



Residual impacts of a wildland urban interface fire on urban particulate matter and dust: a study from the Marshall Fire

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Abstract

The impacts of wildfires along the wildland urban interface (WUI) on atmospheric particulate concentrations and composition are an understudied source of air pollution exposure. To assess the residual impacts of the 2021 Marshall Fire (Colorado), a wildfire that predominantly burned homes and other human-made materials, on homes within the fire perimeter that escaped the fire, we performed a combination of fine particulate matter (PM_{2.5}) filter sampling and chemical analysis, indoor dust collection and chemical analysis, community scale PurpleAir PM_{2.5} analysis, and indoor particle number concentration measurements. Following the fire, the chemical speciation of dust collected in smoke-affected homes in the burned zone showed elevated concentrations of the biomass burning marker levoglucosan (median_{levo} = 4147 ng g⁻¹), EPA priority toxic polycyclic aromatic hydrocarbons (median Σ₁₆PAH = 1859.3 ng g⁻¹), and metals (median Σ₂₀Metals = 34.6 mg g⁻¹) when compared to samples collected in homes outside of the burn zone 6 months after the fire. As indoor dust particles are often resuspended and can become airborne, the enhanced concentration of hazardous metals and organics within dust samples may pose a threat to human health. Indoor airborne particulate organic carbon (median = 1.91 μg m⁻³), particulate elemental carbon (median = .02 μg m⁻³), and quantified semi-volatile organic species in PM_{2.5} were found in concentrations comparable to ambient air in urban areas across the USA. Particle number and size distribution analysis at a heavily instrumented supersite home located immediately next to the burned area showed indoor particulates in low concentrations (below 10 μg m⁻³) across various sizes of PM (12 nm–20 μm), but were elevated by resuspension from human activity, including cleaning.

Keywords Wildland urban interface · Wildfire · Indoor dust · Particulate matter · Organic speciation · Polycyclic aromatic hydrocarbons · Metals and metalloids

Introduction

On December 30th, 2021, the Marshall Fire, fueled by 100 mile per hour wind gusts and a dry landscape, swept through the Colorado Front Range towns of Louisville and Superior, destroying over 1000 homes, damaging property, and blanketing the region in a cloud of smoke. The Marshall Fire boundary extended into suburban and urban neighborhoods along the wildland urban interface (WUI) (Goodrick et al. 2013), and the resulting combustion ignited not only vegetative fuel but also homes, appliances, and vehicles. This combustion of the built environment may produce a different suite of pollutant emissions than conventional wildfires. After the fire, remaining indoor and outdoor environments within the burn zone were covered in ash residue.

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The WUI is commonly defined as the region where urban development borders public and private wildlands (Davis 1990). Similarly, the Wildland Urban Intermix is defined as the area where urban development is scattered throughout the wildland area (Davis 1990). Within the USA, the WUI is expected to reach almost one million km², with the majority of new WUI development occurring in the mountain west region (Theobald and Romme 2007). As the WUI continues to expand, WUI fires that affect urban development, such as the Marshall Fire, are expected to become more commonplace (Radeloff et al. 2005). While several studies have focused on modeling the susceptibility of communities within the WUI to wildfires (Schweizer et al. 2017), calculating the bulk concentration of PM stemming from wildfires (Ager et al. 2019; Prichard et al. 2020), and determination of the emission factors of common fuels found in WUI regions across the USA (Holder et al. 2020), few have focused on the chemical characterization of WUI fire PM and ash emissions after a fire. Fuel type and environmental conditions during combustion influence wildfire smoke and ash composition, resulting in a different combination of chemical species in WUI fire smoke compared to other forms of wildfire (grassland, forest, etc.). However, particulate emissions from WUI fires are poorly understood, as are their impacts on local air and soil.

Fires emit a variety of pollutants, driven by the fuel that is burned and the conditions of the fire. PM with a diameter equal to or less than 2.5 µm (PM_{2.5}) is emitted in large amounts and is important in part due to its wide array of adverse health effects and climate implications. Wildfire emissions of PM_{2.5} often contain greater concentrations of toxic compounds than ambient PM samples collected in the same location (Aguilera et al. 2021; Wegesser et al. 2009). Elevated exposure to wildfire PM_{2.5} has been associated with respiratory morbidity, aggravating conditions such as asthma and chronic obstructive pulmonary disease, as well as an increase in overall mortality (Matz et al. 2020) and some evidence of impacts to the cardiovascular (Evans et al. 2022) and reproductive systems (Wettstein et al. 2018). However, most studies of fire emissions and air quality impacts have focused exclusively on vegetative fires, whereas the Marshall Fire ignited a combination of the built and natural environment. Combustion of anthropogenic materials likely creates PM that is more toxic, as shown in preliminary studies of the California Camp fire of 2018 (Boaggio et al. 2022).

Toxic particulates are transported to ash via a combination of heating, combustion, and deposition (Schlosser et al. 2017). Following wildfires, heavy winds and dry deposition mix soils and ash (henceforth referred to as dust), which contain high concentrations of toxic and carcinogenic compounds, such as heavy metals and polycyclic aromatic hydrocarbons (PAHs) (Balfour et al. 2014; Pereira and Ubeda 2010; Caumo et al. 2022). Dust is often transported

via wind to indoor environments, where it is deposited on household surfaces (Fig. S1), potentially exposing homeowners to harmful compounds, especially during household activities like cleaning that result in PM resuspension.

Virtually all WUI fire studies to date have focused on the immediate impact of wildfires on air pollution, assessing particulate composition and concentrations during fires. However, as residents return to their unburnt homes in the days and weeks after a blaze, they are often faced with uncertainty as to how polluted their homes remain. In this work, we study the impacts of the Marshall Fire in the two months following the event, characterizing the solid phase particulate matter in homes prior to re-occupation post-fire. Understanding the pollutant profile of indoor dust and airborne PM following the blaze is essential to inform homeowners impacted by the Marshall Fire of the pollutant composition in their homes and advise remediation efforts following future WUI fires.

Materials and methods

The Marshall Fire

The Marshall Fire began late morning in east Boulder on December 30th, 2021, and spread via very strong easterly winds to the towns of Superior and Louisville. It was extinguished by heavy snowfall during the late evening of December 31st, 2021. From the 30th–31st, the Marshall Fire burned over 6200 acres, 1300 vehicles, and 1000 homes. Following the fire, we identified eight homes in and near the burned area at which to sample (Fig. 1).

Pollutant sampling

To evaluate the composition of indoor dust samples following the fire, we analyzed the metal and organic components of dust collected in homes within and around the burn zone. To assess ambient PM_{2.5} following the Marshall Fire, we collected airborne indoor PM_{2.5} samples, analyzed the size and number distributions of indoor PM, and examined outdoor concentrations of PM_{2.5} across the burn zone using PurpleAir sensors that were already placed in the community.

Sampling took place in eight single family homes in the Front Range of Colorado. One site (Fig. 1, site S) was selected as a supersite: a single family 2878 ft² home constructed in 2020, within the wildfire burn area, having an air exchange rate (AER) of 0.13 air changes per hour (ACH). The owners evacuated prior to the fire and left the home unoccupied throughout the study. However, on February 7th and 8th, a six-person cleaning crew entered the supersite house and proceeded to clean using a cylindrical HEPA vacuum backpack, shop vac, spray cleaners, scrub mops and

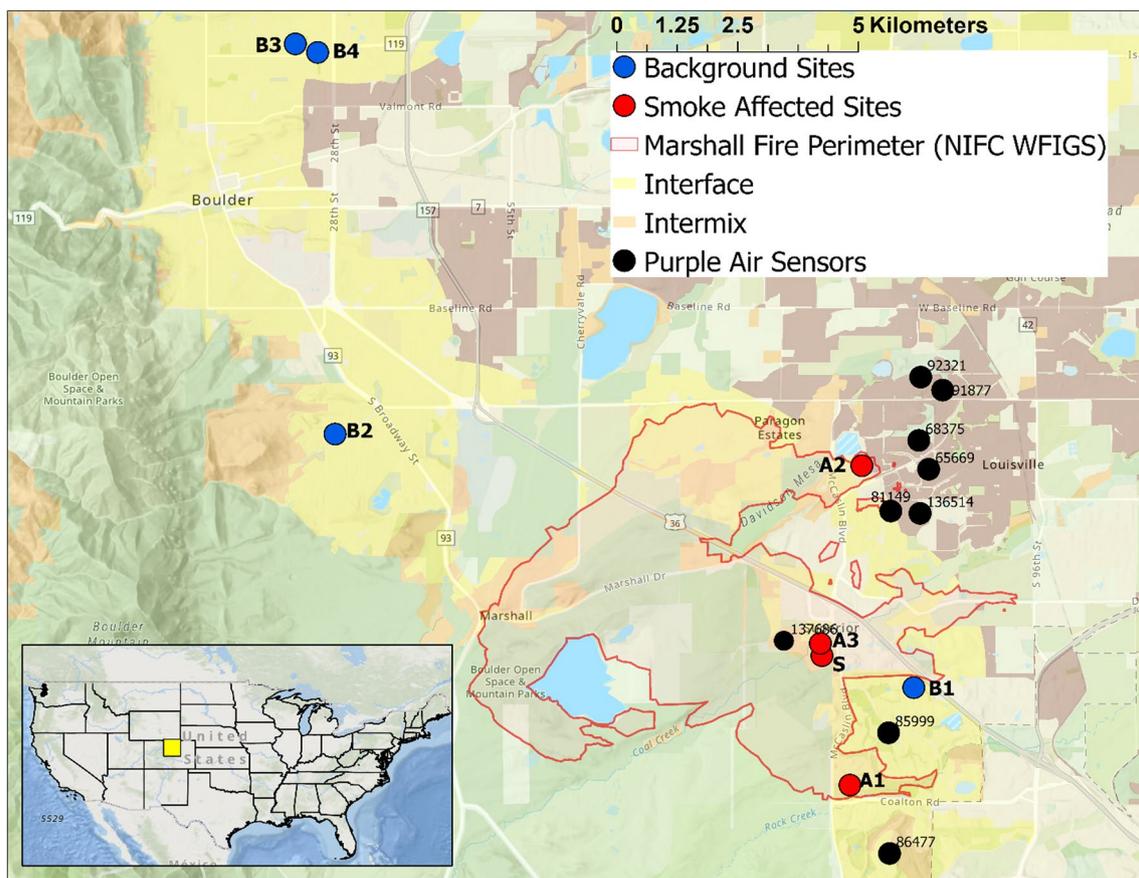


Fig. 1 Marshall Fire boundaries and sampling site locations. Red sites designated **A** represent smoke-affected sampling locations and blue sites designated **B** represent background sampling locations. **S**

refers to the supersite location. Black sites and IDs represent PurpleAir PM sensors used to create an interpolated PM surface after the fire

brushes, soot sponges, and rags. On February 7th, cleaning was predominantly conducted upstairs, and downstairs on the 8th. Windows were opened throughout the cleaning period. Further samples were collected in three uninhabited homes within the burn area, and four outside of the fire boundary. AERs for sample sites A1 and A2 were 0.24 and 0.16 ACH, respectively.

Laboratory analyses

Indoor dust organic analysis

Eight days after the Marshall Fire (January 8th), we collected indoor dust samples in smoke-affected homes impacted by the blaze prior to any cleaning or remediation efforts undertaken by homeowners (Fig. 1, A1–A3 and S). Dust samples appeared to be some combination of wildfire ash and soil deposited by the high winds during the fire. Samples were collected on both the first and second floors of the supersite home (S Upstairs and S Downstairs). An additional dust sample (A3) was collected at a home neighboring

the supersite home. Background samples (B1–B4) were taken in homes outside of the wildfire burn area, upwind and downwind of the burn, approximately 6 months after the fire. Samples were collected by running a sterile metallic scraper over windowsills and porches and transferring dust to a sterile petri dish as outlined in Martikainen et al. (2021). Petri dishes were then labeled and sealed with PTFE tape to avoid potential contamination during transport. Samples were stored in a $-25\text{ }^{\circ}\text{C}$ freezer prior to analysis.

Samples were extracted for organics analysis using ultrasonic agitation as outlined by Agarwal and Ray using an Agilent Gas Chromatograph Mass Spectrometer (GCMS, 6090N GC, 5975 MS) (Agarwal et al. 2006; Ray et al. 2008). Approximately 0.1 g of dust for each sample was dried at room temperature. Prior to extraction, samples were spiked with 25 μL of internal standard. Internal standard spiked dust samples were twice washed in 10 ml of dichloromethane solvent for 15 min with ultrasonic agitation in a water bath. Reference internal standard pairings are displayed in Table S1. The extracted samples were then pressed through a glass fiber filter (Whatman GF/C) and concentrated to 2 ml

under a stream of ultra-high purity nitrogen. Uncertainties were solely a function of the calibration curve uncertainty (Table S7).

Indoor dust metals analysis

Metals concentrations were determined following the methodology outlined by Lough et al. (2005) using an ICP-OES (Thermo Fisher Scientific iCAP 7000 Duo) at CSIR-Institute of Minerals and Materials Technology. Each sample, 50 mg, was digested in trace metal grade HCl:HNO₃ at a 1:3 (volume/volume) ratio and 1 ml of deionized water at 150 °C. Samples were cooled overnight and filtered through Whatman size 542 filter paper. Approximately 7 ml of sample was then injected into the ICP-OES. Samples of HCl:HNO₃ at a 1:3 ratio were blank subtracted from the metals concentrations. Uncertainties were calculated as a function of the standard deviation of replicate analysis (Table S8).

Quartz fiber filter elemental carbon (EC) and organic carbon (OC) analysis

Quartz fiber filter (QFF) sampling began on January 8th in three homes (A1, A2, and S). QFF sampling ran from January 8th to January 28th for location A1, January 8th to February 7th for location A2, and January 8th to February 4th for location S. Filter sampling was conducted prior to significant cleaning or remediation at each sampling site. Indoor PM_{2.5} was monitored using an emission pod (EPOD, Hannigan Lab, Boulder, CO). The inlets for each of the PM sampling devices were placed 1 m above the floor. PM_{2.5} was collected on 25 mm diameter QFFs. Prior to collection, QFFs were conditioned at 500 °C for 12 h to remove potential contamination and were then stored in sterile aluminum foil wrappers in a freezer cooled to -25 °C. Before transporting QFFs to field sites, they were transferred to oven-treated amber glass jars, separated by sterile aluminum disks. Additional transportation and storage procedures for QFFs are described by Pfothenauer et al. (2019) and Piedrahita et al. (2019).

Particulate EC and OC concentrations were measured from the collected QFFs in the laboratory with a Sunset Laboratory OC/EC analyzer (Lab OC-EC Aerosol Analyzer). Following the NIOSH 5040 protocol using the thermal-optical-transmittance method, EC and OC constituents, as well as total EC and OC concentrations, were determined for each of the QFFs. Additional details on the chemical properties of EC and OC constituents are provided in Table S5. Blanks were run at the start of each new day of OC/EC operation to ensure consistent calibration across QFF samples. Uncertainties were calculated using the root sum of squares method (Table S9).

Quartz fiber filter organic speciation

Organic compound measurements were made using an Agilent Gas Chromatograph Mass Spectrometer (6090N GC,5975 MS) (Hannigan et al. 1998; Fraser et al. 1996). Following methods outlined in Dutton et al. (2009) and Xie et al. (2012), organic chemical analysis for QFFs was run via a combination of solvent extraction and GCMS. QFFs were spiked with the same 25 µL internal standard and were analyzed using the same GCMS method as dust samples. Uncertainties were calculated using the root sum of squares method (Table S10).

Low-cost outdoor commercial sensor PM_{2.5} analysis

To understand outdoor air pollution after the Marshall Fire, PM_{2.5} data from publicly available PurpleAir (PA) sensors within the study area (defined as within the cities of Superior, Louisville, and unincorporated Boulder County near to the burn perimeter) and during the study period (January 8th–February 4th) were interpolated. Data were removed from PAs designated as indoor sensors or that showed indoor source PM_{2.5} signatures (i.e., from cooking). These indoor source signatures were identified by looking at time series data for each PA sensor and removing sensors that demonstrated consistent short-term spikes in PM_{2.5} concentrations that demonstrated exponential decay, as activities like cooking are known to lead to these signatures (Shrestha et al. 2019) that do not occur in outdoor settings. Data deemed unreliable by PA were also removed. PA data were corrected and unreliable data were removed using the June 2021 version of the U.S. Environmental Protection Agency (EPA) correction equation for PA data during wildfire periods (AirNow Fire and Smoke Map 2022). The corrected and cleaned PA PM_{2.5} data were then spatially interpolated using inverse distance weighting (IDW) interpolation using the square of the distance as the weighting.

Indoor PM concentrations

PM size measurements

Size distributions of indoor aerosols were measured using a Scanning Mobility Particle Sizer (SMPS, Model 3938, TSI Inc.) outfitted with an X-Ray Neutralizer (Model 3087, TSI Inc.) and long DMA (Model 3080, TSI Inc.) and an Aerodynamic Particle Sizer (APS, Model 3221, TSI Inc.) located in the kitchen of the supersite house (downstairs level), approximately 1.3 m above the floor. Electrical mobility diameters measured by the SMPS were assumed to be equivalent to physical diameters (D_p), and aerodynamic diameters (D_a) measured by the APS were converted to physical diameters using an effective density of 1.2 g cm⁻³ (Li et al. 2016).

Results and discussion

Indoor dust organic composition

The concentrations of 65 semi-volatile non-polar and semi-polar compounds were quantified for each collected dust sample. Concentrations of quantitated compounds and their relevant abbreviations at each measurement site are given in Table S2.

Concentrations of biomass burning markers, such as levoglucosan (median_{levo} = 4147 ng g⁻¹) and syringaldehyde (median_{syring} = 298 ng g⁻¹), were significantly greater in smoke-affected homes than background samples (median_{levo} = 256 ng g⁻¹, median_{syring} = 44 ng g⁻¹), indicating that wood burning and biomass combustion may be greater sources of organic contaminants in Marshall Fire samples (Fig. S2).

Following wildfires, toxic compounds such as PAHs are often stored in dust via dry or wet deposition in organic matter. These compounds are emitted both by anthropogenic activities (such as industrial processes and vehicle emissions) and biomass burning (Kim et al. 2011; Kohl et al. 2019). The median Σ_{16} PAH concentration of background dust samples was 1607.5 ng g⁻¹ dry weight, which is greater than background concentrations measured around the world, and comparable to concentrations found in urban soils (Wilcke 2000). The median Σ_{16} PAH concentration in samples from smoke-affected homes was 1859.3 ng g⁻¹, a significant enhancement over both background samples and soils impacted by wildfires (Kim et al. 2011; Vergnoux et al. 2011). Elevated concentrations of Σ_{16} PAHs in samples from smoke-affected homes indicate pollutants from the Marshall Fire may have longer lifespans in indoor dust than in PM_{2.5}.

Σ_{16} PAHs are often categorized as light (composed of two or three rings) or heavy (composed of four, five, or six rings). Heavy Σ_{16} PAHs have greater toxic equivalency factors (TEFs) than two or three rings and include the seven PAHs the EPA considers potentially carcinogenic (Σ_{7C} : BaA, Chr, BbF, BkF, BaP, Ind, DahA) (U.S. EPA Provisional Risk Assessment 1993; Luch 2005; Nisbet and LaGoy 1992). Σ_{7C} concentrations ranged from 79.3 to 831.2 ng g⁻¹ in homes affected by the Marshall Fire (median = 380.1 ng g⁻¹) and 255.4–1459.4 ng g⁻¹ (median = 454.1 ng g⁻¹) in background households, indicating a slight decrease in carcinogenic PAHs per unit mass of dust in samples affected by the Marshall Fire. However, this does account for the dust loading in each sampling site, as the total amount of dust exposure likely differs between affected and unaffected homes. Though Σ_{16} PAH profiles varied dramatically from home to home in our samples, we found that smoke-affected samples contained substantially

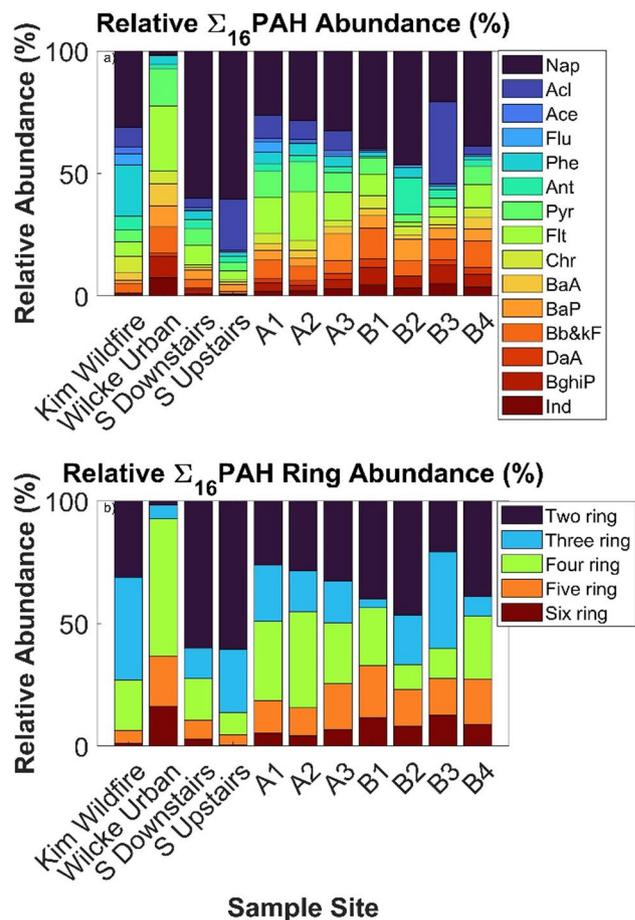


Fig. 2 **a** Relative Σ_{16} PAH abundance by PAH species and **b** PAH ring number in smoke-affected and background dust samples. Σ_{16} PAHs are ordered by ring number from greatest (red) to lowest (blue). Species with the same number of rings are ordered by molar mass. Wildfire Σ_{16} PAH concentrations abundances are from mean ash sample data from Kim et al. (2011) and urban Σ_{16} PAH concentrations are taken from median values found in Wilcke (2000)

greater proportions of lighter PAHs with lower TEFs (Fig. 2). While it is difficult to assess the proportion of PAH deposition due to combustion stemming from the natural and built environment, relative PAH percentages for each sample site were within emission ratios outlined in the literature for various anthropogenic combustion sources (i.e., tires, cars, E-waste, polystyrene, furniture, and PVC, Table S3) and wildfire PAH emissions, suggesting a combination of fire through both built and natural sources contributed to the total PAH deposition in dust samples (Lonnemark and Blomqvist 2006; Lonnemark 2005; Lemieux and Ryan 1993; Valavanidis et al. 2008; Andersson 2005; Andersson et al. 2004).

Indoor dust metals composition

The concentrations of twenty different metal and metalloid species ($\Sigma_{20}\text{Metals}$) were quantified across the nine indoor dust samples (A1–B4, S Upstairs, S Downstairs). These were compared to concentrations of $\Sigma_{20}\text{Metals}$ in soil samples, and median background data for the state of Colorado from Smith (2010; Table S4).

To determine whether combustion during the Marshall Fire enriched metal concentrations in household dust, sample metal concentrations were compared to those of Colorado soil. Wildfire impacted dust samples contained slightly elevated concentrations of heavy metals when compared to background samples (median $\Sigma_{20}\text{Metals}$ = 34.6 mg g⁻¹ and 30.1 mg g⁻¹, respectively). The absolute concentrations of metals in both smoke-affected and background dust samples were significantly lower than previously reported soil concentrations across the state of Colorado (median = 130.28 mg g⁻¹) (Smith 2010). The lower heavy metal concentrations in our samples may be due to the heterogeneous nature of our dust samples, which were not just composed of soil particles but also contained high levels of smoke ash. Figure 3 displays the total and relative composition of quantified metals for each sample site. Smoke-affected samples displayed slightly elevated concentrations of heavy metals such as Zn, As, Co, Cr, and Cu when compared to ambient soils across Colorado. However, the relative metals compositions of both smoke-affected and background dust samples are similar to that of Colorado soils, indicating that for the majority of lighter elements, metal deposition may be due to the presence of soils in the dust rather than contamination by combusted biomass or heavy fuels.

Indoor dust source attribution

To further understand the various sources of dust in our samples, we used principal component analysis (PCA) to examine correlations between $\Sigma_{16}\text{PAH}$ data (Jiao et al. 2017; Golobocanin et al. 2004). For this analysis, two principal components (PCs) were extracted via PCA. PCA loadings displaying correlations of each variable with each PC were determined for BA-30-norhopane (Ba-hop) and syringaldehyde (Syring) alongside $\Sigma_{16}\text{PAHs}$, as Ba-hop and Syring respectively represent specific molecular markers for both petroleum and wood combustion. The factor scores and loadings corresponding to these principal axes are displayed in Fig. 4. Principal Component 1 (PC1) accounts for 49% of the total variance of PAHs in our dust samples and was dominated by high loadings for Ba-hop and heavy PAHs such as BGHIp, Ind, and Bb&kF. Ba-hop and these specific PAHs are typically linked to automotive emissions (Li and Kamens 1993). PC2 is responsible for 28% of the variance of PAHs in our dust samples and corresponds with high loadings for

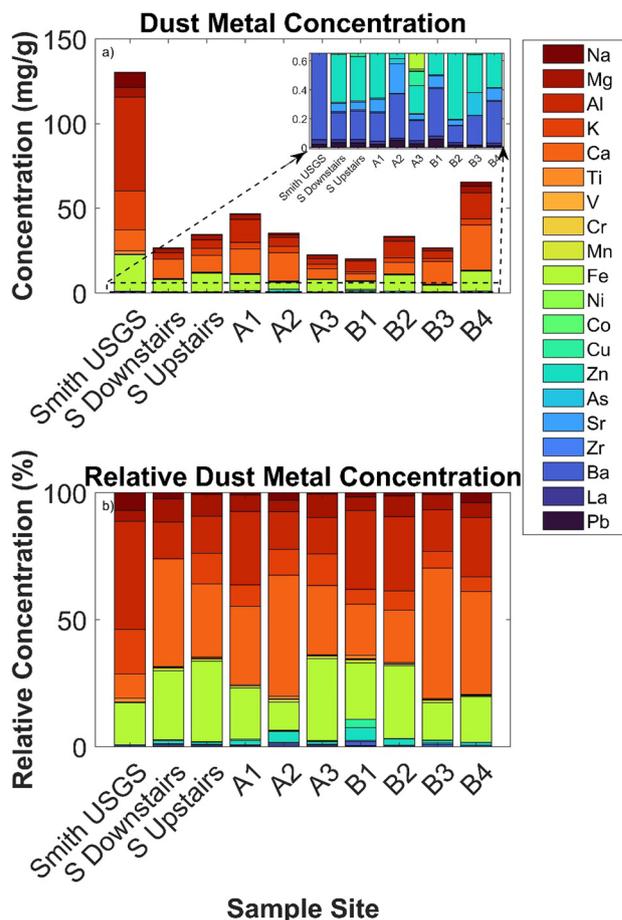
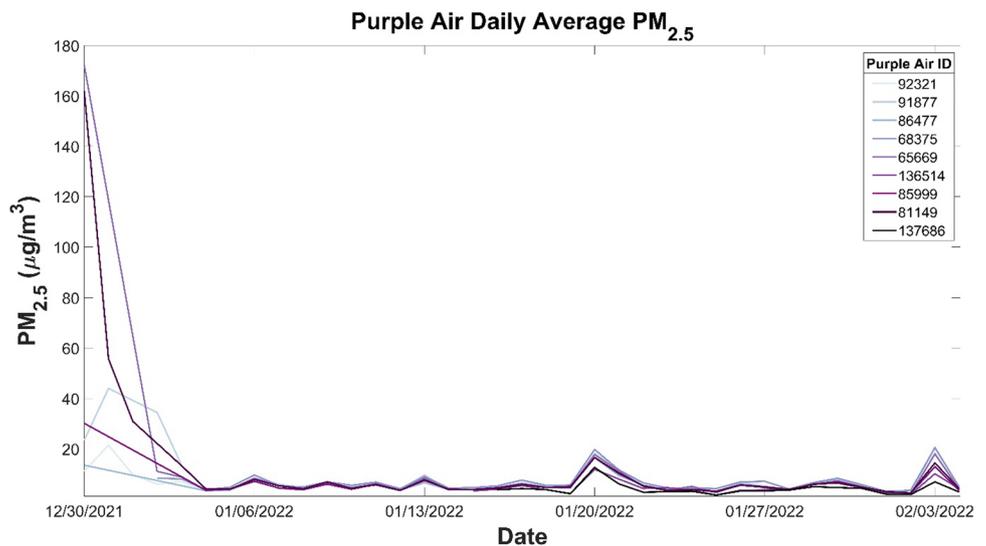


Fig. 3 **a** Absolute and **b** relative concentrations of quantified metals in smoke-affected and background dust samples. Metals are ordered from the greatest molar mass (black) to the lowest (white). Median USGS data for soil in the state of Colorado (Smith 2010) are provided for reference

four-ringed PAHs, as well as Phe, Ace, DbA, and Syring. These species are typically associated with wood, biomass, and coal combustion (Venkataraman et al. 2002; Duval and Friedlander 1981).

Dust samples in Fig. 4a form three distinct groups separated by the two PCs: group one (S up, S down, A3, A1) was separated from group two (A2, B2, B3, B4) by PC2, and was separated from group 3 (B1) via PC1 and PC2. High scores for PC1 were observed at B1, which is located near main roads with high traffic density. PC1 separates group three samples from groups one and two. Additionally, as indicated by Fig. 4, group three is highly associated with five and six ringed PAHs. B1 likely was polluted by vehicle exhaust over a longer period, as lighter, more volatile PAHs either evaporated or degraded over time, leaving behind a greater proportion of heavier, more inert five and six ringed PAHs. The highest scores for PC2 were found at A1 and the supersite samples, which were in the closest proximity to the

Fig. 5 Time series of daily outdoor $PM_{2.5}$ concentrations from PA sensors in the study area from Dec. 30th, 2021, through February 4th, 2022. Sensors are ordered by proximity to the Marshall Fire boundary from farthest (white) to closest (black)

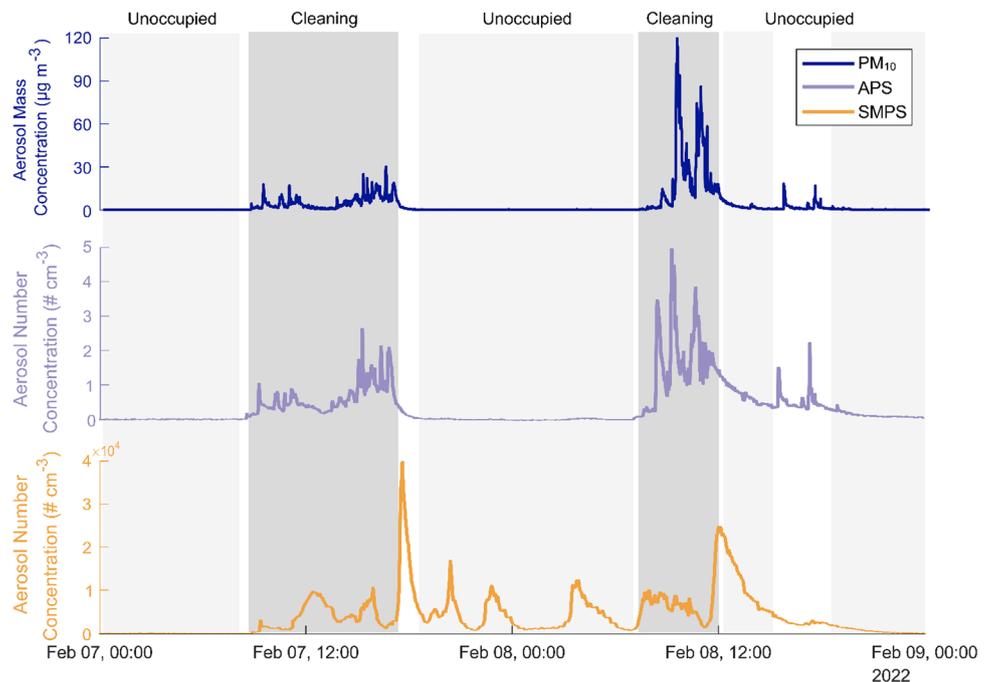


most of the variability in outdoor $PM_{2.5}$ in the area over the study period was temporal (Fig. 5). Elevated daily concentrations of $PM_{2.5}$ observed on January 20th and February 3rd are likely due to changing meteorological conditions. All daily average $PM_{2.5}$ concentrations in the study area were less than $21 \mu\text{g m}^{-3}$, below the daily standard for $PM_{2.5}$ set by the U.S. EPA, suggesting that there was no significant outdoor $PM_{2.5}$ concern during the period of at-home sampling.

Indoor PM analysis

The $PM_{2.5}$ concentrations calculated from the APS and SMPS ($D_a = 0.5\text{--}20 \mu\text{m}$ and $D_p = 12\text{--}500 \text{ nm}$, respectively) throughout the campaign are shown in Fig. S5. Mass concentrations on average are between 0 and $2 \mu\text{g m}^{-3}$ with periodic spikes as high as $8 \mu\text{g m}^{-3}$ that correspond with research and cleaning activity. These concentrations are notably low, suggesting that the majority of the particles from the fire settled rapidly in the house, and remained that way unless disturbed by house activity. The elevation in mass concentration during the cleaning period

Fig. 6 Indoor air number concentration ($\# \text{ cm}^{-3}$) and (estimated) PM_{10} (blue) from APS ($D_a = 0.5\text{--}20 \mu\text{m}$, purple) and SMPS ($D_p = 12\text{--}500 \text{ nm}$, orange) measurements between midnight Feb. 7th and midnight Feb. 9th. Periods when the house was unoccupied, active cleaning, or generally occupied are highlighted in light grey, dark grey, and white, respectively



is even more pronounced in the (estimated) PM_{10} mass loading, as shown in Fig. 6, reaching almost $120 \mu\text{g m}^{-3}$. Also shown in Fig. 6 are the particle number concentrations from the APS and SMPS from the same period. Number concentrations from both instruments were elevated above background levels during cleaning periods (and slightly above house background). The cleaning activity downstairs — closer to the instrumentation — on Feb. 8th resulted in almost double the APS number concentration ($4.9 \text{ particles cm}^{-3}$) than on the 7th ($2.5 \text{ particles cm}^{-3}$) when cleaning was confined to the second floor. However, this is not mirrored in the submicron particles measured by the SMPS. Though the SMPS number concentrations are elevated, they do not reflect the change in cleaning location as clearly as the APS, suggesting different generation and dissipation methods between the coarse mode particles measured by the APS and the sub-500 nm particles measured by the SMPS. Additionally, at the tail end of cleaning activity, just before the cleaning crews leave, SMPS number concentrations spiked sharply. This may be attributed to a direct release of aerosol or secondary aerosol formation from sprays and air fresheners by the cleaning crew. As particulate matter mass and number concentrations were near zero in the period before cleaning, it is important to note that it is difficult to judge the efficacy of cleaning measures employed at the supersite on PM, as concentrations return to baseline following the cleaning. Additionally, during the unoccupied nighttime period between cleaning days, SMPS data show increased concentrations. There were six bursts of new particle formation (NPF) occurring approximately every 22 min during this period, which we attribute to the house heating, ventilation, and air conditioning (HVAC) system (Fig. S6). Comparison of this period to HVAC activity during the house background period on Feb. 5th (Fig. S6) suggests that the NPF may be a result of the evaporation and recondensation of (semi)-volatile compounds from cleaning products that may have deposited in the HVAC system (rather than a house background reservoir).

Carbonaceous analysis of $PM_{2.5}$

Indoor $PM_{2.5}$ concentrations were low in the days and weeks after the Marshall Fire. OC concentrations post Marshall Fire were comparable to background values in the Denver urban area (DUA) reported by Vedal et al. (2009). While our sampling period does not include the $PM_{2.5}$ plume associated with the fire itself, high winds and human activity after the fire likely resulted in the resuspension of $PM_{2.5}$ deposited during the blaze. EC concentrations were greatly diminished (and OC/EC ratios were greatly inflated) when compared to baseline concentrations reported in the Colorado Front Range (Vedal et al. 2009).

OC values at sites A1, A2, and S ranged between 0.5 and $5 \mu\text{g m}^{-3}$ (median = $1.91 \mu\text{g m}^{-3}$) (Table S5). EC concentrations varied between 0.01 and $0.47 \mu\text{g m}^{-3}$ (median = $0.02 \mu\text{g m}^{-3}$). Indoor OC was likely produced by research activity, including particle resuspension during device setup and human foot traffic.

Over the last decade, ambient EC concentrations have drastically decreased across the western USA (Requia et al. 2019). Reduced ambient concentrations of EC coupled with road closures in neighborhoods affected by the Marshall Fire may have resulted in substantially diminished vehicular exhaust emissions near our monitoring devices, which may have contributed to the negligible EC concentrations observed in the sample homes.

During our field campaign (January 8th–February 4th), OC concentrations decreased by an average of $1.47 \mu\text{g m}^{-3}$, and EC concentrations remained constant. OC was predominantly composed of more volatile and less polar constituents, especially immediately after the fire. EC was primarily composed of less volatile constituents. Specifically, more volatile EC constituents are considered markers for gasoline and diesel combustion (Cao et al. 2004). Volatile EC constituents' contributions were negligible, which is consistent with our assumption that low EC concentrations may be driven by a lack of vehicle traffic.

$PM_{2.5}$ samples were analyzed for the same organic species as dust samples (Table S6). Since ambient OC concentrations were low across our measurement period, filters at each location were analyzed as an ensemble to achieve higher organic carbon loadings. The concentrations of the majority of quantitated compounds were equal to or below ambient concentrations measured in the DUA (Dutton et al. 2009). Anthropogenic and biogenic combustion markers, such as retene, pyrene, and methoxyphenols, were lower than previously measured concentrations found in the DUA (Dutton et al. 2009). PAH concentrations, which are typically elevated during wildfires (Ghetu et al. 2022), were much lower at sites A1, A2, and S than monitoring sites across the USA, indicating the impacts of combustion during the Marshall Fire on indoor $PM_{2.5}$ were likely confined to the period during and immediately after the fire. The sum of the 16 EPA priority PAHs ($\Sigma_{16}\text{PAH}$, denoted with an * in Table S6) ranged from 0.64 to 1.03 ng m^{-3} .

Concentrations of petroleum combustion markers such as ba-30 norhopane, ab-hopane, benzo[g,h,i] perylene, and indeno[1,2,3-cd]pyrene were an order of magnitude less than previously recorded concentrations in the DUA (Dutton et al. 2009). Emissions associated with petroleum combustion were likely reduced after the fire due to restrictions on the flow of traffic into and out of affected neighborhoods. Additional analysis probing the sources of petroleum markers in $PM_{2.5}$ is included in Fig. S8.

Conclusions

Following the Marshall Fire, PM concentrations across the burn zone remained relatively low for the month after the blaze. The majority of quantified organic compounds in PM_{2.5}, including toxic and carcinogenic species commonly found after combustion events, occurred at levels comparable to background. Concentrations of pollutants associated with specific urban sources, such as vehicle emissions, were greatly reduced when compared to background concentrations. Similarly, metals concentrations in indoor dust samples were significantly lower than those found in soils across Colorado on a per mass basis, and most deposited metals were likely geogenic in origin. However, dust samples after the Marshall Fire contained elevated concentrations of toxic PAHs. Accordingly, upon returning home, residents should prioritize the removal of indoor dust caused by the Marshall Fire. PAHs were likely deposited to dust via a combination of natural and anthropogenic processes that include both the Marshall Fire and common sources of urban pollution (such as vehicle emissions). The sources of indoor pollutants resulting from WUI fires likely differ from those generated during conventional wildfires due to additional combustion of the built environment. In the future, source apportionment models will need to be developed to discern how the Marshall Fire quantitatively impacted pollutant concentrations in dust and soil.

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Author contribution All authors contributed to this study. Sample collection, data acquisition, and analysis were performed by all authors. Supervision and funding acquisition were conducted by CR, CW, MV, JDG, and MH. The manuscript was written by JS, LM, and ER. All authors have read and approved the final manuscript.

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Data availability Datasets generated and analyzed for this study are provided in the supplemental information. Files include images of typical post-fire indoor dust and ash infiltration, ratio-ratio diagnostic plots for PM_{2.5} samples, estimated supersite PM_{2.5} concentrations, SMPS aerosol concentrations, mass, and size distribution heatmaps from supersite HVAC cycles, median dust metal enrichment factors for smoke-affected and background homes, median dust sterol and methoxyphenol concentrations for smoke-affected and background homes, GCMS and ECOC filter concentrations, GCMS dust organics concentrations, ICP-OES dust metals concentrations, and PurpleAir sensor locations.

Declarations

Ethics approval and consent to participate Not applicable.

Consent for publication Not applicable.

Competing interests The authors declare no competing interests.

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