors, resulting in unreliable data. r = 0.9873 (n= 90).

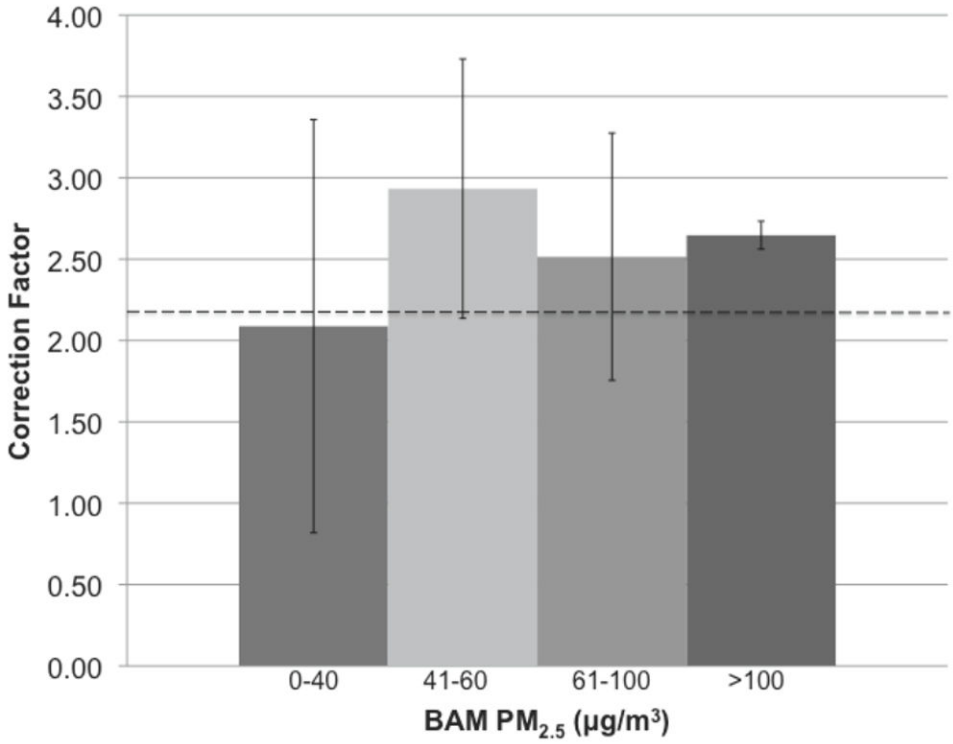

Fig. 2.

Average correction factor values ± standard deviation under laboratory conditions at concentration ranges of 0-40, 41-60, 61-100, and >100 µg/m^3 $PM_{2.5}$ (n=79, 6, 3, 2, respectively). One-way ANOVA p = 0.34. Dashed line indicates overall average of 2.18.

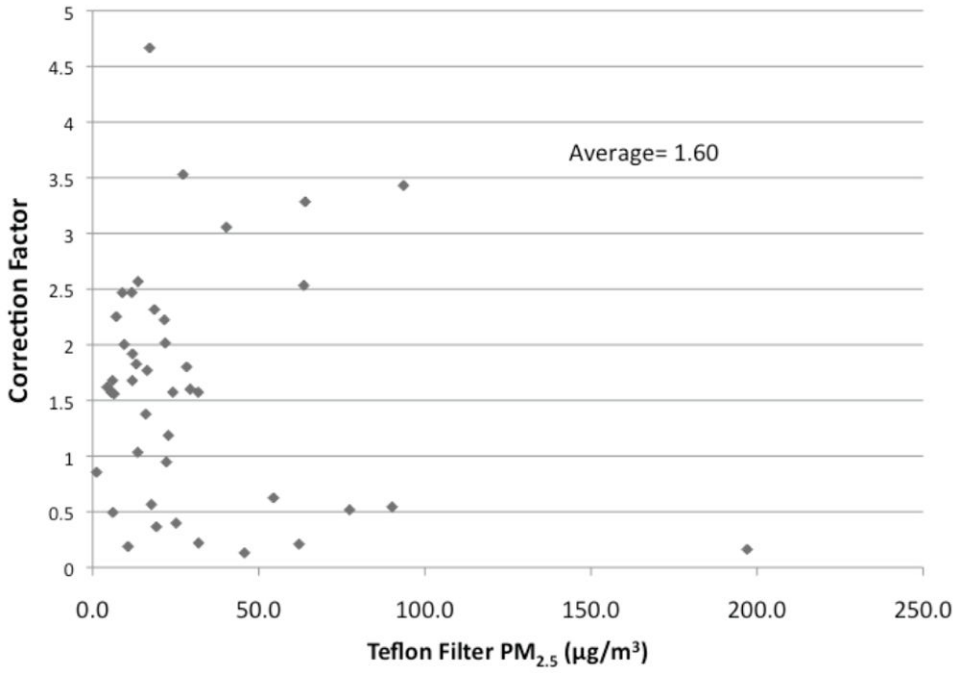

Fig. 3.

Correction factor values (24-hour DustTrak average/Gravimetric filter calculation) versus teflon filter values (µg/m^3) for residential indoor air in Libby, MT during the winters of 2007/2008 and 2008/2009 (n= 43).

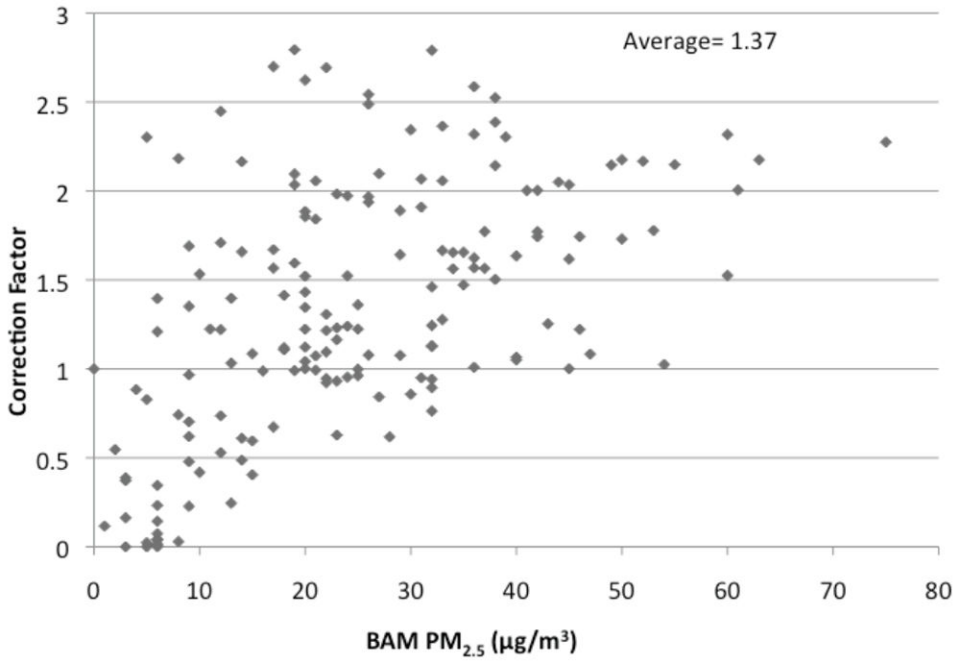

Fig. 4.

Correction factor values (Hourly DustTrak value/Hourly BAM value) for ambient air in Libby, MT during the winter of 2006/2007 (n= 168).

Table 1

Indoor $PM_{2.5}$ sampler comparison during the 2007 forest fire season.

	Date	FRM	Leland Legacy/PEM	DustTrak
Non-Forest Fire	6/13/07	3.8	10.6	2
	6/20/07	5.5	9.5	4
	6/25/07	1.5	3.2	0
	8/3/07	5.1	6.4	6
Forest Fire	8/8/07	10	11.3	15
	8/13/07	19.9	21.2	33
	8/16/07	53.6	55.3	104

Note: all units in $\mu g/m^3$.

Table 2

Summary of DustTrak wood smoke correction factors.

Sampling Scenario	Wood Smoke Source	Correction Factor ± Std Dev	Description
Controlled Laboratory Woodstove Burns	Wood stove	2.18 ± 1.22	DustTraks vs MetOne BAM (FEM)
Indoor Forest Fire Smoke	Wildland forest fire	1.59 ± 0.28	DustTrak vs Leland/PEM
Indoor Forest Fire Smoke	Wildland forest fire	1.70 ± 0.22	DustTrak vs BGI PQ200 (FRM)
Indoor/Residential Air	Wood stove homes	1.60 ± 1.05	DustTrak vs Leland/PEM
Ambient Air	Wood stove community	1.43 ± 0.61	DustTrak vs MetOne BAM (FEM)