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Original article

A comparative analysis of temporary and permanent beta attenuation monitors: The importance of understanding data and equipment limitations when creating $PM_{2.5}$ air quality health advisories

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ABSTRACT

Mobile particulate monitors are being widely used for smoke monitoring throughout the western United States. While this provides valuable additional data for public health decisions, quantifying the field performance of this equipment is necessary to understand measurement limitations when being compared with federal compliance instruments. Met One Instruments, Inc. Environmental Beta Attenuation Monitors (EBAMs) were co-located at permanently established Beta Attenuation Monitor (BAM) sites to determine agreement under normal field operating conditions. Monitors were assessed for agreement between fine particulate matter ($PM_{2.5}$) measurements. The instruments correlated for hourly (R^2 0.70) and daily (R^2 0.90) means. Mean difference for EBAM to BAM comparison showed the EBAM over-predicting the BAM by 24% (3 $\mu g\ m^{-3}$). Hourly concentrations fluctuated more in the EBAM. Daily mean concentrations were the most equitably comparable measurement for these monitors. Increases in relative humidity (RH) were associated with increased disagreement between monitors. When EBAM internal RH was below 40%, R^2 increased (0.76 hourly, 0.93 daily). The EBAM produced higher hourly AQI estimates. As a result of this study, it is advised to invalidate hourly data when the internal RH is greater than 40% and to only use daily AQI estimates to limit the EBAM AQI over-prediction.

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1. Introduction

Beta attenuation mass monitors are widely used to measure fine particulate matter ($PM_{2.5}$) throughout the United States of America. The Met One Instruments, Inc. model BAM-1020 (BAM) is used at many sites to assess $PM_{2.5}$ concentrations and determine compliance with state and federal air quality standards. The environmental beta attenuation monitor (EBAM) model from Met One Instruments, Inc. is intended for rapid temporary deployment and is not a federal reference or equivalency method approved for $PM_{2.5}$ compliance monitoring in the USA. BAMs are designed to operate in a temperature controlled enclosure while the EBAM is self-contained.

Both BAMs and EBAMs provide continuous hourly measurements of particulate matter concentrations using the relationship between attenuation of beta particles and particle deposition on a glass filter tape (Macias and Husar, 1976). There have been many studies that compare and contrast different particulate monitors that typically use regression to determine accuracy between monitors (Chung et al., 2001; Hains et al., 2007; Liu et al., 2013; Shin et al., 2011; Takahashi et al., 2008; Tasić et al., 2012) and include attempts to increase agreement through correction factors (McNamara et al., 2011). Various methods can help summarize agreement and offer different perspectives on the data (Haber et al., 2010; Liao, 2003; Lin et al., 2002). The use of correlation alone does not measure agreement and can be inappropriate to determine closeness between two measurements (Altman and Bland, 1983). This can be true for closely related measurements and if the points are clustered along the 1:1 line (Bland and Altman, 1995).

BAMs are widely used throughout California for compliance determination and to advise public health officials of air quality.

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EBAMs are regularly deployed throughout California to help air and public health regulators during wildland fires. This smoke monitoring is used to supplement the compliance sites and help public health officials in the determination of air quality impacts in areas typically long distances from the nearest permanent air quality monitor. Widespread use of the EBAM during smoke and other air pollution events has led to routine use of the data produced by an EBAM and its subsequent use as fully representative of BAM data when assessing public health exposure. While this supplemental data is of great value to public health exposure determination in less populated areas, understanding the comparability of data from these two similar monitors when operating in typical field conditions is of critical importance.

$PM_{2.5}$ monitoring equipment including the BAM and EBAM can be expected to perform at different precision and reliability (Baldauf et al., 2001; Takahashi et al., 2008). Error can be introduced from difference in monitor design or parts (Liu et al., 2013). Appropriate use of $PM_{2.5}$ data for public health protection and the determination of attainment and exceedance of federal and state air quality standards need include an understanding of the bias between mass measuring techniques (Chow et al., 2006).

Effects from field conditions can impact measurement agreement between instruments. Temperature has been shown to cause differences even between monitors that meet federal reference method (FRM) or federal equivalency method (FEM) requirements (Zhu et al., 2007). Volatilization from filter samples has been widely documented to reduce correlation between measurements (Chow et al., 2005; Hains et al., 2007). Relative Humidity (RH), both ambient and internal heater controlled, influence BAM $PM_{2.5}$ measurements (Huang and Tai, 2008). To offset effects from humidity, BAMs and EBAMs use an inlet heater which is designed to keep the internal humidity at or below a programmed set point.

The U.S. Environmental Protection Agency (EPA) has established the Air Quality Index (AQI) system to disseminate health impacts the public may experience over the course of hours to days. EBAMs are used to better assess AQI information disseminated to the public during temporary events (i.e. wildland fire smoke) by providing additional ground based monitoring in areas often long distances from federal reference monitors. Quite often EBAMs are deployed to less populated areas where air quality monitors, typically placed in the more populated urban areas, may not accurately reflect localized air quality conditions from a temporary short duration emission event or sited to capture specific transport of temporary emissions that may be missed by a permanent monitor.

During wildland fire used for fuel reduction and ecological benefit, data analysis of particulate matter measurements necessarily must be much more nuanced as any additional $PM_{2.5}$ exposure to a population is critical to effective smoke management. $PM_{2.5}$ concentrations in this analysis are typical of scenarios across federally managed lands in the Sierra Nevada, California during a smoke event from a wildland fire the size and intensity historically seen in this ecosystem (Schweizer and Cisneros, 2014). Low concentrations are typical in these scenarios and often burning can be subjected to management actions including full suppression on the basis of an individual temporary monitor reaching a single hour of 35 $\mu g\ m^{-3}$. We attempt to quantify with what certainty the data from mobile equipment can be used to make these difficult fire management decisions when in direct comparison to BAMs used for regulatory compliance.

Understanding the reliability and precision of monitoring systems under field conditions is important when attempting to accurately represent data using different measurement techniques. We attempt to quantify the differences between the BAM and EBAM while operated under conditions typical of field deployment in California during 2006, 2009, and 2011. In this paper we are attempting to understand BAM and EBAM agreement and highlight limitations that exist when comparing measurements. In particular we endeavor to determine the efficacy of using the portable EBAM when determining air quality impacts to human health when being utilized as a temporary monitor during a wildland fire or similar temporary emission scenario.

2. Methods

2.1. Monitor locations

BAMs and EBAMs were co-located at 5 locations (Fig. 1). A large high intensity wildland fire or other event that would create high $PM_{2.5}$ conditions was not encountered during any of these deployments. Thus, the following comparison does not include the high concentrations and ranges experienced during a full suppression wildland fire smoke event when correlation between equipment may be adequate for air managers to simply understand where the largest impacts are occurring and precision is less important.

2.2. Instrument descriptions

The BAM is designated by the U.S. Environmental Protection Agency (EPA) as an FEM with hourly measurements having a standard range of 0–1000 $\mu g\ m^{-3}$, resolution of ± 0.1 $\mu g\ m^{-3}$, and 24 h average lower detection limit less than 1.0 $\mu g\ m^{-3}$ (Met One Instruments, 2008a). The BAM is suitable for meteorological conditions in California (Chung et al., 2001) and is widely used throughout the state for regulatory monitoring of $PM_{2.5}$.

EBAMs have a measurement range of -5 to 65 530 $\mu g\ m^{-3}$, a data resolution of 1.0 $\mu g\ m^{-3}$, an accuracy of $\pm 10\%$ of the indicated value for hourly measurements (or 2.5 $\mu g\ m^{-3}$), and a 24 h average lower detection limit less than 1.2 $\mu g\ m^{-3}$ (Met One Instruments, 2008b). The EBAM does not meet FEM requirements. The EBAM is not appropriate for compliance determination but is a more portable and less expensive version of a BAM and used for temporary monitoring of particulate matter. The EBAM hourly measurement is obtained by using two 4 min counts. The first count in minutes 2–5 at the top of the hour establishes a zero reading while minutes 57–60 establish the total count for the hour. Tape advance can be set so a sample is accumulated over multiple hours on the same sample with the first and last 4 min counts coming at the top of each hour. A negative value can thus be determined if the count from the beginning of the hour to the end of the hour reduces which may indicate error in the hourly sample that is being measured by the individual hour reading negative. This negative hourly value is included in our calculations when stated to determine if the negative value increased comparative accuracy when calculating 24 h mean concentrations.

2.3. Instrument deployment and maintenance

Two EBAMs were run simultaneously at Thousand Oaks and Simi Valley sites in 2009 while at other sites one EBAM was used for comparison. Sampling at each site was from 1 to 3 months duration and all monitoring occurred between April and November (Table 1). Ojai and Simi were sampled in 2006; Simi, Thousand Oaks, and Springville in 2009; and Kernville in 2011. EBAMs were installed with inlets as near as possible to the same vertical height (within 0.5 m) and 1–2 m distance from BAM inlets. Internal RH set points on the BAMs and EBAMs were set to 40 or 45%. Tape advance was set between 1 and 8 h. Protocol for equipment function included integrity of the flow (leak check), temperature, and flow audits which were performed at a minimum of once every two weeks.

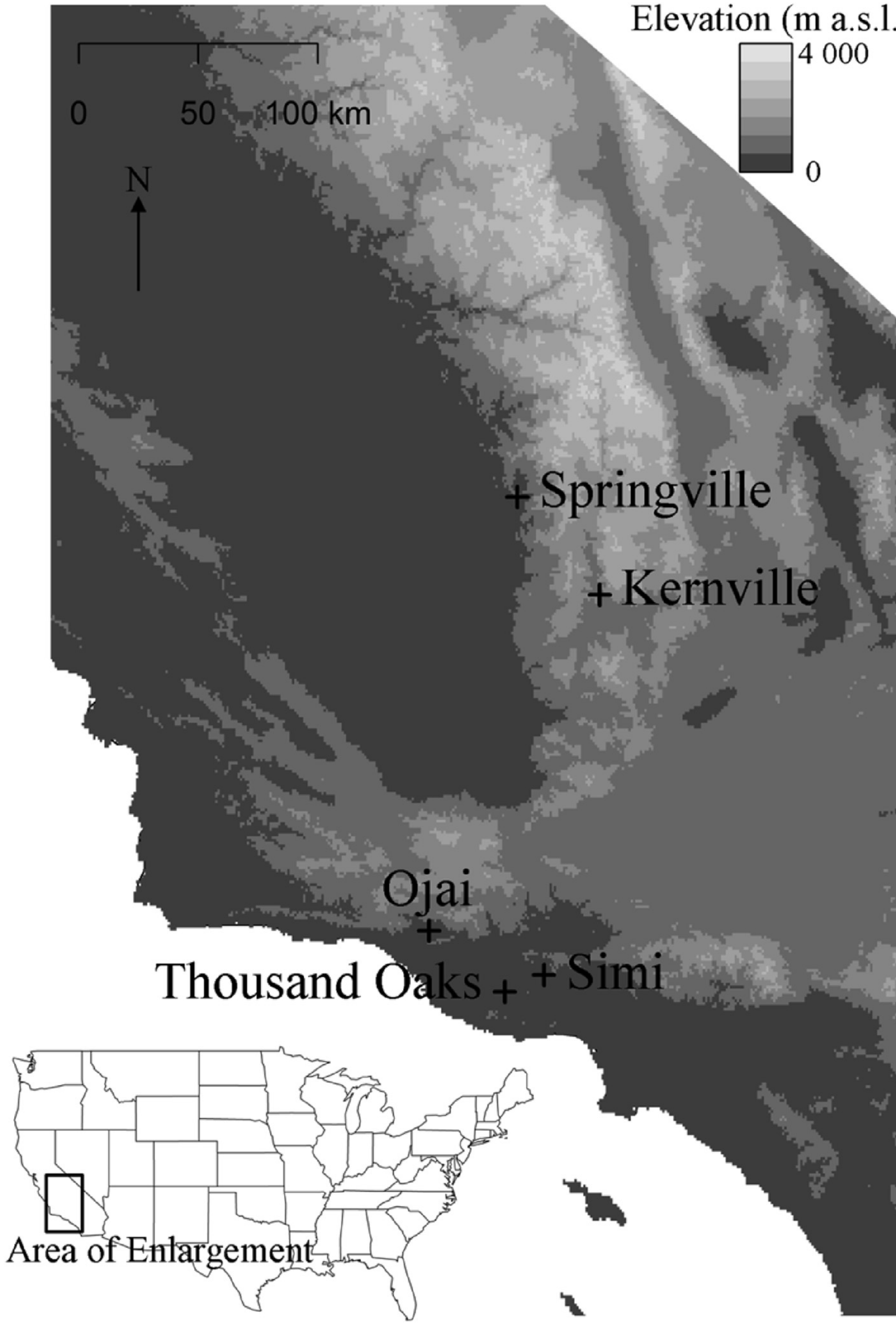

Fig. 1. Site locations (California, U.S.A.).

Table 1
Site locations and monitoring dates.

Site	Latitude (N)	Longitude (W)	Start	End
Kernville	35.75512	−118.4175	8/9/2011 12:00	11/28/2011 12:00
Springville	36.13588	−118.811	7/10/2009 12:00	8/4/2009 11:00
Thousand Oaks 1	34.21014	−118.8705	9/9/2009 13:00	10/26/2009 10:00
Thousand Oaks 2	34.21014	−118.8705	8/5/2009 15:00	10/26/2009 9:00
Thousand Oaks 3	34.21014	−118.8705	8/5/2009 15:00	9/4/2009 3:00
Simi 1	34.2764	−118.68375	8/5/2009 12:00	9/9/2009 9:00
Simi 2	34.2764	−118.68375	8/5/2009 11:00	9/9/2009 9:00
Simi 3	34.2764	−118.68375	4/19/2006 13:00	5/19/2006 12:00
Ojai	34.44804	−119.23131	4/20/2006 12:00	5/12/2006 11:00

Data was considered valid if there were no errors logged by the instrument and all audits were passed. To correct for the noise band of several micrograms on the BAM (Met One Instruments, 2008a), the occasional negative values for the BAM were set as zero for all calculations. EBAM hourly values were calculated setting negative values to zero and including the negative values as noted in the text, figures, and documents as a separate calculation. EBAMs frequently produced a negative number before and/or after an hourly value that was much higher than the BAM. Therefore, the negative hourly values were included in one set of calculations to determine the significance of including negative hourly EBAM measurements when comparing 24 h mean values between instruments.

2.4. Measurement effects from site conditions

Meteorological conditions were assessed using linear modeling with the difference (BAM-EBAM) in $PM_{2.5}$ hourly concentration (PM_{diff}) for individual explanatory variables of internal (heater controlled) relative humidity (RHi), ambient relative humidity (RHx), ambient temperature (t) in degrees Celsius (C), wind speed (WS) in meters per second (mps), and wind direction (WD) in degrees. Multiple linear regression was used with PM_{diff} described by multiple meteorological site conditions as:

$$PM_{diff} = t + RHx + RHi + WS + WD$$

Statistical calculations and graphics were produced using the software environment R (R Core Team, 2015).

2.5. Correlation and agreement

Regression of BAM and EBAM measurements was used to determine the linear relationship and variance between these two methods of measuring $PM_{2.5}$. Mean difference between EBAM and BAM measurements were used to assess the agreement between hourly, 3 h mid-point mean, and daily mean averages. The mean difference (m) was calculated (EBAM-BAM) along with the standard deviation of the differences (s). The m was considered the bias between the 2 measurements with the levels of the limits of agreement determined as $m + 2s$ and $m - 2s$ (Bland and Altman, 1986) with smaller values being better numerical agreement between EBAM and BAM measurements. Daily averages required 18 or more valid hourly readings.

2.6. AQI

The AQI was calculated to compare the effectiveness of data being used in smoke advisories. AQI was used to determine any differences when communicating to the public. The AQI reporting system has 6 categories (good, moderate, unhealthy for sensitive groups, unhealthy, very unhealthy, and hazardous) with thresholds depending on a given pollutant. Daily and 1 (and 3) hour $PM_{2.5}$ AQI is calculated using established breakpoints from the EPA and California Office of Environmental Health and Hazard Assessment (Lipsett et al., 2013). The daily or 24 h breakpoints for $PM_{2.5}$ are good 0–12, moderate 12.1–35.4, unhealthy for sensitive groups 35.5–55.4, unhealthy 55.5–150.4, very unhealthy 150.5–250.4, hazardous 250.5–500 $\mu g\ m^{-3}$. For 1 and 3 h $PM_{2.5}$ exposure breakpoints of good 0–38, moderate 39–88, unhealthy for sensitive groups 89–138, unhealthy 139–351, very unhealthy 352–526, and hazardous >526 $\mu g\ m^{-3}$ are used.

3. Results

Hourly concentrations of $PM_{2.5}$ at all sites were between 0 and 93 $\mu g\ m^{-3}$ for BAMs and −5 and 102 $\mu g\ m^{-3}$ for EBAMs (Table 2). Hourly concentrations measured by the EBAM routinely exceeded the BAM with EBAM hourly maximum concentrations typically greater than those of the BAM (the difference between the maximum values recorded at the instruments ranged from 7 to 39 $\mu g\ m^{-3}$). The exception was at Kernville where the BAM hourly maximum was 11 $\mu g\ m^{-3}$ greater than the EBAM. Daily mean concentrations were 0.9–40.1 $\mu g\ m^{-3}$ for BAMs and 0.1–48.2 for EBAMs. The EBAMs typically measured the highest daily mean concentrations which were 0.5–26.4 $\mu g\ m^{-3}$ above the co-located BAM. Kernville and Simi (during the sampling in 2006) had maximum BAM daily mean concentrations higher (2.0 and 6.0 $\mu g\ m^{-3}$ respectively) than the EBAM (Table 2).

3.1. Site condition impacts

WS and RHi had the strongest correlation to PM_{diff} (Table 3) with higher RH leading to a pattern with noticeable loss of agreement while WS was more evenly distributed (Fig. 2). RH is understood to impact $PM_{2.5}$ measurements with ambient temperature impacting agreement between gravimetric and beta attenuation sampling through loss attributed to volatilization of particulate nitrate and organic compounds from sample heating particularly above 20 °C (Zhu et al., 2007). We found differences between the BAM and EBAM to be primarily correlated to a RHi above 40%. Atmospheric temperature even at higher temperatures (>=20 C) showed little correlation (Table 3). Since both the BAM and EBAM use a similar system of sample heating, we expected the differences between these samplers would be limited to the effectiveness of the EBAM to control RHi.

Fig. 3, showing the partial residual plots from the multiple linear regression model, illustrates the potential impact of higher RHx and particularly RHi >45% on agreement between the BAM and EBAM. The impact of RHx and RHi for our data was in agreement with the published results for these types of monitors (Huang and Tai, 2008; Takahashi et al., 2008).

3.2. Correlation

Co-located sampling of ambient concentrations of $PM_{2.5}$ between the BAM and EBAM had strong correlation at each individual site with a range of R^2 for hourly data of 0.6828–0.8194 ($p < 0.001$) and for daily of 0.9100–0.9964 ($p < 0.001$). Correlation was strong when all sites were analyzed together with hourly measurements having an R^2 of 0.7042 ($p < 0.001$) and daily values with an R^2 of 0.9039 ($p < 0.001$). Best correlation of BAM and EBAM data found was an R^2 of 0.7663 for hourly and 0.9334 for daily ($p < 0.001$) when the internal RH of the EBAM was below 40% and negative EBAM hourly values were replaced by zero (18 or more hourly values were required for a daily mean). Additionally, correlation of BAM and EBAM 3 h mean (mid-point) $PM_{2.5}$ (minimum of 2 hourly values) resulted in an R^2 increase from 0.7663 for the hourly EBAM to 0.8230 using the 3 h mean (Table 4).

3.3. Mean difference agreement

Internal RH had the largest effect on both hourly and daily agreement. Mean difference calculations (EBAM-BAM) all showed a positive bias of ~3 $\mu g\ m^{-3}$ with standard deviation of the differences decreasing when higher EBAM internal humidity was removed (Table 5). Upper and lower limits of agreement for hourly values were closest when EBAM internal RH was below 40% and

Table 2
Site specific summary statistics for hourly (a) and daily (b) concentrations including correlation (R^2) between the BAM and EBAM.

Site	R^2	Mean		Median		Max		Min		Standard deviation	
		BAM	EBAM	BAM	EBAM	BAM	EBAM	BAM	EBAM	BAM	EBAM
(a) Correlation and comparison of hourly $PM_{2.5}$ at each site.											
Kernville	0.7180	9.6	12.6	8	12	93	82	0	−5	9.1	9.5
Springville	0.7440	9.2	18.0	9	19	28	49	0	−5	4.5	8.7
Thousand Oaks 1	0.6894	9.4	10.7	8	9	46	76	0	−5	6.5	12.8
Thousand Oaks 2	0.6828	11.4	10.5	10	9	51	90	0	−5	7.1	11.4
Thousand Oaks 3	0.7339	14.1	26.2	14	23	51	90	0	−5	7.4	18.3
Simi 1	0.8194	17.1	22.8	17	20	79	102	0	−5	9.3	15.1
Simi 2	0.8000	17.1	19.9	17	18	79	86	0	−5	9.3	13.5
Simi 3	0.6993	19.6	16.5	19	14	67	99	0	−5	11.7	15.6
Ojai	0.7412	15.4	14.1	15	14	29	46	0	−5	5.1	9.3
all data	0.7042	12.5	15.1	11	14	93	102	0	−5	9.1	9.5
(b) Correlation and comparison of daily $PM_{2.5}$ at each site.											
Kernville	0.9432	9.6	12.8	8.6	12.4	30.8	28.8	0.9	6.5	5.3	4.3
Springville	0.9737	9.3	18.3	9.5	18.4	12.3	20.9	3.6	14.5	2.3	1.6
Thousand Oaks 1	0.9100	9.5	11.3	9.0	11.0	23.3	39.5	2.3	1.4	4.8	8.3
Thousand Oaks 2	0.9704	11.5	11.1	11.8	11.6	22.5	28.0	2.4	0.1	5.2	5.6
Thousand Oaks 3	0.9459	14.3	26.5	14.8	25.8	21.8	48.2	3.1	7.6	5.0	11.8
Simi 1	0.9798	17.3	22.2	18.5	22.1	27.8	42.3	3.9	5.4	6.0	8.5
Simi 2	0.9764	17.3	20.2	18.5	19.2	27.8	40.1	3.9	6.3	6.0	8.9
Simi 3	0.9835	19.6	17.1	20.2	16.6	40.1	34.1	2.7	6.3	10.0	8.2
Ojai	0.9964	15.6	14.4	15.5	14.8	19.3	19.8	11.4	8.8	2.9	3.5
all data	0.9432	12.6	15.4	11.2	13.8	40.1	48.2	0.9	0.1	6.6	8.2

Table 3
Assessment of individual meteorological explanatory variables on difference in $PM_{2.5}$ hourly concentrations (BAM-EBAM).

Variable	R^2	p-value
Temperature (°C)	0.0011	0.001
RH	0.0037	p < 0.001
Internal RH	0.0192	p < 0.001
Wind speed	0.0229	p < 0.001
Wind direction	0.0015	0.013
Internal RH ≥ 40%	0.1247	p < 0.001
Temperature ≥ 20	0.0003	0.215

zero was used in place of negative values. When comparing hourly BAM readings to mid-point 3 h mean concentrations of the EBAM, the EBAM over-predicted the BAM. Replacing negative EBAM hourly values with zero and removing hourly EBAM values with internal RH below 40% showed an increase in agreement. The use of 3 h mean concentrations helped to reduce the mean difference between monitors. Best agreement for this data was with daily mean concentrations where hourly EBAM internal RH <40% was removed (Table 5).

As ambient humidity increased above 65%, hourly mean difference and standard deviation increased (Fig. 4a). Mean difference with external humidity between 55% and 60% was 1.7 μg m^{-3} with a standard deviation of 8.6 μg m^{-3}. When external humidity increased to 60–65%, mean difference and standard deviation increased to 2.4 μg m^{-3} and 10.9 μg m^{-3} respectively. Mean difference and standard deviation were highest when external humidity was 90–95% with a mean difference of 6.9 μg m^{-3} and standard deviation of 18.0 μg m^{-3}. EBAM hourly values were higher throughout all levels of external RH except when ambient RH was 95–100% where the EBAM hourly value was less than the BAM (Fig. 4a). Although the EBAM typically is over-estimating the BAM, when ambient RH is near 100% the EBAM can underrepresent BAM readings representing the complex impacts of temperature, RH, and inlet heating on PM measurements particularly when one

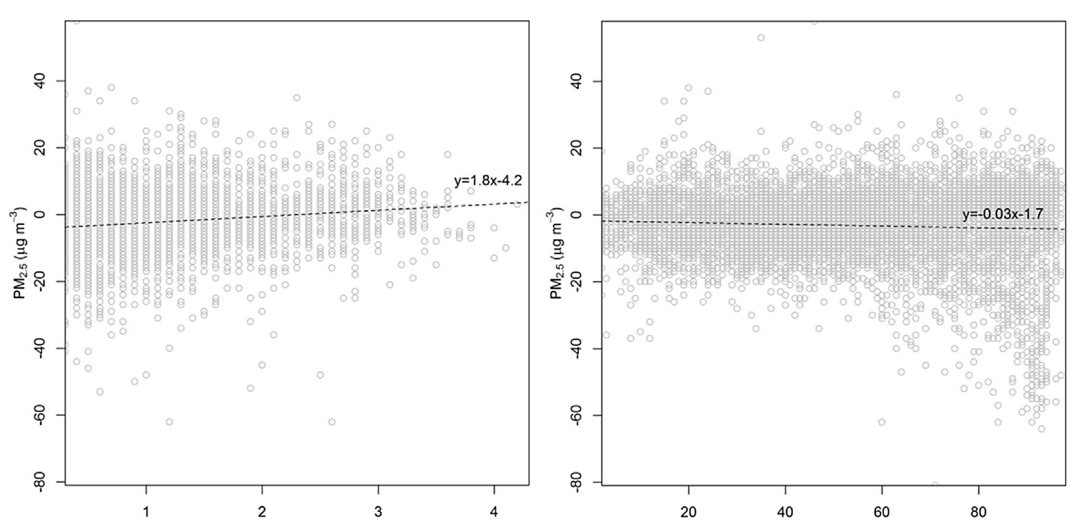

Fig. 2. Difference in hourly concentration of $PM_{2.5}$ (BAM-EBAM) as a function of wind speed and relative humidity.

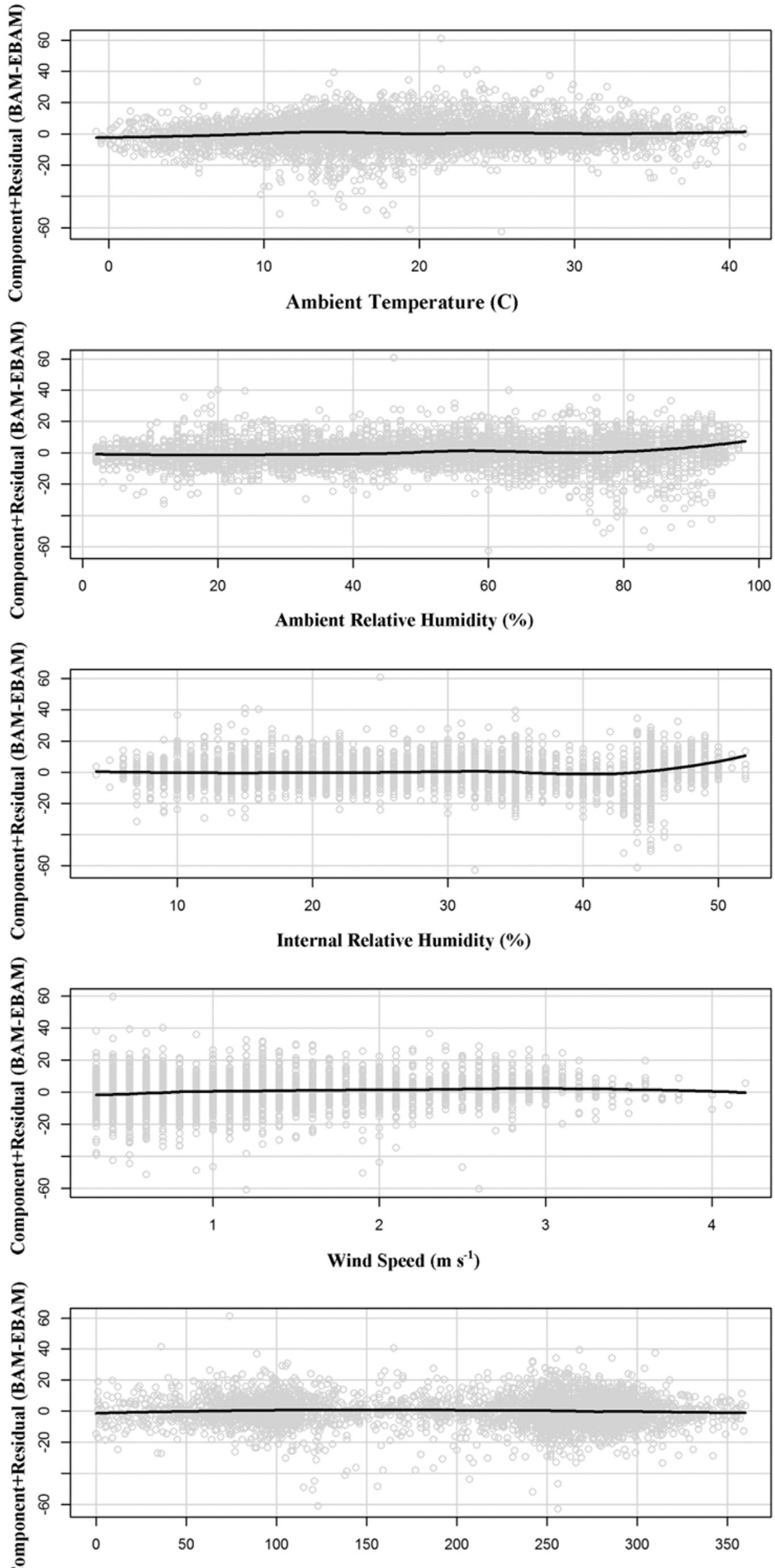

Fig. 3. Partial residual plots with locally weighted scatterplot smoothing (LOESS) line for site environmental variables.

Table 4
BAM and EBAM comparison statistics[a] for hourly, 3 h, and daily data.

Description of hourly data used (daily mean requires 18 valid hours unless otherwise stated)	R^2		
	Hourly	3 h	Daily
All data	0.7042	–	0.9039
Daily mean with no hourly minimum	–	–	0.8983
Negative EBAM hourly values replaced with zero	0.7178	0.7685	0.9075
Ambient RH<60%	0.7594	–	0.9275
Ambient RH<60% and EBAM hourly values of negative replaced with zero	0.7714	0.8184	0.9267
EBAM internal RH <40%	0.7550	–	0.9325
EBAM internal RH <40% and EBAM hourly values of negative replaced with zero	0.7663	0.8230	0.9334

Probability (p) < 0.006 for all R^2 values.

Table 5
Mean difference and limits of agreement statistics with upper and lower limits of EBAM to BAM comparison.

	Mean difference	Standard deviation	Number of samples	Lower limit	Upper limit
All hourly data	2.8	10.8	8972	−18.8	24.4
Negative EBAM hourly values replaced with zero	3.1	10.4	8972	−17.8	24.0
EBAM internal RH <40%	2.4	8.5	6293	−14.6	19.4
EBAM internal RH <40% and EBAM hourly values negative replaced with zero	2.6	8.2	6293	−13.8	19.0
3-h mean	3.2	9	8976	−14.7	21.1
3-h mean with EBAM internal RH <40% and EBAM hourly values negative replaced with zero	2.8	6.7	6295	−10.6	16.1
Daily mean	2.8	5.3	367	−7.8	13.5
Daily mean negative EBAM hourly values replaced with zero	3.2	5.2	367	−7.2	13.5
Daily mean with EBAM internal RH <40% and EBAM hourly values negative replaced with zero	3.1	3.2	193	−3.4	9.6

monitor is operating at or near ambient conditions (EBAM) and the other is in an enclosed climate controlled environment (BAM).

Increased ambient RH is in part corrected by the inlet heater controlled internal RH of both the BAM and EBAM. This internal RH, when operating at levels <45%, had an hourly mean difference typically between 2 and 5 $\mu g\ m^{-3}$. When internal RH was below 5% the BAM was over-predicting the EBAM (mean difference $-4.4\ \mu g\ m^{-3}$) although the smallest number of hourly samples (28) fell into this group. Mean difference was 2–3 $\mu g\ m^{-3}$ when internal RH was 5–40%, increased to 5 $\mu g\ m^{-3}$ at 40–45%, fell to $-4\ \mu g\ m^{-3}$ between 55 and 60% then increased to 5 $\mu g\ m^{-3}$ at 55–60% then increasing to a maximum of 35 $\mu g\ m^{-3}$ when internal RH was 70–75%. Standard deviation of the mean difference was ~8–9 $\mu g\ m^{-3}$ when the internal RH was below 40% (6 $\mu g\ m^{-3}$ at 0–5%; 9 $\mu g\ m^{-3}$ at 5–15% and 30–40%; 8 $\mu g\ m^{-3}$ at 15–30%) then increased to 13 $\mu g\ m^{-3}$ for 40–45%, 7 $\mu g\ m^{-3}$ for 55–60%, and 14 $\mu g\ m^{-3}$ at 60% and higher (Fig. 4b). When internal RH was kept below 40% agreement between the BAM and EBAM was the most consistent.

Relative humidity was the cause of the largest discrepancies between EBAM and BAM measurements. High ambient RH (>90%) produced the largest differences in measurements and standard deviations. Inlet heating of the EBAM to control internal RH reduced high ambient RH impacts. When internal RH was below 40%, EBAM measurements were most consistent when comparing to the BAM (Table 5).

3.4. AQI agreement

Assessing AQI for BAM to EBAM showed an overall increase in AQI when using data from an EBAM. None of the hourly measurements, 3 h mean, or daily mean was above an AQI of unhealthy for sensitive groups. Removing hourly values when the internal RH of the EBAM met or exceeded 40% reduced the disparity between AQI category estimations but also resulted in the loss of roughly a quarter to a third of all measurements.

AQI for hourly BAM to EBAM (when both the BAM and EBAM recorded an hourly reading) was typically categorized good (8862 (98.8%) BAM; 8502 (94.8%) EBAM) with 109 (1.2%) BAM to 460 (5.1%) EBAM moderate and 1 (<0.1%) BAM to 10 (0.1%) EBAM unhealthy for sensitive groups. AQI hour counts were 6215 (98.8%) BAM to 6160 (97.9%) EBAM good, 77 (1.2%) BAM to 128 (2.0%) EBAM moderate, and 1 (<0.1%) BAM to 5 (0.1%) EBAM unhealthy for sensitive groups.

Using 3 h mean concentrations in comparison resulted in 8986 (98.8%) BAM to 8676 (95.5%) EBAM good, 96 (1.2%) BAM to 403 (4.5%) EBAM moderate, and 0 (0.0%) BAM to 3 (<0.1%) EBAM unhealthy for sensitive groups values. When internal RH was below 40%, AQI estimates were more consistent (good 6329 (98.8%) BAM, 6293 (98.3%) EBAM; moderate 72 (1.2%) BAM, 108 (1.7%) EBAM) with 0 (0.0%) BAM and 1 (<0.1%) EBAM unhealthy for sensitive groups.

Daily AQI comparison resulted in 203 (55.3%) BAM to 138 (39.5%) EBAM good, 164 (44.7%) BAM to 138 (59.1%) EBAM moderate, and 0 (0.0%) BAM to 5 (1.4%) EBAM unhealthy for sensitive groups. Controlling for internal RH (including only hourly data where the internal RH was below 40%) produced 136 (70.5%) BAM to 85 (44.0%) EBAM good days, 57 (29.5%) BAM to 108 (56.0%) EBAM moderate, and no unhealthy for sensitive groups. Removing these hours reduced the number of daily comparisons by 47% (367–193 days) and invalidated the highest daily AQI (unhealthy for sensitive groups) estimates which were only found with the EBAM data.

A plot of the distributions, using a normal (or Gaussian) kernel function with bandwidth 0.9 times the minimum of the standard

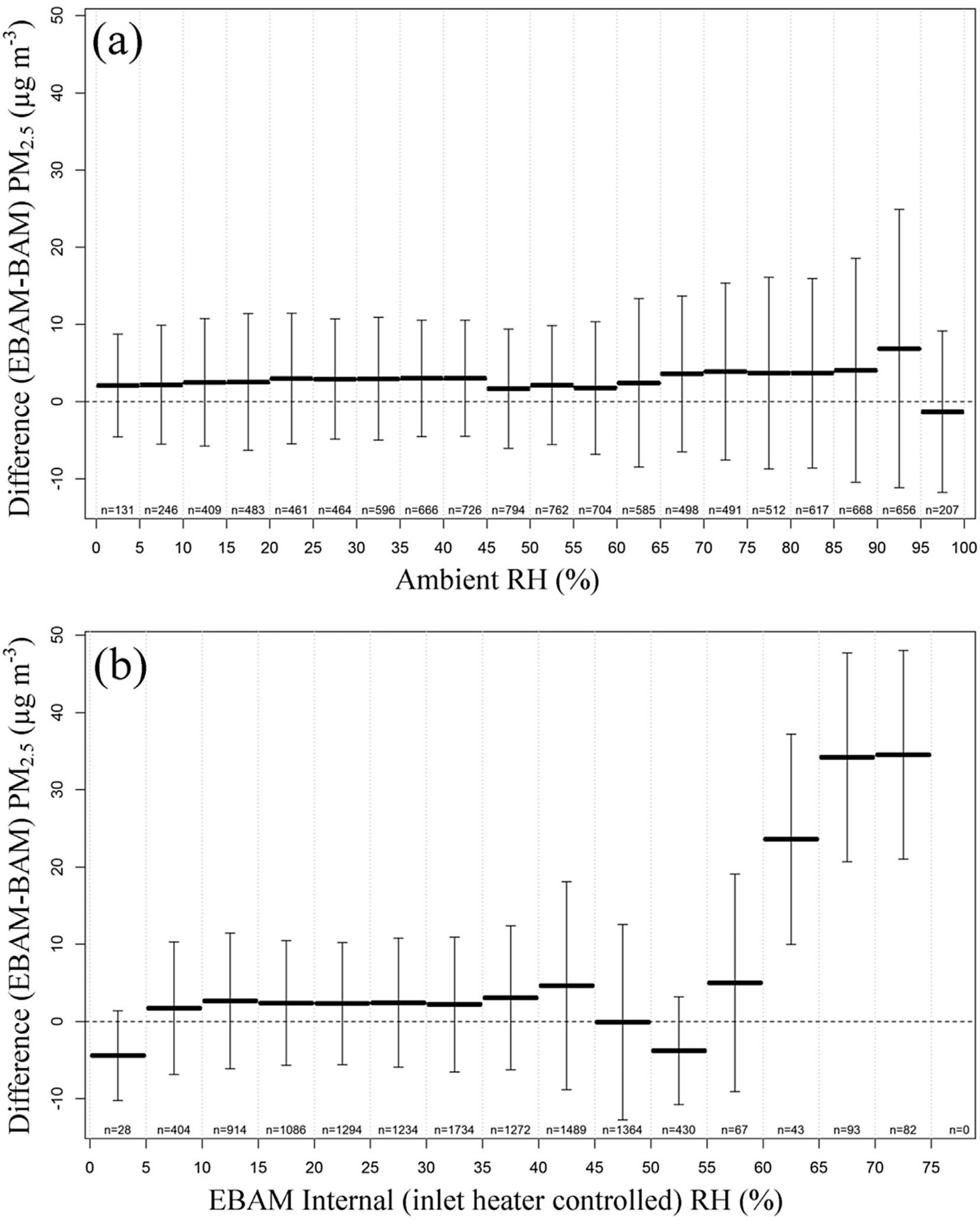

Fig. 4. $PM_{2.5}$ agreement using difference from the mean (+/− 2 standard deviation error bars) of EBAM and BAM with (a) similar external RH and (b) similar internal (inlet heater controlled) RH. RH levels have a range of ± 2.5% with n number of hourly values.

deviation and the interquartile range divided by 1.34 times the sample size to the negative 1/5 power (Silverman, 1986) for a kernel density estimation as a non-parametric representation of the density of the hourly $PM_{2.5}$ variable, helped us to visualize the increase in hourly measurements when comparing an EBAM to a BAM (Fig. 5). The overall hourly EBAM distribution is noticeably higher than the BAM. The impact of setting negative hourly EBAM values to zero illustrates the shift in values. Including only EBAM hourly values where the internal RH is less than 40% produces a distribution where the highest hourly values (>~30 μg m^{-3}) run closer to the BAM.

4. Discussion

Temporary mobile particulate monitors are being widely utilized for smoke monitoring throughout California. The EBAM is an excellent source allowing for mobile, temporary monitoring of $PM_{2.5}$. The EBAM has been widely deployed during emergency events and provides an indispensable source of additional data for public health protection that otherwise would not be available. Analysis in this paper is intended to document and understand limitations of using the EBAM; in particular when being used for

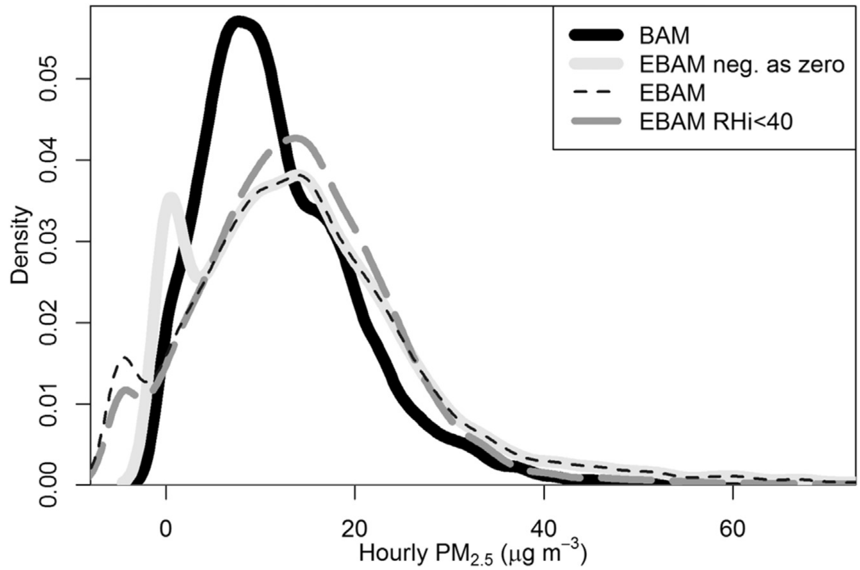

Fig. 5. Distribution of hourly $PM_{2.5}$ using kernel density estimation.

smoke management decisions through direct comparison to monitors used for federal and state compliance.

The BAM is widely used by California air regulators as the "gold" standard for compliance monitoring, thus the EBAM (or any other monitor) being used for smoke monitoring and management has a default assumption of comparability. EBAMs are particularly useful when monitoring large, high intensity, full suppression wildfires when "conservative" estimates allow public health officials to error on the side of caution and indeed our analysis suggests that using EBAMs is an excellent way to provide additional targeted data and over-represent ground concentrations of $PM_{2.5}$ when compared to BAMs. This monitoring assumption can be useful when determining compliance and impacts from large full suppression wildfires (Preisler et al., 2015) while consistent over-estimation may be hindering a more nuanced policy approach to smoke management desperately needed for effective wildland fire management in the western United States (North et al., 2015a). Wildland fire being used for fuel reduction and ecological benefit through prescribed and natural ignition often localizes smoke impacts heavily dependent on non-regulatory temporary monitors (Schweizer and Cisneros, 2014) where even slight increases in the estimated background concentrations can determine fire management actions and perpetuate entrenched disincentives for full suppression (North et al., 2015a). While this policy debate involving fuel loading, fire ecology, and smoke management continues (Boer et al., 2015; North et al., 2015b; Thompson et al., 2015), understanding smoke monitoring equipment strengths and weaknesses will help inform the discussion.

The supplemental data being generated by EBAMs provides an invaluable addition to the urban FEM monitoring network during a wildland fire. Current advisories and assessment of smoke impacts on public health at many times are reliant on the EBAM. Although these instruments are providing data in areas where it is not feasible to establish a permanent monitor, our data suggests the EBAM is over representing the exposure levels and is of particular concern at lower concentrations where EBAM measurements are being relied on for smoke management actions. This is particularly apparent when higher humidity and more stagnant air masses may be combining to introduce error into the EBAM hourly measurements.

RH is an important meteorological component to wildland fire dynamics with increasing RH reducing burn severity (Collins et al., 2007) and often aiding containment of extreme wildfires. Although high humidity may not be present in all wildland fire events, evening and overnight RH can exceed 40% (when BAMs and EBAMs begin humidity control heating) even during large high intensity wildfires (Peterson et al., 2015). Higher RH is often typical at the end of a managed fire where emissions are primarily generated from interior smoldering during burn down. RH is also a component to a prescribed fire plan and is typically stipulated for levels that both moderate fire behavior and provide desired fuel consumption. Ambient RH can be above 40% for much of a prescribed fire (Knapp et al., 2005) with lower RH being specified for the desired burn intensity and higher RH utilized as a controlling mechanism. RH can frequently exceed the 40% internal RH set point for the heater controlled sample of the BAM and EBAM during a wildland fire.

High RH can be a particular concern at the end of a full suppression or managed natural ignition wildland fire when meteorological conditions can include high humidity and precipitation events. Error introduced from high RH almost solely manifests as a higher hourly $PM_{2.5}$ readings on the EBAM. Using this data can lead to incorrectly including high values in data analysis of smoke impacts. Although an erroneously high measurement may be considered conservative in the protection of human health, and indeed be helpful to punctuate the seriousness of an extreme air quality event, the application of consistently over-estimating $PM_{2.5}$ in a fire adapted ecosystem where smoke is inevitable can easily create unintended consequences at lower concentrations where misrepresentation of ground level $PM_{2.5}$ results in suppression biased smoke management actions.

Over-estimation of smoke impacts to $PM_{2.5}$ for the EBAM in large part can be remedied by both using 24 h average concentrations while assessing impacts and exposure of a given population and giving extra consideration to internal RH of the EBAM before including individual hourly values in the calculation. Hourly AQI consistency between monitors was increased simply when we only included hourly data when the internal RH was <40% and improved further with 3 h averaging.

It is important to recognize limitations and uncertainties when comparing fine particulate monitors. $PM_{2.5}$ measurements included in this study reflect the accuracy and precision expected when temporary monitors are used in the field to measure $PM_{2.5}$ at lower concentrations (hourly concentration range 0–102 μg m^{-3}). With agreement encompassing a range that includes about half the hourly data points and a quarter of the range of daily values, the possibility exists that at high concentrations of $PM_{2.5}$ the effect is similarly large and would encompass a wider concentration disparity which could lead to even greater differences between instruments when determining AQI for high levels of smoke.

Data presented in this study agree with the documented effects from humidity on PM mass measurements (Heber et al., 2006; Tsyro, 2005). For best agreement between monitors, heater controlled internal RH should be held below 40% and special consideration should be given when ambient RH rises above 65% with any readings where the ambient RH is above 90% being used with caution as the high external RH is likely reducing the effectiveness of the internal heater manifesting in the EBAM overestimating $PM_{2.5}$ concentrations. When the RH remains above the EBAM inlet heater set point for extended periods of time this effect seemed more pronounced.

The hourly concentrations in this data are relatively low (maximum BAM 93 μg m^{-3}, EBAM 102 μg m^{-3}) with good AQI. Our data shows instrument selection and RH have an impact on calculated AQI for $PM_{2.5}$. AQI estimates during hours of high humidity typically were higher for an EBAM than a BAM giving rise to erroneously high estimates from areas relying on data from an EBAM. Concentrations of $PM_{2.5}$ are frequently much higher during incidents with large emissions from full suppression wildfire. More study is needed to compare particulate monitors at higher concentrations to ensure AQI is being adequately reflected between temporary and permanent monitors.

While EBAMs provide useful information to help determine specific hour or hours during a day when air quality poses the largest threat to human health, the hourly data from EBAMs should be used with caution when comparing to BAM measurements. Although inlet heating to control relative humidity can minimize this effect (Huang, 2007), EBAM inlet heaters did not always keep internal RH below 40%. Invalidating hourly EBAM data with high internal RH hours when calculating daily averages helped increase agreement with BAM measurements. Additionally, we saw an overestimation of AQI when EBAMs are compared to BAMs. This is particularly true using hourly measurements for AQI. We recommend using hourly (or 3 h mean) with extreme caution when providing near real time public health advice. Daily AQI is a much more appropriate matrix when using EBAMs for assessing air quality for $PM_{2.5}$.

5. Conclusions

Mobile monitors such as EBAMs are an incredibly useful source of data for research and public health protection during short duration events. The increased use of mobile monitors during temporary events such as wildfire makes understanding the limitations of mobile monitors important to providing accurate information to the public.

The EBAM is susceptible to over-estimation of $PM_{2.5}$ concentrations when ambient RH is high. Agreement between BAMs and EBAMs are weakest when internal RH is above 40% with EBAMs typically over-estimating $PM_{2.5}$ concentrations that additionally result in elevated EBAM estimations of AQI category. Co-located monitoring during wildland fire smoke events would be useful in determining the presence and extent of this impact at higher $PM_{2.5}$ concentrations than were presented in this study.

The co-location comparisons in this study suggest EBAM hourly measurements are not precise enough to warrant comparison to an EPA certified BAM site. Additionally, EBAM hourly AQI estimates should be used with extreme caution as hourly (and 3 h mean concentrations) often result in over-prediction even at the lower concentrations included in this study. Thus, it is recommended to use only daily AQI estimates when using the EBAM. EBAM daily mean concentrations calculated using only hourly $PM_{2.5}$ concentrations when the internal RH is below 40% are the most appropriate measurements to use when comparing $PM_{2.5}$ concentrations and AQI estimates to BAM data.

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