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An evaluation of the U.S. EPA's correction equation for Purple Air Sensor data in smoke, dust and wintertime urban pollution events

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Abstract

PurpleAir Sensors (PASs) are low-cost tools to measure fine particulate matter (PM) concentrations and are now widely used, especially in regions with few regulatory monitors. However, the raw PAS data have significant biases, so the sensors must be calibrated to generate accurate data. The U.S. EPA recently developed a national correction equation and have integrated corrected PAS data onto its AirNow website. This integration results in much better spatial coverage for $PM_{2.5}$ across the U.S. The goal of our study is to evaluate the EPA correction equation for three different types of aerosols: typical urban wintertime aerosol, smoke from biomass burning, and mineral dust.

We identified 50 individual pollution events, each having a peak hourly $PM_{2.5}$ concentration of at least 47 µg m^{-3} and a minimum of 3 hours over 40 µg m^{-3} and characterize the primary aerosol type as either typical urban, smoke or dust. For each event, we paired an PAS sampling outside air with a nearby regulatory $PM_{2.5}$ monitor to evaluate the agreement. All 50 events show statistically significant correlations (R values between 0.71-1.00) between the hourly PAS and regulatory data, but with varying slopes. Using the standard EPA correction for the typical urban and smoke aerosols, we find average slopes of 1.00 and 0.99, respectively. This means that the standard EPA correction is highly effective at generating accurate data for these aerosol types. For heavy smoke events, we find a small change in the slope at very high $PM_{2.5}$ concentrations (>600µg m^{-3}), suggesting a ~20% under-estimate in the corrected PAS data at these extremely high concentrations. For dust events, while the PAS and regulatory data still show significant correlations, the PAS data using the standard correction underestimates the true $PM_{2.5}$ by a factor of 5-6.

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We also examined several years of co-located regulatory and PAS data from a site near Owens Lake, CA, which experiences high concentrations of $PM_{2.5}$ due to both smoke and dust. For this site we find similar results as above; the PAS corrected data are accurate in smoke, but are too low by a factor of 5-6 in dust. Using these data we also find that the ratios of PAS measured PM_{10} to PM_1 mass and 0.3 µm to 5 µm particle counts are significantly different for dust compared to smoke. Given the ability of the PAS data to identify dust aerosols, we propose a modified correction algorithm that significantly improves the PAS data for dust events.

Introduction

Low-cost air sensors are becoming a ubiquitous way for the general public to measure local air quality. There are now thousands of these sensors publicly reporting data in real time to the PurpleAir map (https://purpleair.com/map). As one example, there are now more than 700 active PurpleAir sensors (PAS) in the Puget Sound region of Washington State (from Tacoma to Everett), compared to ~15 regulatory monitors in the same area. This provides an enormous increase in spatial information on $PM_{2.5}$ (particulate matter with diameters less than 2.5 µm). However, there are no clear performance standards for accuracy or precision of low-cost sensors. Several studies have examined the performance of low-cost sensors, including the PAS (Singer and Delp, 2018; Li et al., 2020; Ardon-Dryer et al., 2020; Manibusan and Mainelis, 2020; Tryner et al., 2020). The PAS uses the Plantower PMS5003 laser particle counter to count particles that scatter light in the optical range (particles greater than about 0.2 µm in diameter). Most outdoor PAS include two identical PMS5003 sensors that can be compared to enhance quality control. Tryner et al. (2020) evaluated three low-cost particulate matter sensors, including the PMS5003 by exposing them to five different types of aerosols in the laboratory. They found that the ratios of PMS5003-reported to filter-derived $PM_{2.5}$ mass concentrations were inversely proportional to mass median diameter (MMD). Wood smoke had the smallest MMD, 0.42 µm; its PMS5003 $PM_{2.5}$ mass had a mean that was 2.5 times the filter-derived mass. Conversely, oil mist had the largest MMD at 2.9 µm; its PMS5003 PM2.5 averaged only 0.23 times the filter-derived PM2.5. These lab results are consistent with the physical-optical model developed for the PMS5003 by Ouimette et al. (2022). The model predicted that the PMS5003 response decreases relative to an ideal nephelometer by about 70-90% for particle diameters ≥1.0 µm. This is a result of using a laser that is polarized, the angular truncation of the scattered light, and particle losses (e.g., due to aspiration) before reaching the laser. Their model predicted that the PMS5003 would underestimate $PM_{2.5}$ for dust particles by approximately 70-90%, depending on the coarse particle size distribution.

The Plantower sensor reports PM mass concentrations in three bins (PM_1, $PM_{2.5}$, and PM_{10}) and particle counts in 6 size bins (>0.3, >0.5, >1, >2.5, >5 and >10 µm), presumably based on the pulse height of the scattered radiation, although the exact procedure is not documented by Purple Air. A number of field and laboratory studies have found that the PMS5003 size distributions are not correct. Several studies have reported that the PMS5003 tends to create an invariant normalized size distribution, independent of the actual size distribution and concentration (Tryner et al., 2020; He et al., 2020; Kuula et al., 2020, Ouimette et al., 2022). However, the PMS5003 normalized size fractions above 1 µm increased by a factor of 2–5 in one high-$PM_{2.5}$ windblown dust episode observed at Keeler,

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California (Ouimette et al., 2022). So, at present, there remains some ambiguity over how the PAS reported $PM_{2.5}$ concentrations respond to different aerosol types.

Aerosol size distributions can vary considerably depending on the source type. Previous studies have shown that the aerosol size distributions for smoke events are similar to the distributions in typical urban pollution events, with geometric mean diameter of around 0.2 – 0.3 µm (Kleeman et al., 1999; Laing et al., 2016). The mass fraction of $PM_{2.5}/PM_{10}$ smoke and urban pollution is also similar at 0.55-0.75 (Xu et al., 2017). Dust events are known to have size distributions that are shifted towards larger particles, compared to typical urban and smoke aerosols. Jiang et al. (2018) report an average $PM_{2.5}/PM_{10}$ ratio of 0.1 for dust events in China. In addition, aerosol particles from some cooking methods, such as barbeque, may also have a size distribution that is shifted to larger sizes (Kleenman et al., 1999; Song et al., 2018). If this is correct, then this may have implications for using PAS data to examine indoor air quality.

The South Coast Air Quality Management District (South Coast AQMD) has completed a rigorous evaluation of a variety of sensors, including the PAS (South Coast AQMD, 2015). This evaluation has shown that the PAS gave precise results, showed little response to temperature or humidity, and had relatively small variations between units. The U.S. Environmental Protection Agency (EPA) also provides information about these sensors via its "Air Sensor Toolbox for Citizen Scientists, Researchers and Developers" portal (U.S. EPA, 2022). All of these evaluations have demonstrated that the raw PAS measurements are precise, but biased high compared to regulatory $PM_{2.5}$ measurements. Several groups have developed correction equations for the PAS measurements. The Lane Regional Air Protection Agency (LRAPA), the University of Utah, and the EPA have empirical corrections for $PM_{2.5}$ and these can be implemented directly on the PurpleAir website (PurpleAir, 2021). Barkjohn et al., (2020) (hereafter referred to as Barkjohn) did a comprehensive evaluation of PAS $PM_{2.5}$ data against regulatory $PM_{2.5}$ data and developed a U.S. wide correction equation:

$$\textbf{Corrected PAS } PM_{2.5} = \textbf{raw PAS } PM_{2.5} \textbf{ data (CF=1) *0.52 - RH*0.085 + 5.71} \quad (1)$$

The LRAPA and the Barkjohn corrections are in close agreement, whereas the University of Utah correction gives somewhat higher values. The EPA recently incorporated PAS data using the Barkjohn correction into its national Fire and Smoke Map (AirNow, 2021). For our analysis, we will refer to the Barkjohn correction equation as the "standard correction equation". Note that the PAS data can be downloaded either as raw data (CF=1) or with various calibration factors applied (e.g., LRAPA, EPA, etc). In the present analysis, we start from raw data with CF=1.

Because many PAS devices are now installed around the world, both outside and inside homes and workplaces, they can experience a wide range of aerosol types. Thus it is essential to understand the accuracy and precision of the PAS for various aerosol events, which could differ based on the particle size distribution or other aerosol characteristics. In this study, we evaluated the standard correction equation (Barkjohn et al., 2020) for 50 different aerosol pollution events, encompassing typical urban aerosols, as well as smoke and dust aerosols. Our goals are:

1. Evaluate the accuracy of the standard correction equation for each aerosol type;

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2. Examine whether the correction changes at very high $PM_{2.5}$ levels (e.g. > 250 µg m^{-3});
3. Identify whether the PAS data can provide an indication of the aerosol type and, if so, whether this information can be used to improve the correction algorithm.

Below we first describe data treatment and events and aerosol type identification. Then we report on results comparing regulatory and PAS observations for different aerosol types for 50 short term pollution events. We also use a longer time series from a single site (Keeler, CA) that experiences frequent high dust and smoke pollution episodes. Our results demonstrate that the PAS sensors can give accurate $PM_{2.5}$ data in urban pollution and smoke, but more work is needed to develop an improved correction for dust aerosols.

Methods

Part I

PAS data was downloaded for each sensor from the PurpleAir website (map.purpleair.com). The raw data (CF=1) was downloaded as hourly averages. Regulatory PM data for the nearest monitoring site was downloaded from the EPA "Air Data" website (https://www.epa.gov/outdoor-air-quality-data) or the AirNow-Tech website (airnowtech.org), except for the monitoring site at Portland Cully Helensview School in Portland, OR (AQS Id 410512011), which was downloaded from the Oregon Department of Environmental Quality website (https://www.oregon.gov/deq/aq).

For each paired PAS-regulatory site, we identified an intense pollution event that had an hourly peak $PM_{2.5}$ value at the regulatory site with hourly $PM_{2.5}$ values >40 µg m^{-3} for at least 3 hours. We also required that there be a good correlation between the regulatory and PAS data. For the 50 events we analyzed, the correlation coefficients between the regulatory and PAS corrected data ranged from 0.77 to 0.996. For each pollution event, we identified the most likely source of elevated aerosols: typical urban-wintertime pollution, biomass burning smoke or dust. Table 1 summarizes the method used to characterize each pollution event. Table 2 gives a summary of the events and SI Table S1 show details on each of the 50 individual events, including PAS site, regulatory site, event dates and distance between the two sites. The average distance between the PAS and regulatory sites was 5.4 km, with a range of 0-15 km. There is not a significant relationship between the correlation coefficients and distance between sites.

Typical urban pollution events were identified for the non-wildfire season (winter months) and with no evidence of smoke or dust. The PM sources for those events reflect typical urban, wintertime pollution (vehicles, power plants, industry, and residential wood combustion), and the $PM_{2.5}$ mass is dominated by particles with diameters in the range of 0.30–0.60 µm (Zhang et al., 2010; Herner et al., 2005). The typical urban pollution events had peak hourly $PM_{2.5}$ values at the regulatory site of 47-259 µg m^{-3}.

Smoke events were identified by elevated $PM_{2.5}$ during the summer fire-season and confirmed using the Hazard Mapping System (HMS) Fire and Smoke Product (Rolph et al., 2009; Kaulfus et al., 2017). The HMS product is

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derived from multiple satellite images and updated multiple times each day. Details on the HMS product are given in the references above. The HMS imagery was obtained via the AirNow-Tech website. The smoke events had the highest peak $PM_{2.5}$ values at the regulatory sites with peak hourly values of 60-713 µg m^{-3}.

Dust events were identified by examining large scale spatial patterns of $PM_{2.5}$, media reports and the measured $PM_{10}/PM_{2.5}$ ratios from regulatory sites, if available. In Part I of our analysis all 6 dust events occurred during the well-known June 2020 Saharan dust cloud that was transported to the U.S. and impacted surface concentrations across the Southern U.S. (Francis et al., 2020; Lovely et al., 2021; Pu and Jin, 2021). This event brought huge amounts of dust to the southern U.S. and resulted in daily average $PM_{2.5}$ concentrations of 60-103 µg m^{-3} at many locations. The six dust events included in our analysis had peak hourly $PM_{2.5}$ values at the regulatory site of 52-72 µg m^{-3}. Figure S1 shows the impact of this dust on $PM_{2.5}$ across the southeastern U.S.

In total, we identified 50 events as either typical urban, smoke, or dust, lasting from 24 to 528 hours. We verified that each had an operating PAS and a nearby regulatory monitoring site. For typical urban pollution 16 cases were identified, with the majority (13) being located in California, and the remainder in Utah. We identified 28 smoke cases, with locations in Alaska, California, Idaho, Oregon, and Washington. Six dust cases were identified, with locations throughout the southeast U.S. Of the 50 events identified, 17 have co-located PM_{10} data (3-urban, 8-smoke and 6-dust).

Data quality control

The data were quality controlled and screened using five criteria.

1. Since most PAS contains two sensors, A and B, we compared data from the two sensors and the data were only used if the values were within 30%. Most values are much closer than this, with an average difference of 10% across all events considered (4.6% for the Keeler, CA PAS data).
2. The PAS raw A and B values were averaged and excluded if less than 1 µg m^{-3}.
3. The PAS values were corrected using the standard correction and only included if greater than 2 µg m^{-3}. Note that at high relative humidity, negative values can occur.
4. Regulatory $PM_{2.5}$ data must be greater than 1 µg m^{-3} (a number of 0 and negative values are in the EPA's $PM_{2.5}$ data records).

After screening, the PAS data were corrected using the EPA correction (equation 1). We evaluated the raw and corrected PAS data using the same linear relationship:

$$\textbf{Regulatory data} = \textbf{Slope} * \textbf{PAS data (raw or corrected)} + \textbf{Intercept} \quad (2)$$

Generally, the intercepts were small (a few µg m^{-3}), so we can interpret the slopes as giving the overall indication of bias to the data. A slope near 1 with a zero intercept would indicate no bias. A slope <1 indicates that the PAS data (raw or corrected) are biased high compared to the regulatory data, a slope >1 indicates the PAS results (raw or corrected) are biased low compared to the regulatory data.

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Part II.

To further understand the nature of the PAS response to dust aerosol, we also used data from Keeler, CA near Owens Lake. Owens Lake is now a dry lake bed due to diversion of its primary water source, the Owens River, to Los Angeles. As a result, the dry lake bed is now one of the largest sources of dust in North America (Cahill et al., 1996; Gilette et al., 1997) and the region experiences many significant dust events each year. With increasing drought, it appears that the dust flux from Owens Lake is increasing (Borlina and Rennó, 2017). We use regulatory $PM_{2.5}$ and PM_{10} data from February 2019-May 2022 from a California Air Resources Board (CARB) site located in Keeler, CA as well as a nearby PAS site. Regulatory $PM_{2.5}$ and PM_{10} for the Keeler site are from the Great Basin Unified Air Pollution Control District website (https://www.gbuapcd.org/). For the Keeler PAS data, as in Part I, we use the mean of channels A and B, which have a mean difference of 4.6%. More details on the Keeler site are given in Table S2. For the Keeler, CA analysis we did not specifically identify event types. Instead we consider only hours where the Keeler regulatory $PM_{2.5}$ >25 µg m^{-3}, which provides 1366 hours of data. We also restrict the analysis of the Keeler data to hours where regulatory PM_{10} exceeds $PM_{2.5}$ by at least 0.1 µg m^{-3} and where simultaneous regulatory and PAS data are available. This yields 1257 hours of data with mean $PM_{2.5}$ and PM_{10} concentrations from the regulatory monitors of 59 and 118 µg m^{-3}, respectively.

Results

Part I: Event analysis

Figure 1 shows PAS and regulatory data during a major smoke event in Washington State during July-August 2021. The regulatory $PM_{2.5}$ exceeded 200 µg m^{-3} at this site. This figure shows the well-known overestimation of $PM_{2.5}$ concentrations measured by the PAS during smoke events and also shows that the standard EPA correction yields excellent bias correction of the data. Regression was calculated twice for each event: (i) using ordinary linear regression (OLR) and (ii) using reduced major axis regression (RMA). In practice, there was rather little difference in the results, given that the correlation coefficients were fairly high. Both are reported in Table S1. For the smoke event shown in Figure 1 (event #44), the OLR slope is 0.81 and the RMA slope is 0.86. Figure 1 also shows data from a dust period in June 2020 (event #45). In contrast to the smoke event, raw PA data are already well below the regulatory data and the standard correction only makes the bias greater. While there is still a good correlation between the regulatory and PAS data (R value of 0.82), the OLR slope is 6.43, indicating that the standard correction is under-estimating the true concentrations by a factor of 6 or more (note that this figure shows the PA data on the right axis). Table 2 summarizes the results for all 50 events. Table 2 also lists p values, where p values <0.05 indicate a statistical difference at 95% confidence.

Table 3 summarizes the results by aerosol type and includes all hourly data for each identified aerosol type. For urban, smoke and dust aerosols, the slope of regulatory $PM_{2.5}$ versus the PAS corrected data were 1.02, 1.08 and 4.98, respectively, using all hourly data of each type. This indicates that the standard equation yields excellent bias

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correction for typical urban and smoke events, but that it generates a large negative (low) bias for dust events. Using the standard EPA correction on the PAS data during dust events gives values that are low by approximately a factor of 5-6.

We also want to examine whether there is evidence that the PAS data respond differently at very high PM concentrations. Figure 2 shows the corrected slopes versus peak $PM_{2.5}$ concentration for the urban, smoke and dust events. While the slopes for urban and smoke events are generally close to 1, there is a small, but statistically significant trend in the slope with maximum $PM_{2.5}$ concentrations for the urban and smoke cases. The results suggest that at the highest concentrations (600-700 µg m^{-3}), the standard PAS correction is reporting data that are 20-30% low compared to the regulatory.

We show above that the PAS data, using the standard correction, is substantially under-reporting $PM_{2.5}$ concentrations during dust events. The next question is whether the PAS data can give some information about dust events (i.e., the presence/absence of dust), despite significant issues with the reported size distribution (Ouimette et al., 2022). To address this question, we calculated the slopes of the PM_1 to PM_{10} mass ratios and the 0.3 µm to 5 µm particle counts ratio, both using the PAS data for each event. The results are reported by event type in Table 2. The results show that the PAS reports a greater fraction of coarse mass and proportionally more larger particles, compared to the 0.3 µm particles, in the dust aerosols, compared to urban or smoke aerosols. Both the PM_1/PM_{10} mass ratio and the 0.3 µm to 5 µm particle counts ratio increases in the order dust<smoke<urban. These differences are statistically significant for urban versus dust, but not for smoke versus dust at the 95% confidence level. These relationships will be explored further below in Part II of this analysis.

We have also looked at the coarse aerosol fraction (CAF) for these events using the regulatory data. We define the CAF as:

$$\mathbf{CAF = (PM_{10} - PM_{2.5}) / PM_{10}} \quad \mathbf{(3)}$$

Figure 3 shows the CAF, calculated using both the regulatory data and the PAS raw data for all hours for the 17 events with both $PM_{2.5}$ and PM_{10} data. For the PAS data, we use the raw values for both $PM_{2.5}$ and PM_{10}, since there are no known correction algorithms for the PM_{10} data. Several things are apparent in Figure 3. First, the CAF values using the regulatory data are much higher than CAF values obtained from the PAS data. Nonetheless, both the regulatory and PAS data show the expected pattern of higher CAF in dust compared to the other aerosol types. In addition, the number of data points is much higher for the PAS, due to the relative sparsity of regulatory PM_{10} data. We note that these relationships change very little if the PAS data are restricted to the same times as the regulatory data.

Part II

Here we use the multi-year dataset from the Owens Lake/Keeler, CA site. Figure 4 shows a histogram of the CAF based on the regulatory data. There is a clear bimodal distribution, indicating two very different aerosol types during these pollution events. Given that Keeler is ca 150 km from the urban areas of Bakersfield and Fresno, CA,

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this aerosol is likely either dust generated from Owens Lake, or smoke from the many California wildfires during 2019-2022. For the points with CAF<0.5 (n=1013 hours), the vast majority (99%) occurred in August-October 2020 or August-September 2021, both times when large fires were burning in central CA and influencing air quality across the region. Thus, it is reasonable to conclude that those hours with CAF<0.5 are predominantly wildfire smoke and those with CAF>0.7 are predominantly dust. In contrast to the smoke data, the dust data tend to occur in the winter and spring periods. There are a relatively smaller number of points with 0.7>CAF>0.5 and these appear to have a mixed character, as shown below.

Table 4 and Figures 5 and 6 show results grouped by the CAF calculated using the regulatory data. For all values of CAF below 0.5, there is a similar behavior. For this group the PAS $PM_{2.5}$ with standard correction shows a good fit to the regulatory $PM_{2.5}$. There is also a fairly consistent set of PAS measured PM_1/PM_{10} and 0.3 µm to 5µm count ratios. For the values with CAF>0.7, there is similar consistency in the PAS measured ratios (PM_1/PM_{10} and 0.3 µm/5µm counts), but for this group the PAS standard correction significantly under-estimates the regulatory concentrations. For the group with CAF<0.5 and CAF>0.7, the primary aerosol types are smoke and dust, respectively. The middle group (0.7>CAF>0.5) appears to have a more mixed character.

Figure 5 shows a plot of the regulatory $PM_{2.5}$ vs PAS $PM_{2.5}$ with the standard correction, sorted by these three groups (CAF<0.5, 0.7>CAF>0.5 and CAF>0.7). For the smoke aerosols, the PAS with standard correction shows a slope of 0.99 and an R^2 of 0.92, whereas for the dust aerosols, the slope is 5.6, similar to the slopes shown in Table 2 (5.5) and Table 3 (5.0). Thus, we conclude that for dust aerosols the PAS standard corrected values show a 5-6x underestimate of the $PM_{2.5}$ regulatory values. The mixed aerosols show behavior that is more difficult to characterize, with some showing more similarity to dust and others to smoke.

Table 4 also shows that the ratios of PM_1/PM_{10} and the 0.3 µm/5µm counts, as measured by the PAS, vary by CAF. This provides a tool that can identify dust aerosols and apply a separate correction. We explored both the PM_1/PM_{10} and the 0.3 µm/5µm count ratios and found better separation using the count ratios. Table 4 shows that using a 0.3 µm/5µm count ratio of somewhere between 150-250 will provide the best separation of dust and mixed aerosols. By examination of various plots of regulatory vs corrected PAS for the Keeler, CA data, we found an optimum value of 190. The value of 5.6 comes from the slope of the dust aerosols in Figure 5. So this leads to a new correction equation that depends on PAS measured values:

If PAS 0.3 µm/5µm>190, use standard correction;

If PAS 0.3 µm/5µm<190, use standard correction * 5.6 (4)

Figure 6 shows a plot of the Keeler, CA regulatory $PM_{2.5}$ versus PAS $PM_{2.5}$ with the new correction applied. There is very little change to the smoke data as most of these points have PAS measured 0.3 µm/5µm counts >190. For the dust aerosols, the majority of the data points are now much closer to the regulatory values. The mean bias for the points with CAF >0.7 is now 1.3 µg m^{-3} compared with 51.4 µg m^{-3} for the dust data using the standard correction. Figures S4 and S5 show how the choice of 0.3 µm/5µm ratio impacts the analysis. Using a higher threshold in equation 4 results in identifying some points (smoke) with corrected $PM_{2.5}$ values that are substantially

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too high. Using a lower threshold in equation 4 results in missing some dust points and, for those points, generating PAS corrected $PM_{2.5}$ values that are too low. While using a value of 190 in equation 4 does miss a small number of dust points, it appears to be the best balance in finding and correcting the dust data points for this location.

Equation 4 was developed based on data from one site that has strong dust and smoke occurrence and with the sensors in close proximity (Keeler). We apply equation 4 to the dust events identified in Part I and find a significant improvement in the mean bias (6.1 µg m^{-3} using equation 4, in contrast to 24.2 µg m^{-3} using the standard correction). However the requirement that the A and B channel 0.3 and 5 5 µm counts agree within 30% reduces the number of data points by more than half. This largely reflects the low aerosol numbers in the 5 µm channel and therefore relatively large variability, due to the factors identified in Ouimette et al. (2022). It is possible that there is a better dust correction algorithm that could integrate more of the PAS measured parameters. Given that the PAS substantially underestimates $PM_{2.5}$ concentrations during dust events using the standard correction, we propose that further evaluation of this new algorithm is warranted.

Conclusion

PAS are now ubiquitous around the world and far outnumber the more accurate, regulatory-grade instruments for $PM_{2.5}$. These low cost sensor data are proving to be highly valuable for a variety of analyses, but especially for improving our understanding of the spatial distribution of $PM_{2.5}$. However, to use these data, it is essential to understand the measurements. Using the EPA's correction algorithm for PAS data, we find that the sensors give reasonably accurate results for $PM_{2.5}$ for typical urban-wintertime pollution and smoke events, but give concentrations that are a factor 5-6 too low for dust events. Using the PAS ratios of PM_{10} to PM_1 mass concentrations and 0.3 µm to 5 µm counts we find that there are significant differences for smoke and dust aerosol types. Using this result we propose a new PAS correction algorithm that significantly improves the correction for dust aerosols and does not change the results for smoke aerosols. While this new equation needs further evaluation, it demonstrates that an improved PAS correction algorithm could be developed which would identify dust and provide a better estimate of the $PM_{2.5}$ concentrations when dust is present.

Data availability. All data used in this analysis are publicly available. Most regulatory data were obtained from the EPA Air data site (https://www.epa.gov/outdoor-air-quality-data) and the AirnowTech site (https://www.airnowtech.org/. Data for the Keeler, CA site were from the Great Basin Unified Air Pollution Control District (https://www.gbuapcd.org/). Data for the Cully Helensview School in Portland, OR was downloaded from the Oregon Department of Environmental Quality website (https://www.oregon.gov/deq/aq). PurpleAir data were from the PurpleAir site (http://map.purpleair.com)

Author contributions. DJ designed the study, developed the analysis protocols and wrote the manuscript. CM, KT and NM conducted data analysis. B.F, J.O. and E.A reviewed the manuscript and provided comments on the analysis.

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Competing interests. The authors have no competing interests to declare.

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Tables/Figures

Table 1. Methodology for identification of pollution events for 50 cases in Part I.

Event	Method of Identification	$PM_{2.5}/PM_{10}$ (if available)
Typical urban	One hour regulatory $PM_{2.5}$ measurements exceeded 47 µg m^{-3} during non-wildfire season with no known presence of smoke or dust.	>0.5
Smoke	One hour regulatory $PM_{2.5}$ measurements exceeded 47 µg m^{-3} in the presence of smoke as indicated on the NOAA Hazard Mapping System-Fire and Smoke Product.	>0.5
Dust	One hour regulatory $PM_{2.5}$ measurements exceeded 47 µg m^{-3} during a known dust event.	<0.5

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415 **Table 2. Peak regulatory $PM_{2.5}$, slope and R^2 results from analysis of regulatory and PAS data, raw and with standard correction, for 50 individual pollution events. (N gives number of events of each type, SD is standard deviation. p-values compare the means of each group. p- values less than 0.05 indicate significant differences.**

	Statistics	Peak reg $PM_{2.5}$ ($\mu g\ m^{-3}$)	Slope of Reg $PM_{2.5}$ versus PAS $PM_{2.5}$ raw data	Slope of Reg. $PM_{2.5}$ versus PAS $PM_{2.5}$ w/std. corr.	Slope of raw PAS PM_1 versus PM_{10} mass conc	Slope of raw PAS 0.3 µm versus 5 µm counts
Urban	**mean (N=16)**	85.15	0.53	1.00	0.56	732.6
	SD	56.69	0.12	0.23	0.10	440.8
Smoke	**mean (N=28)**	280.32	0.52	0.99	0.45	376.7
	SD	226.28	0.09	0.18	0.14	268.8
Dust	**mean (N=6)**	59.76	3.16	5.54	0.29	132.6
	SD	7.91	0.50	1.13	0.08	76.6
Urban versus Smoke	**p value**	0.00	0.60	0.28	0.01	0.00
Urban versus Dust	**p value**	0.24	0.00	0.00	0.00	0.00
Smoke versus Dust	**p value**	0.01	0.00	0.00	0.02	0.06

420 **Table 3. Relationship between hourly regulatory $PM_{2.5}$ and PAS $PM_{2.5}$ with standard correction. Data are included for all simultaneous measurements for the 50 identified events. (N gives number of hours of data of each type).**

	Mean Reg $PM_{2.5}$ ($\mu g\ m^{-3}$)	Mean PAS $PM_{2.5}$ w/std corr ($\mu g\ m^{-3}$)	Slope for Reg versus PAS w/std corr (R^2)	Intercept ($\mu g\ m^{-3}$)	RMSE ($\mu g\ m^{-3}$)	Mean bias ($\mu g\ m^{-3}$)
Urban (N=966)	33.9	28.7	1.02 (0.793)	4.60	10.9	5.2
Smoke (N=6536)	66.4	66	1.08 (0.866)	-4.68	36.0	0.4
Dust (N=240)	30.5	6.3	4.98 (0.661)	-1.09	27.9	24.2

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Table 4. Mean regulatory (reg) $PM_{2.5}$, PAS $PM_{2.5}$ (with standard (std) and new corrections), ratio of PAS PM_1/PM_{10} concentration and ratio of PAS 0.3 to 5µm counts by CAF bins. The CAF bins are centered on the indicated value.

CAF bin midpoint	N (hrs)	Reg $PM_{2.5}$ (µg m^{-3})	PAS $PM_{2.5}$ w/std correction (µg m^{-3})	PAS $PM_{2.5}$ w/new correction (µg m^{-3})	Mean ratio of PAS PM_1/PM_{10}	Mean ratio of PAS 0.3 to 5µm counts
0.05	260	89.5	91.4	91.4	0.55	730
0.15	334	59.4	61.5	61.5	0.55	697
0.25	231	41.6	43.4	43.4	0.56	723
0.35	131	37.6	38.5	38.5	0.54	623
0.45	57	36.9	37.3	37.3	0.54	611
0.55	14	40.6	25.1	33.0	0.44	474
0.65	30	52.5	16.0	45.7	0.33	249
0.75	104	68.4	13.5	63.8	0.25	151
0.85	86	59.3	11.2	60.7	0.20	105
0.95	10	57.2	12.4	66.1	0.21	111

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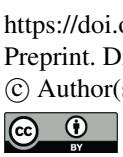

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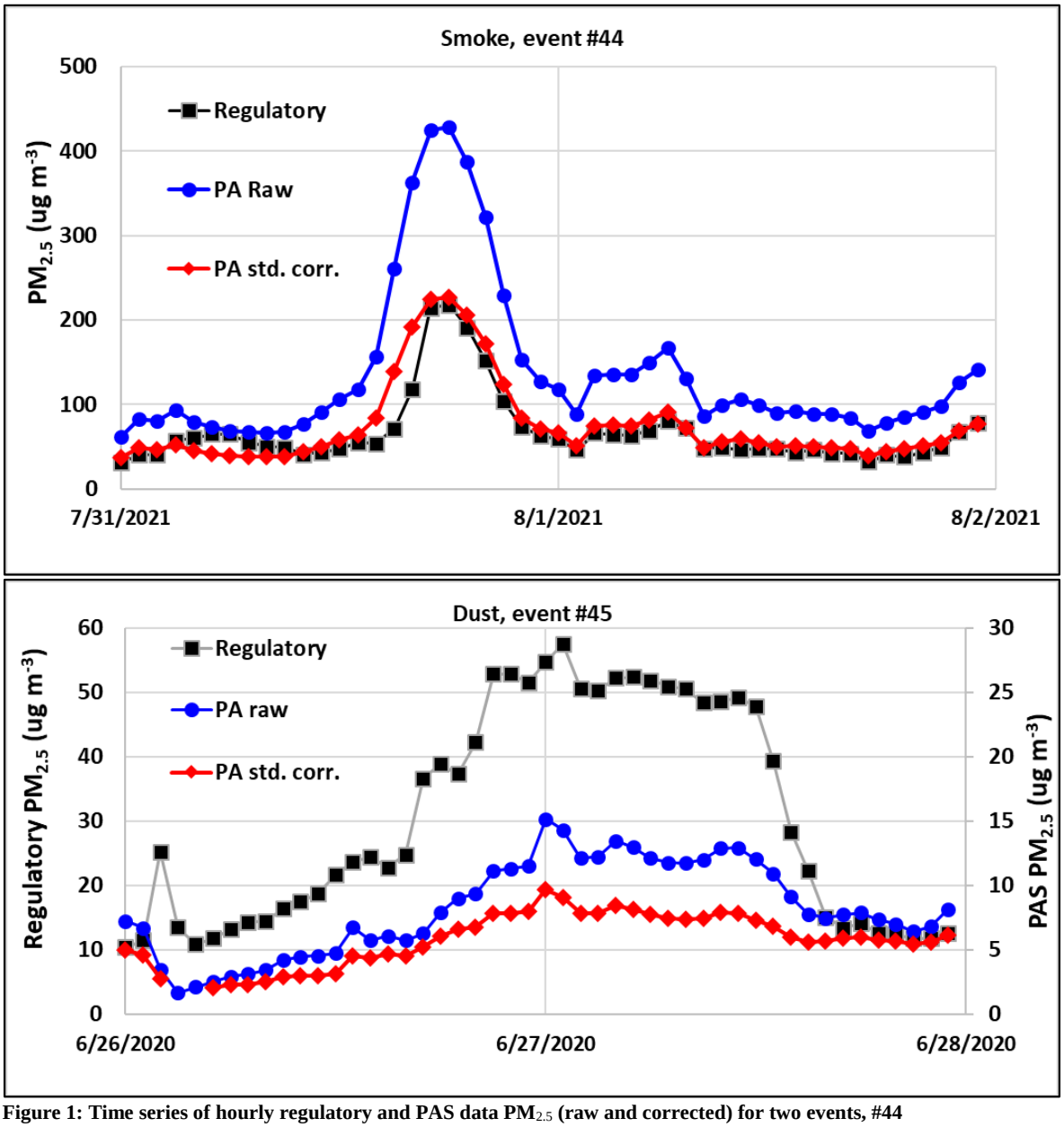

440 **Figure 1: Time series of hourly regulatory and PAS data $PM_{2.5}$ (raw and corrected) for two events, #44 (smoke, top) and #45 (dust, bottom). Time is in UTC. Note that for the dust event, the PAS data are plotted using the right axis values.**

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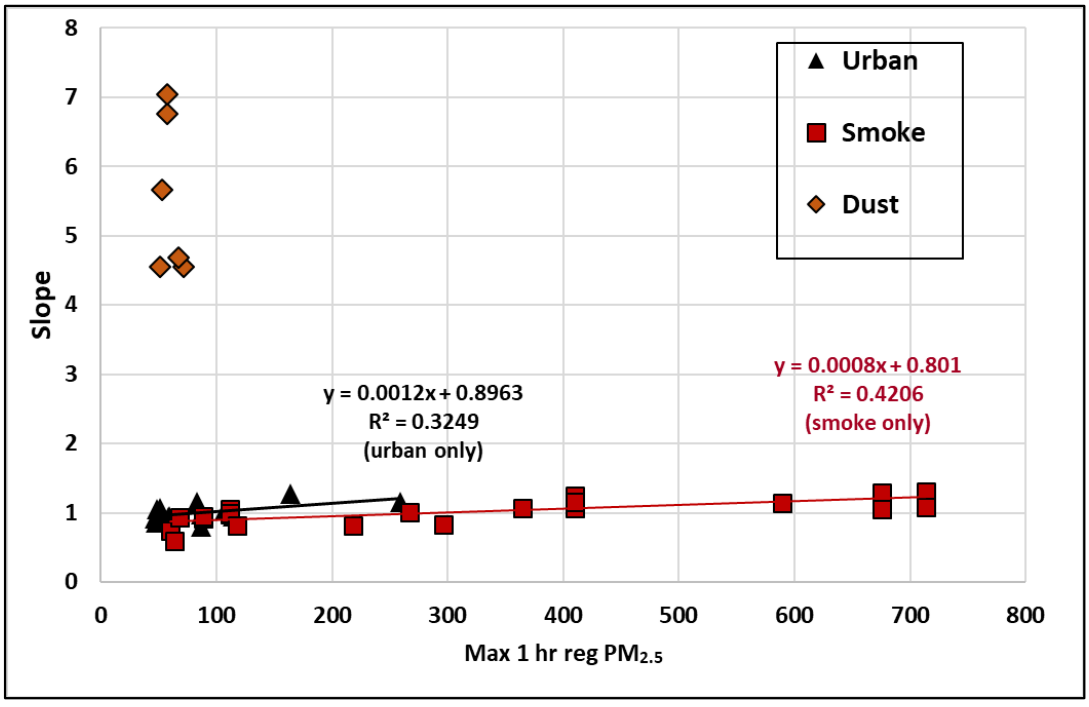

Figure 2: Slope of regulatory $PM_{2.5}$ versus PAS $PM_{2.5}$ with standard correction for 50 individual pollution events versus the maximum 1-hour regulatory $PM_{2.5}$. Slopes are given per Equation 2 and are unitless. The regression line for urban and smoke events shows a significant correlation ($p<0.05$ and $p<0.01$, respectively).

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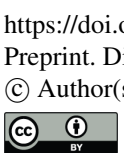

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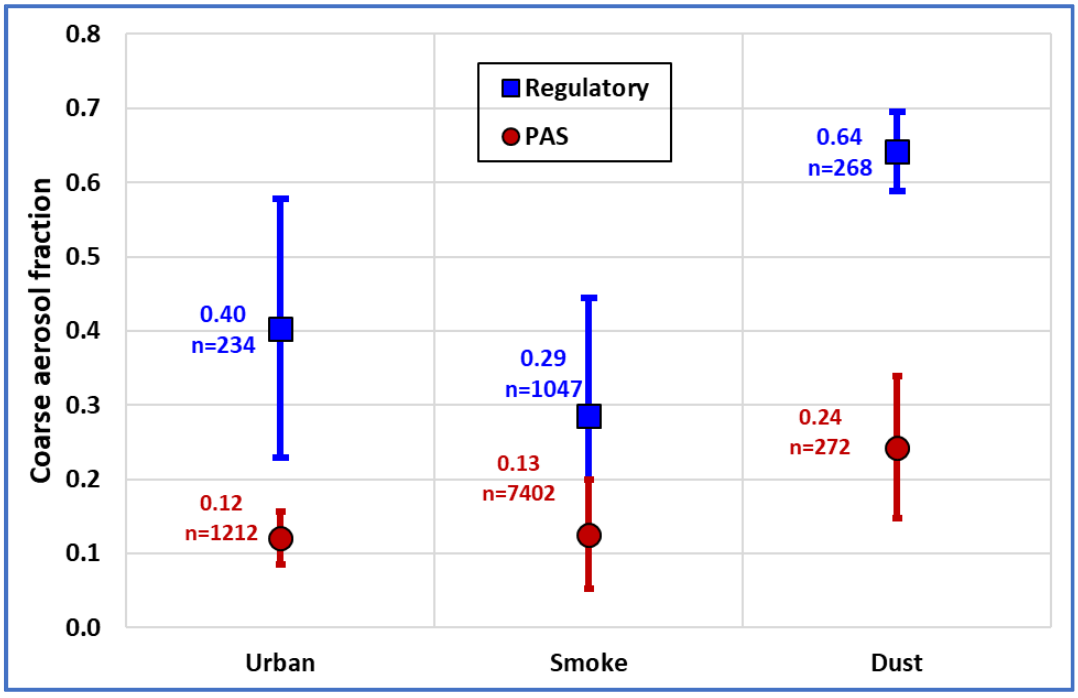

Figure 3: Mean coarse aerosol fraction (Equation 3) calculated using the regulatory data and the PAS raw data. The values near each point give the mean and number of data points (hours) in each bin.

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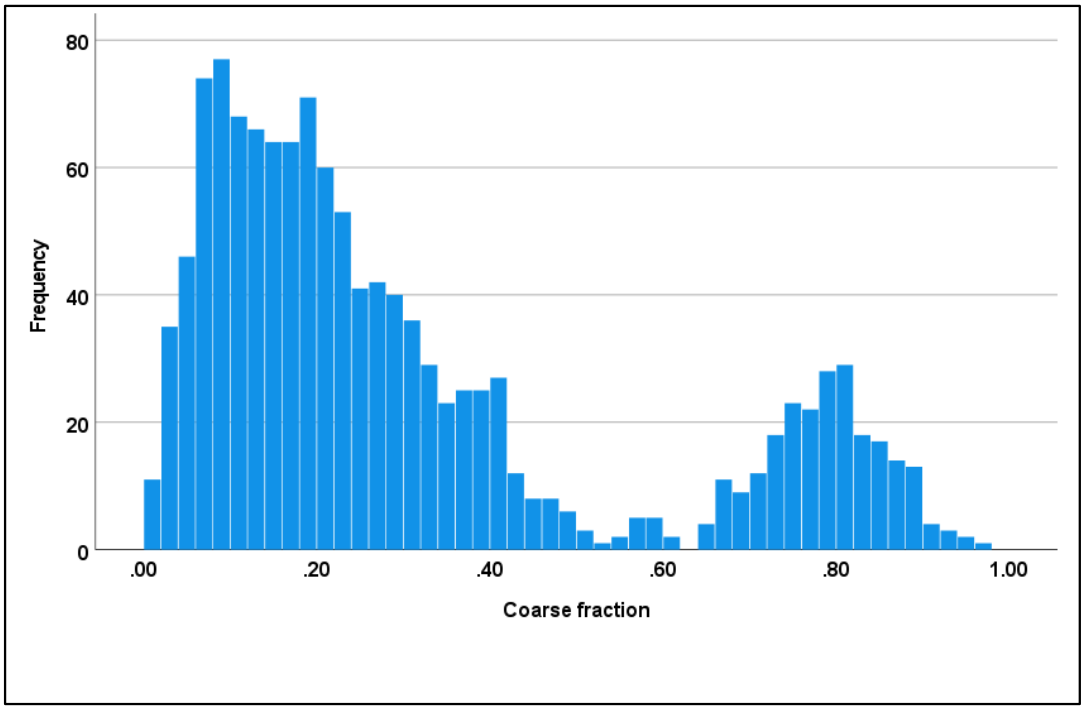

Figure 4. Histogram of coarse aerosol fraction (CAF) at Keeler, CA using regulatory $PM_{2.5}$ and PM_{10}data for hours with $PM_{2.5}$ >25 µg m^{-3}.

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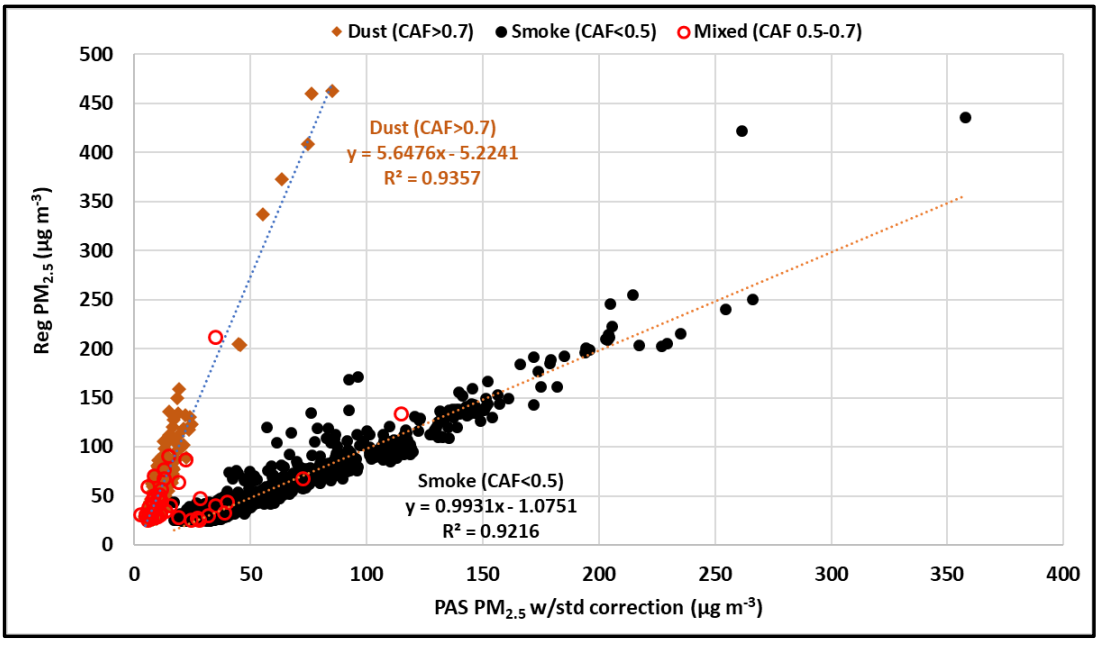

Figure 5. Regulatory $PM_{2.5}$ versus PAS with standard correction at Keeler, CA for hours with regulatory $PM_{2.5}$ >25 µg m^{-3}. The data are separated by the CAF, as measured by the regulatory data.

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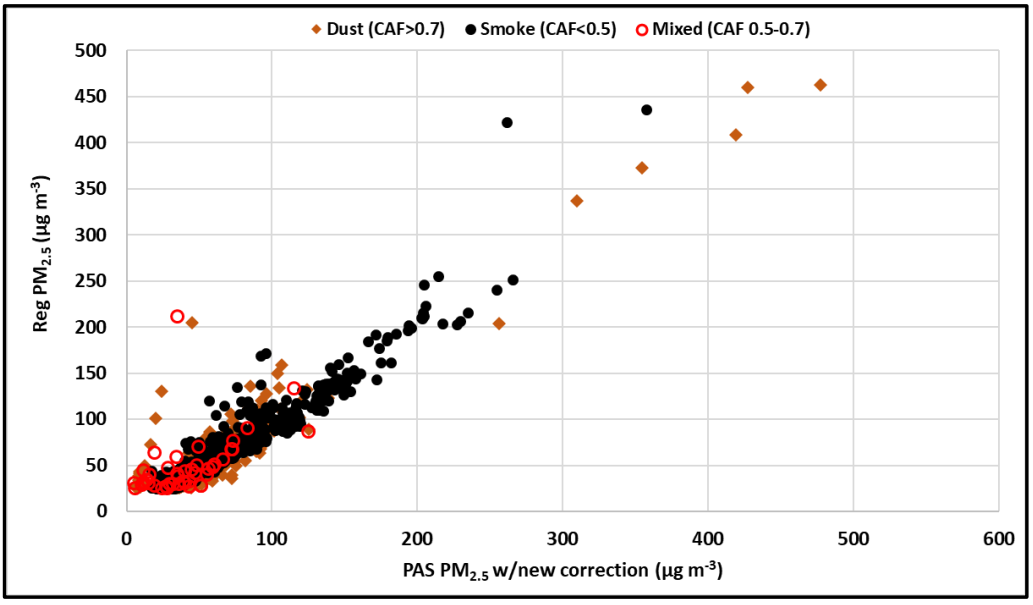

Figure 6. Regulatory $PM_{2.5}$ versus PAS with new correction at Keeler, CA for hours with regulatory $PM_{2.5}$ >25 µg m^{-3}. The data are separated by the CAF, as measured by the regulatory data.

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