Post-Wildfire Indoor Pollution in WUI Areas Following the 2025 Los Angeles Fires — Part I: Establishing Baseline Contaminant Levels Prior to Home Reoccupation

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Abstract

Wildland-Urban Interface (WUI) fires pose unique environmental pollution challenges due to the combustion of both natural vegetation and synthetic building materials. Following the 2025 Palisades and Eaton wildfires in Los Angeles, we conducted a field study to characterize indoor air quality and surface contamination in 19 homes before re-occupancy. Indoor $PM_{2.5}$ and PM_{10} concentrations averaged 3.45 and 31.66 µg/m³, respectively, with several homes showing indoor-to-outdoor (I/O) ratios >1 (particularly for PM_{10}) indicating persistent indoor particle reservoirs. Regression analysis suggested that proximity to the fire, absence of air purifiers, use of non-HEPA vacuums, and open windows during the fire significantly increased indoor PM levels, explaining 73% (PM_{10}) and 86% ($PM_{2.5}$) of the variation across homes. Airborne metal concentrations were below health-based thresholds; however, surface wipe samples revealed widespread contamination, with potassium, magnesium, aluminum, and iron frequently exceeding 1,000 µg/ft², and detectable levels of zinc, copper, and manganese in many homes. Lead concentrations exceeded EPA's dust clearance levels in multiple homes, especially on window sills and entry floors. Our findings highlight that while airborne risks may subside within weeks after the fire, indoor surfaces can retain fire-related pollutants, presenting ongoing exposure risks even two months after the fire.

Keywords

Wildland-Urban Interface, Wildfires, Air Toxics, Indoor Air Pollution, the Palisades and Eaton Fires

1. Introduction

As urban areas continue to expand, more people are living in or near wildland regions, known as the Wildland-Urban Interface (WUI). This development has brought communities closer to nature but has also significantly increased their exposure to WUI fire hazards. As a result, the risks of structural damage, economic loss, and adverse health impacts have risen dramatically. Currently, more than 70,000 communities (home to nearly 160 million people) are at risk, with WUI areas expanding by approximately two million acres annually.[1] Historically, 7 of the 15 largest fire-related disasters in the U.S. have been WUI fires.[2] Recent devastating WUI fires include the 2023 Maui Wildfire in Hawaii, the 2021 Marshall Fire in Colorado, the 2016 Gatlinburg Fires in Tennessee, and the 2014 Bastrop County Complex in Texas. Most recently, in January 2025, the Palisades and Eaton fires, fueled by dry and strong Santa Ana winds, caused 30 deaths, forced over 100,000 people to evacuate, and led to the destruction of 6,822 and 9,413 structures, respectively.[3,4] Moreover, air quality significantly deteriorated across the region, with elevated levels of particulate matter (PM) and toxic metal particulates (e.g., Pb), as well as benzene, toluene, ethylbenzene, and xylenes (BTEX), particularly impacting communities in Malibu and Altadena.[5–7] In response, the California Air Resources Board (CARB), the South Coast Air Quality Monitoring District (SCAQMD), and collaborative research initiatives (such as the Post-fire airborne Hazard Observation Environmental Network for Integrated Xposure-monitoring (PHOENIX) project and the LA Fire HEALTH Study) launched comprehensive stationary and mobile air quality measurements in the fire-affected areas to investigate pollutant dynamics and assess long-term public health impacts.[8–11]

Unlike wildland fires that primarily burn vegetation, WUI fires ignite a combination of natural and synthetic materials, including plastics, treated wood, electronics, and household furnishings, which are present in various quantities and densities. The combustion of these materials releases a complex mixture of air pollutants, including volatile organic compounds (VOCs), polycyclic aromatic hydrocarbons (PAHs) and other semi-VOCs, metal compounds, and carbon-containing particulates.[12] These outdoor air pollutants produced during WUI fire events can infiltrate homes through windows, doors, cracks, and HVAC systems, negatively impacting indoor air quality (IAQ).[13,14]

The health risks of WUI fire emissions extend beyond the immediate fire events, as many residents return to their homes within hours or days after evacuation orders are lifted, often unaware of the hazardous pollutants that remain indoors.[15] This is particularly concerning given that individuals spend the majority of their time indoors, especially in their homes, where long-term exposure to these pollutants can significantly increase the risk of cardiovascular diseases, respiratory problems, and neurodegenerative disorders.[16–18] In the absence of readily accessible indoor air monitoring data, residents often rely on sensory cues (such as the distinct odors of smoke, which can range from a metallic smell of burned electronics or rubber to a woody scent from the combustion of organic materials) to judge air quality.[19] These methods are subjective and unreliable, making them inadequate for guiding personal protection and exposure reduction. Thus, there is a need to move beyond subjective assessments of air quality and develop reliable methods to characterize indoor pollutants and evaluate their long-term health impacts following WUI fire events.

While some efforts have been made to monitor ambient air quality affected by WUI fires, IAQ remains relatively understudied. Kirk et al. (2018) conducted IAQ measurements in two homes in the Pacific Northwest during the summer 2015 wildfire season and found that indoor $PM_{2.5}$ concentrations reached an average of 15 µg/m³ during active fire periods.[20] A more recent study in Western Montana monitored 20 homes throughout the 2022 wildfire season and found that indoor $PM_{2.5}$ levels rose substantially during smoke events.[21] Specifically, the mean indoor $PM_{2.5}$ during wildfire periods was 15.9 µg/m³, compared to 5.6 µg/m³ during non-wildfire periods (i.e., near threefold increase). Although protective actions, such as closing windows and using portable air cleaners, can mitigate indoor $PM_{2.5}$ exposure during wildfire events, many homes still exhibited elevated indoor $PM_{2.5}$ levels. In a large-scale California study, Liang et al (2021) reported that indoor $PM_{2.5}$ concentrations during fire events in over 1,400 buildings show nearly tripled values compared to non-fire days.[13] The study also found that newer buildings and the use of air filtration during fires can mitigate indoor exposure by 18% and 73%, respectively.

Some studies have shown that the retention of fire-related pollutants in indoor environments varies significantly by compound class. For example, based on indoor dust collected eight days after the Marshall Fire, Silberstein et al. (2023) reported that PAHs concentrations in smoke-affected homes reached a median of 1859.3 ng/g, representing a significant enhancement over background levels.[22] In contrast, Kohl et al. (2019) found limited persistence of wildfire-derived PAHs in house dust 14 months after the Fort McMurray fire, with concentrations generally lower than those in unaffected urban areas.[23] This contrast highlights that while PAHs can remain embedded in indoor dust shortly after a fire, their long-term persistence appears to decrease over time. Meanwhile, studies by Dresser et al. (2025) and Li et al. (2023) showed that although airborne VOC levels declined rapidly within hours after smoke exposure, VOCs continued to off-gas from indoor materials for weeks, indicating the importance of surface reservoirs.[24,25] In comparison, the elevated $PM_{2.5}$ concentrations in Marshall fire-affected homes were found to decline to near-background levels within weeks, and only a few metals (As, Cr, Cu, Pb, Zn) exhibited moderate enrichment (enrichment factors of 2–5) in indoor dust samples, likely due to pre-existing urban dust and rapid dispersal of fire-emitted particles.[22] Similarly, the trace metal concentrations in house dust 14 months after the Fort McMurray fire were similar to other Canadian cities, with only As showing a modest 62% elevation in neighborhoods where buildings had burned.[23]

To date, most studies on IAQ following WUI fires have measured conditions at isolated time points (such as during the fire, within days or weeks afterward, or over a year later), without capturing before-and-after comparisons necessary to assess how indoor environments evolve during recovery, reoccupation, and renewed human activity. To address this gap, we initiated a multi-phase field study focused on homes impacted by the 2025 Palisades and Eaton fires. This article presents the first publication from the study, aimed at establishing baseline levels of airborne pollutants and surface contamination inside homes before reentry. Subsequent phases of this study will examine air pollutant dynamics throughout the post-fire recovery timeline during re-entry and reconstruction, as well as the long-term persistence of indoor contamination. Collectively, this work will offer foundational data to support evidence-based remediation guidance and inform safe reentry strategies for fire-impacted communities.

2. Methods

2.1. Volunteer Recruitment, Home Selection, and Sampling

We responded after the fires by sending out a questionnaire on February 11th, 2025 to recruit participants from fire-affected areas for our study. The questionnaire included questions such as extent of fire damage, timing of reentry, power availability, cleanup and reconstruction plans, and consent for on-site sampling. The full questionnaire is provided in the Supplementary Information. As of February 20th, 262 residents expressed interest in participating, with 167 reporting noticeable smoke odors indoors and 142 indicating that accumulated ash had not yet been removed. Additionally, many residents reported respiratory issues, eye irritation, and concerns about long-term exposure to residual ashes. Some noted visible soot accumulation on indoor surfaces, while others experienced headaches and worsened allergy symptoms since returning home. Power outages and incomplete remediation efforts have further compounded these challenges, leaving many homeowners uncertain about the safety of their indoor environment.

Based on the responses we received from homeowners, we categorized the houses into three distinct groups according to the fire impact and smoke exposure, as shown in Figure 1. The green category includes homes that met all three criteria: 1) experienced some level of structural damage from the fire, 2) had not been cleaned, and 3) exhibited noticeable indoor smoke odors. The blue category includes homes that met two partial criteria: 1) had not undergone any cleaning 2) either visible fire damage or noticeable indoor smoke odors were present. Lastly, the grey category represents homes lacking sufficient criteria to be classified as green or blue. To capture a diverse range of sampling environmental conditions, we selected a total of 16 homes in Altadena and Pasadena, mainly from the green and blue categories, with a limited number from the less-affected grey category. Additionally, three homes from Malibu, affected by the Palisades fire, were included, resulting in a final sample set of 19 homes. Prior to finalizing our sampling plan, we confirmed with homeowners that no smoking occurs inside these residences, thereby ensuring that our measurements were not influenced by lingering tobacco smoke contamination.

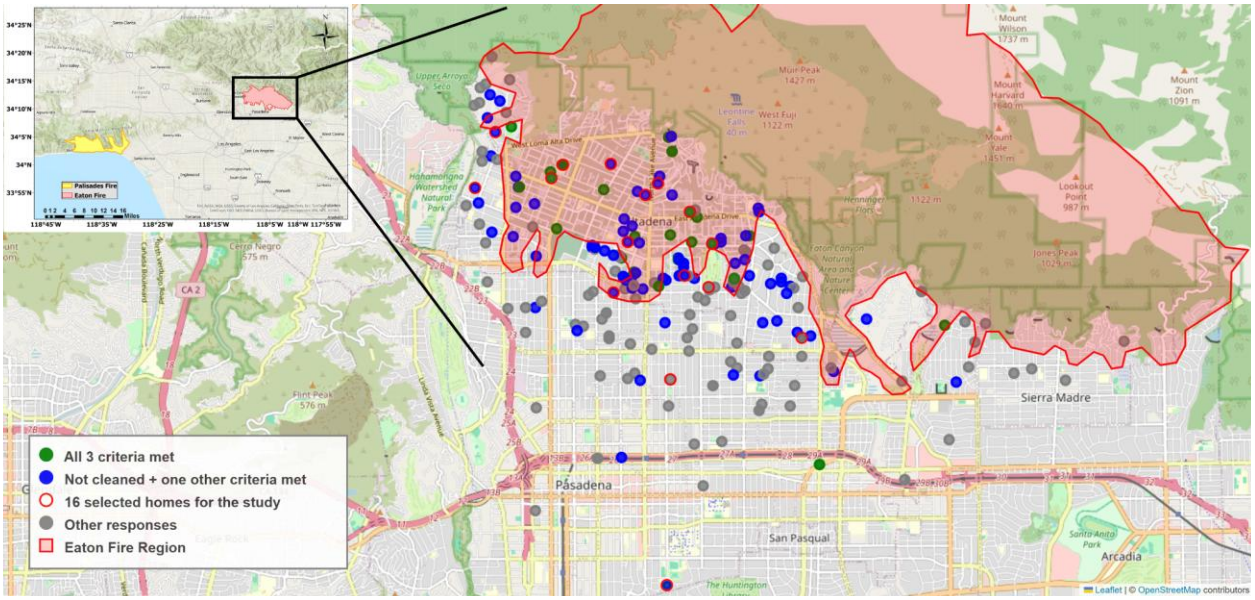

Figure1. Detailed view of the Eaton fire region with the categorization of homes selected for this study. Exact home locations are generalized to protect resident privacy.

Following coordination with the selected homeowners, our monitoring campaign began on March 6th and spanned six consecutive days, including five days of indoor measurements in the Eaton Fire region and a final day in the Palisades Fire region (March 11th). Each day, we visited and sampled three to four homes sequentially, allowing for detailed assessments while maximizing spatial coverage across the affected areas. During a three-hour sampling session at each home, we conducted indoor air quality measurements at a height near the breathing zone (~1.5 m) in one of the commonly used spaces, such as the living or family room. Alongside air quality sampling, we collected surface wipe samples from various contaminated indoor surfaces (See Section 2.3 for more details on the dust sample collection). Measurements were accompanied by contextual information, including ventilation status, use of air purifiers, and surface characteristics (e.g., carpet versus hard flooring). At each home, we also took photographs to document significant ash and soot contamination, structural damage, and surfaces from which we took wipe samples. Additionally, we conducted interviews with homeowners to gather self-reported information such as the building age and structural material, window types, recent cleaning activities, and future remediation plans.

To protect participant privacy, we will use anonymized identifiers (e.g., Homes 1306, 2306, 1307, etc.) when referring to the 19 sampled homes throughout this manuscript. These identifiers were assigned logically to reflect the study region and sampling order without revealing specific addresses. Among these identifiers, the three sampled homes in Malibu begin with the prefix “M”. One residence, Home 2307, was relatively larger compared to other sampling homes, so we sampled two distinct areas separately and assigned them identifiers Home 2307-1 and Home 2307-2 to better capture intra-home spatial heterogeneity. Among all sampled homes, Home 3307 was less affected by the fire due to its distance from the burn zone and had been professionally cleaned prior to our visit; it is used as a benchmark to represent a remediated indoor environment in the study for comparison with more impacted homes.

2.2. Measurement Instruments

The indoor $PM_{2.5}$ and PM_{10} concentrations were measured using the QUANTAQ MODULAIR, a real-time PM monitor with a one-minute temporal resolution and up to 2,000 $\mu g/m^3$ measurement range. The sensor integrates nephelometry, which measures total light scattering from a particle population across multiple angles, and an optical particle counter (OPC), which counts and sizes individual particles as they pass through a laser beam, to determine particle mass concentration.[26,27]

The AETHLAB MA200, with a limit of detection (LOD) of 30 ng/m^3 and a five-minute sampling resolution, was used to measure indoor black carbon (BC) concentrations.[28] This instrument collects BC particles on a PTFE tape and simultaneously passes light at multiple wavelengths (ultraviolet, blue, green, red, and infrared) through the tape. As the BC particles accumulate on the tape, they absorb more light, resulting in the attenuation of transmitted light intensity. The attenuated light is then measured through an optical detector, which is then used to quantify the concentration of carbon-containing particles based on changes in light intensity at each wavelength. In this study, we used the concentrations at the infrared band, as it is found to be the dominant wavelength absorbed by BC.[29,30]

Total VOC (TVOC) concentrations were measured using the HVX501 Handheld VOC Meter[31] (Hal Technology), which features a one-minute sampling resolution, a LOD of 10 ppb, and a measurement range up to 20 ppm. This instrument is equipped with a photoionization detector (PID) sensor that uses an ultraviolet lamp to ionize VOC molecules as they pass through the sensor chamber. The resulting ions are attracted to a charged electrode, generating a small electrical current proportional to the VOC concentration. Since the PID responds collectively to a broad range of VOCs without distinguishing individual compounds, the sensor provides an aggregate measure of TVOCs rather than speciated data. Compared to other low-cost VOC sensors such as metal oxide semiconductor (MOS) or electrochemical sensors, PID-based sensors offer higher sensitivity and better linearity, making them more suitable for accurate real-time VOC monitoring.[32,33]

Toxic metal particulate concentrations were measured using Toxic-metal Aerosol Real-Time Analyzer (TARTA) Version 2.0, developed by San Diego State University and the University of California, Davis.[34–37] In this study, TARTA was configured to sample total suspended particles (TSP) without a size-selective inlet. TARTA employs Spark-Induced Breakdown Spectroscopy (SIBS) to quantify airborne particulate metal concentrations in near real time. In this setup, aerosol particles are collected on the surface of an electrode over a 30-minute sampling period. A high-voltage spark (~5 kV) is then applied to ablate the deposited particles, causing the ionized materials to emit light as they return to their ground states. The emitted light is captured by an optical spectrometer, and metal concentrations are quantified based on the intensity of element-specific emission wavelengths (See Section 2.4 for more details). Compared to TARTA 1.0, this new version features a bigger ground electrode (Ø2.4 mm), no optical lens (less alignment effort needed), lower weights (~ 6 lbs), smaller dimensions (8” x 10” footprint), and better limits of detection (LODs).[38]

All four instruments used in this study for indoor measurements were portable and battery-powered, enabling deployment in post-fire environments where many homes had not yet regained power due to damage to local infrastructure. To facilitate mobility and setup, we mounted the instruments and three power banks on a lightweight, wheeled cart (37.5 × 35 × 62.5 cm) to ensure compactness, easy maneuverability between rooms and areas, and up to nine hours of continuous sampling per day without the need for external power sources (Figure S1a).

The outdoor $PM_{2.5}$ and PM_{10} concentrations were extracted from the PHOENIX network, a collaborative initiative by the California Institute of Technology and local community members in Altadena and Pasadena.[10] The PHOENIX network consists of 28 monitoring stations deployed across the Eaton Fire area, each equipped with a QUANTAQ PM sensor (the same sensor as used in our indoor sampling) that provides one-minute resolution data. As of our sampling date, 11 of the 28 sensors had already been installed and therefore used in our estimation of the outdoor PM concentrations corresponding to each indoor sampling location and time period.

2.3. Dust Sampling and Analytical Method

We collected a total of 30 surface wipe samples, with at least one sample per home, from a variety of indoor surfaces, including carpets, floors, counters, and tables, specifically targeting areas most visibly affected by ash and soot. In some homes, we collected two wipe samples to assess the variability in surface contamination between rooms. Our surface sampling followed the U.S. EPA protocol for lead dust wipe collection (Figure S1b).[39,40] A detailed description of the sampling procedure is provided in the Supplementary Information.

Metal concentrations in surface wipe samples were quantified using an inductively coupled plasma mass spectrometry (ICP-MS) instrument (Thermo iCAP-RQ, ASX-560 Autosampler) following acid digestion at the University of California San Diego's Environmental and Complex Analysis Laboratory (ECAL). Each wipe sample was treated with 10 mL of 1% HNO_3, while the blank sample was processed identically using 20 mL of 1% HNO_3. Samples were then filtered (0.2 μm PTFE) to reduce the total dissolved solids (TDS) content to < 0.2%, which is preferable for ICP-MS analysis.[41] Liquid samples were then diluted 100-fold in 1% HNO_3 prior to analysis and introduced into the ICP-MS system. Calibration metal standards (Inorganic Ventures) were analyzed alongside the samples to quantify target metal concentrations. The acidified water samples were spiked with 50μL (per 10 mL) ICP-MS 71D (Inorganic Ventures) prior to analysis. Each sample was measured in five replicate scans, and standard deviations of these replicates were used to estimate measurement uncertainty. The data were processed using Qtegra ICP-MS software (ThermoFisher Scientific).

2.4. Data Analysis

In this study, we processed TARTA's measurements using a previously developed calibration model that links spectral intensity to the concentrations of target metals. The model was constructed using standard reference metal materials of known concentrations and has been validated and applied in prior field studies to quantify airborne metal particulates across diverse environmental settings.[36] TARTA 2.0 can detect 16 metals (Al, As, Be, Cd, Co, Cr, Cu, Fe, Hg, Li, Mg, Mn, Ni, Pb, V, and Zn), with LODs ranging from 2.1 ng/m^3 for Mn to 53.2 ng/m^3 for As, based on a 30-minute sampling resolution[38] (Table S1).

To evaluate whether differences in environmental conditions and post-fire mitigation behaviors contributed to variations in indoor PM concentrations across homes, we conducted statistical comparisons using nine categorical predictor variables. These variables, derived from on-site observations and homeowner interviews, included window type, proximity to the fire zone, presence of persistent smoke odor, burned external structures, home occupancy status, HVAC use after the fire, closed windows or doors during the fire, indoor air purifier usage, and cleaning activities (Table S2). Each variable was categorized into two or three levels, representing distinct environmental or mitigation conditions. For example, proximity to the fire zone was categorized into three groups: backyard on fire, close (< 100 m), and far (> 100 m) from the fire zone or the nearest burned structure. Cleaning activity was divided into: not cleaned, vacuum cleaning (non-high-efficiency particulate air (HEPA)), and professional cleaning, while window type was classified as single-pane, double-pane, or mixed. All other remaining variables were binary (yes or no). We then used two-sample t-tests for binary variables and one-way analysis of variance (ANOVA) for variables with three levels to evaluate whether indoor PM concentrations differed significantly across categories.[42,43]

Variables that showed statistically significant variation in univariate tests ($p < 0.05$) were subsequently included in a multiple linear regression model to assess whether post-fire indoor PM concentrations varied systematically across combinations of environmental and mitigation conditions in different homes. In this model, homes with the most adverse theoretical conditions—located closest to the fire, not cleaned, with no active air purifiers, and with doors or windows left open during the fire—were used as the reference category. To estimate outdoor $PM_{2.5}$ and PM_{10} concentrations at each household, we applied a wind-adjusted Inverse Distance Weighting (IDW) interpolation using data from the 11 PHOENIX PM monitoring stations.[44,45] The basic IDW formulation is shown in Equation 1:

$$Outdoor\ PM_i = \frac{\sum_{j=1}^{N} \frac{PM_j}{WCD_{ij}^2}}{\sum_{j=1}^{N} \frac{1}{WCD_{ij}^2}} \tag{1}$$

Where $Outdoor\ PM_i$ is the estimated outdoor concentration at household i, PM_j represents the observed concentration at the j_{th} PM station, and $WCD_{i,j}$ is the wind speed and direction-corrected distance weight factor between household *i* and station *j*.

3. Results

3.1. Indoor Air Pollutants

Figure 2 represents the distribution of indoor $PM_{2.5}$, PM_{10}, and BC concentrations across the 19 sampled homes. The U.S. EPA's national ambient air quality standards (NAAQS) for $PM_{2.5}$ and PM_{10} are shown as surrogate benchmarks for reference.[46] Average $PM_{2.5}$ concentrations (Figure 2a) in most homes remained below the EPA's primary (i.e., health-based) annual standard of 9 µg/m³, and all measurements were below the primary 24-hour standard of 35 µg/m³. The narrow interquartile ranges of PM concentrations indicate relatively small variability in indoor levels across the sampling period within each home. Notably, Home 4308 exceeded the 9 µg/m³ reference for most of the sampling period, while Homes 2307-1, 2307-2, and M3311 exhibited transient exceedances during the first and last five minutes of the sampling, likely due to a short-term spike caused by the resuspension of settled dust during instrument setup and the research team's movement within the space. The mean PM_{10} concentrations (Figure 2b) across all homes were generally below the EPA’s 24-hr standard of 150 µg/m³. Nonetheless, four homes (including Homes 2307-2, 4308, 3306, and M3311) showed maximum values exceeding this reference, with the first three also showing elevated $PM_{2.5}$ levels. Of note, while Home 4308 had the highest mean $PM_{2.5}$ concentration, its mean PM_{10} level was not the highest among all homes. The variation of BC concentrations across homes followed a similar pattern as observed for $PM_{2.5}$ (Figure 2c). Most of the BC concentrations measured in this study were below 500 ng/m³, falling well within the lower range reported in previous studies, where average BC levels in occupied residential environments generally remain under 1000 ng/m³.[47,48] Interestingly, Home 1310 showed elevated BC levels (with an average above 1000 ng/m³) despite relatively low PM levels (See Section 4 for further discussion on potential contributing factors). Validated TVOC readings above the LOD of 10 ppb were observed in only three of the 19 households, with average measured concentrations of 42.5, 22.6, and 20.8 ppb in Homes 1307, 2307-1, and 2307-2, respectively. While there are no U.S. regulatory standards for indoor TVOC exposure, the World Health Organization (WHO) recommends an indoor exposure guideline of 125 ppb[49], a level below which all sampled homes fall.

Indoor pollutant levels were found to be typically higher in occupied homes compared to unoccupied ones, where human activities (e.g., cooking, cleaning, smoking, ventilation practices, and infiltration from outdoor air) contribute directly to particle generation.[22,50,51] However, in this study, of the four homes with PM concentrations exceeding the EPA standards, only Home 4308 was occupied at the time of sampling. While Homes 3307 (the benchmark home) and 2310 were also occupied, they comparatively exhibited lower concentrations, suggesting that occupancy alone may not account for elevated particulate levels in fire-affected homes; rather, other factors such as the effectiveness of post-fire cleaning, ventilation practices, and the persistence of residual ash and soot may play a more significant role. This is particularly evident in our benchmark home, which differed from all other sampled residences by implementing the most protective measures (including professional cleaning, closed windows and doors during the fire, and consistent air purifier use) and consistently exhibited among the lowest concentrations of indoor PM (see Section 4 for further discussion).

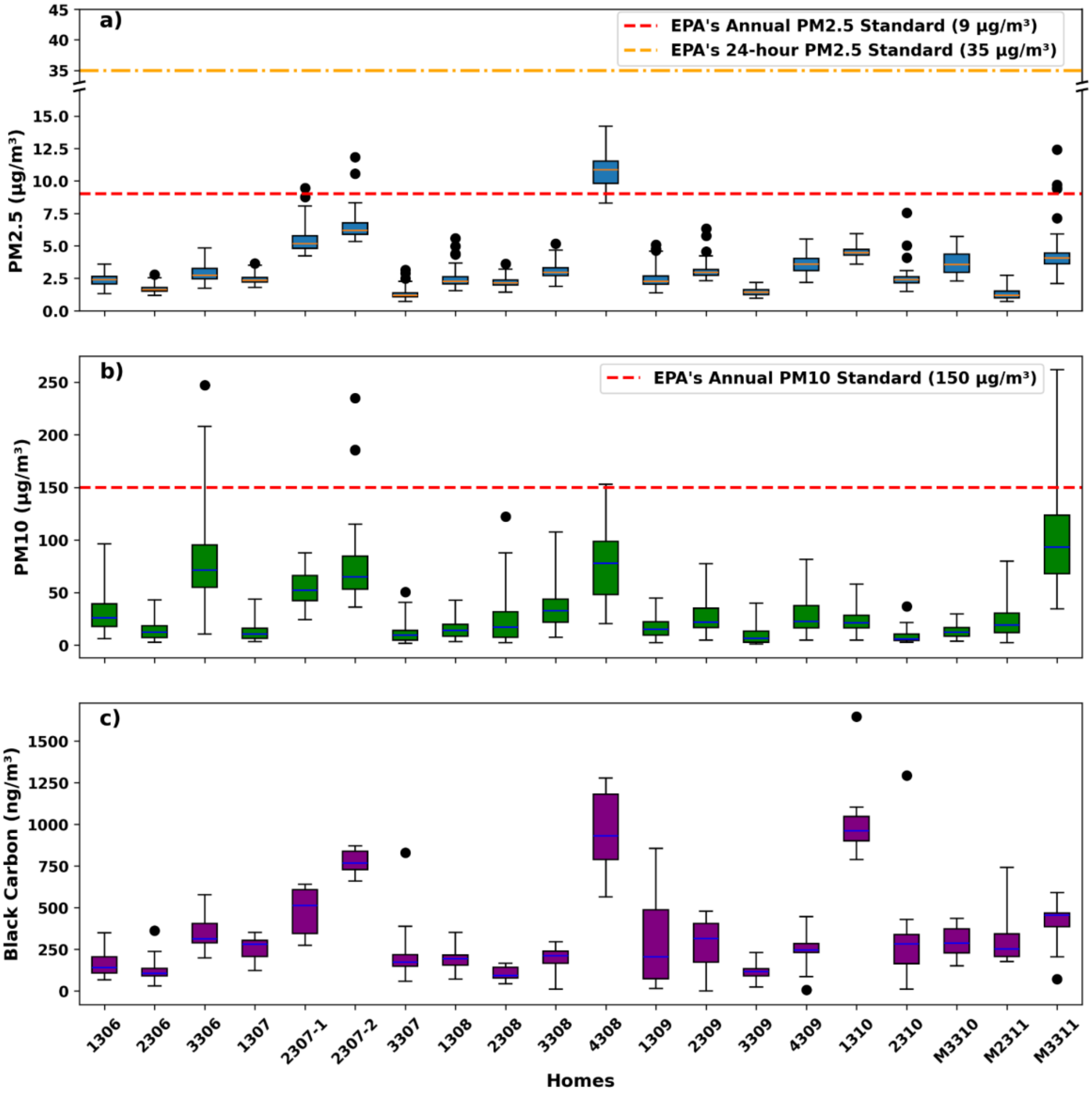

Figure 2. Variation of indoor concentrations of a) $PM_{2.5}$ (y axis is broken from 17 to 33), b) PM_{10}, and c) BC across the sampled homes.

Table 1 presents the average and standard deviation of elemental concentrations measured in indoor air across the 19 sampled homes. Although TARTA can detect up to 16 metal species (Table S1), only Al, Fe, and Mg were found above the instrument's LODs in a majority of homes, with an average and standard deviation of 223.8 ± 153.9, 61.9 ± 52.3, and 305.1 ± 168.7 across all homes, respectively. The greatest Mg concentration was observed at Home 1306 (531.6 ng/m³), while Fe peaked at 183.8 ng/m³ at Home 3307 and Al peaked at 519.1 ng/m³ at Home M2311. Cu and Cr were detected in only a few homes: Cu was found in three homes, with a higher level of 5.9 ng/m³ at Home 3309, where Cr was also detected at an average of 5.8 ng/m³. The limited detection of other metals is not unexpected, as ambient elemental concentrations in the Eaton and Palisades regions were reported to reflect generally low urban levels. For example, SCAQMD's analysis of filter samples using ICP-MS, which collected TSP similar to TARTA's configuration, reported average ambient concentrations on March 8th and 11th of of 515 ng/m³ (Al), 447 ng/m³ (Fe), 242 ng/m³ (Mg), 12 ng/m³ (Cu), and 3 ng/m³ (Cr) at the Eaton sites and 1584 ng/m³ (Al), 1513 ng/m^3 (Fe), 688 ng/m^3 (Mg), 12 ng/m^3 (Cu), and 5 ng/m^3 (Cr) at the Palisades sites [9] (a summary of the other elemental results from SCAQMD can be found in Table S3). Although few metals were detected by TARTA, this is a favorable outcome, as the measured concentrations of metals with established EPA reference inhalation concentrations (RfCs), including Be, Co, Cr(VI), Hg, Mn, Ni, Pb, and V, were all below their respective RfC thresholds.[52] However, residual ash deposited on indoor surfaces remains a concern (see Sections 3.3 and 4.2), though the low airborne metal concentrations suggest these particles are not being significantly resuspended into indoor air at the time of sampling.

Table 1. Airborne elemental concentrations measured across 19 homes (ng/m^3). Values represent mean ± standard deviation based on four 30-minute indoor air samples collected at each home. [a]

	1306	**2306**	**3306**	**1307**	**2307-1**	**2307-2**	**3307**	**1308**	**2308**	**3308**
Al	<LOD	<LOD	392.2 ± 168.7	<LOD	<LOD	<LOD	<LOD	201.9 ± 116.2	<LOD	150.1 ± 126.7
Cr	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD
Cu	<LOD	<LOD	3.2± 1.5	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD
Fe	18.9 ± 12.2	22.5 ± 11.3	96.1 ± 55.6	73.9 ± 24.6	37.1 ± 25.3	<LOD	183.8 ± 133.4	52.3 ± 28.1	<LOD	<LOD
Mg	531.6 ± 166	249.6 ± 43	524.8 ± 180.9	470.6 ± 113	220.9 ± 93.4	304.8 ± 16.8	176.5 ± 76.8	310.9 ± 208.8	121.9 ± 77.5	362.8 ± 88.9
	4308	**1309**	**2309**	**3309**	**4309**	**1310**	**2310**	**M3310**	**M2311**	**M3311**
Al	<LOD	121.9 ± 95.4	<LOD	206.1 ± 19.8	138.1 ± 108.7	<LOD	<LOD	<LOD	519.1 ± 27.2	61.3 ± 46.1
Cr	<LOD	<LOD	<LOD	5.8 ± 3.5	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD
Cu	<LOD	<LOD	<LOD	5.9 ± 3.4	<LOD	<LOD	<LOD	<LOD	2.6 ± 1.2	<LOD
Fe	26.2 ± 5.5	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	48.2 ± 24.3
Mg	362.8 ± 88.9	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD

[a] The LODs of TARTA are 14.8 ng/m^3 (Al), 4.2 ng/m^3 (Cr), 2.4 ng/m^3 (Cu), 11.8 ng/m^3 (Fe), and 7.6 ng/m^3 (Mg). "<LOD" in the table denotes concentrations below these LODs.

3.2. Indoor to Outdoor (I/O) ratio of PM concentrations

In this section, we use indoor-to-outdoor (I/O) PM ratios to assess how outdoor air quality affects indoor environments. An I/O ratio > 1 indicates indoor air is more polluted than outdoors, while an I/O < 1 suggests indoor air is cleaner than outdoors.[53] Figure 3 presents I/O ratios for PM$_{2.5}$ and PM$_{10}$ across the sampled homes. These results are limited to homes in the Eaton fire-affected area, where both indoor and outdoor PM levels were measured using the same QuantAQ sensors. The average I/O ratio for $PM_{2.5}$ was 1.08, with six homes exhibiting ratios > 1 (Homes 3306, 2307-1, 2307-2, 4308, 2309, and 1310). In contrast, PM_{10} ratios were higher, with an average of 2.99 and 12 out of 16 homes exhibiting I/O > 1. Three homes (3306, 2307-1, and 2307-2) had PM_{10} I/O ratios exceeding 5, with Home 3306 reaching a peak ratio of approximately 14.

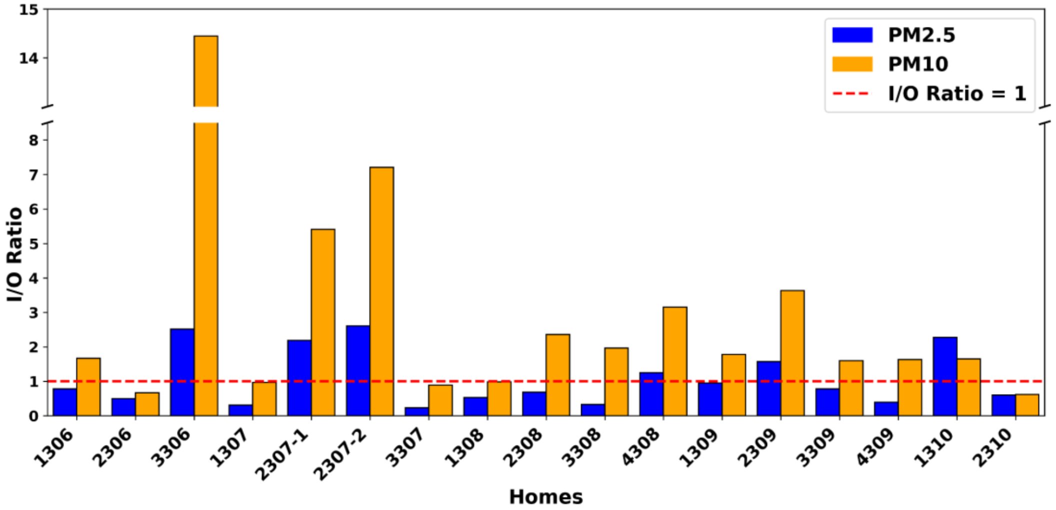

Figure 3. Indoor to outdoor ratios of $PM_{2.5}$ and PM_{10} concentrations over all sampled homes in the Eaton Fire region. Note that the y-axis is broken from 8.5 to 13.5.

Previous studies have reported that I/O ratios typically remain below 1 during non-fire days, unless periodic indoor sources such as cooking or heating are present.[51,54,55] In contrast, our study observed I/O ratios > 1 in many homes, particularly for PM_{10}. Most of the homes in our study were unoccupied, had remained closed since the fire, and had not undergone any cleaning prior to sampling (with the exception of Homes 4308, 3307, and 2310). As such, the elevated I/O ratios are unlikely to result from occupant behavior or active indoor emissions during the sampling period, but instead likely reflect the legacy of the fire smoke, as further explored in Section 4.1.

3.3. Surface Contaminants of Metals

Figure 4 presents the elemental concentrations of 14 metals in 30 surface wipe samples collected across all homes, including from carpets, tables, floors, countertops, and window sills. To ensure quantification accuracy, all reported metal concentrations were corrected by subtracting concentrations measured in field blanks, which were obtained as described in section 2.3. Among all elements, K was the most abundant, dominating in 23 of the 30 samples, with an average concentration of 2235.4 ± 3054.2 µg/ft² (Table S4). Mg, Al, and Fe were also prominent, averaging 1995.1 ± 3317.2, 1823.5 ± 2457.6, and 494.4 ± 694.3 µg/ft², respectively. Their consistent presence across homes indicates widespread indoor contamination resulting from the inflation and deposition of local soil dust and burned vegetation residuals following the fires.[56,57]

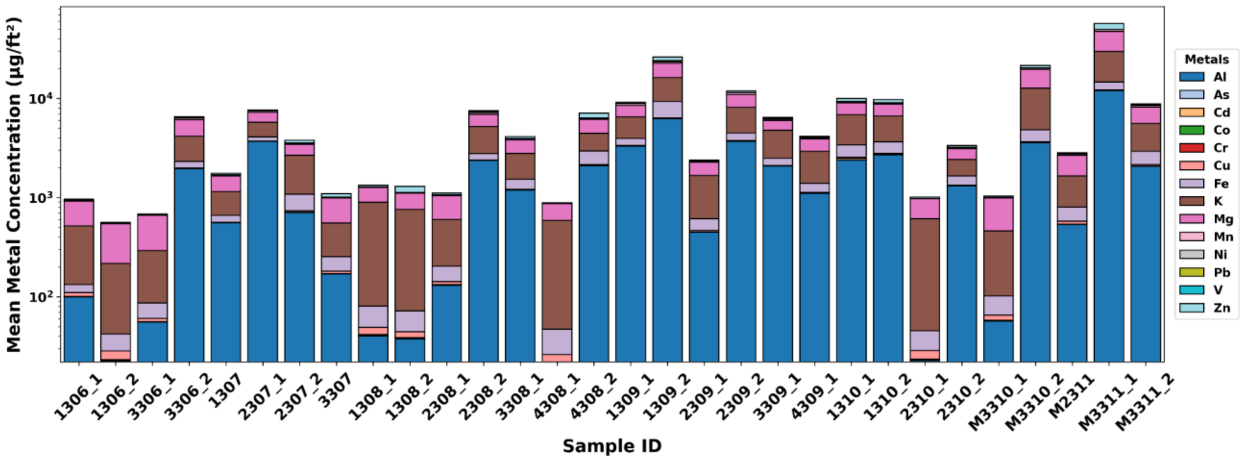

Figure 4. Concentration of 14 quantified metals from 30 surface wipe samples across all sampled households. Note that the y-axis is on a $\log_{10}$ scale.

In addition to these crustal and biomass-related elements, Zn, Cu, and Mn were also detected, with average concentrations of 505.5 ± 1300.2, 26.6 ± 28.1, and 235.9 ± 450.1 µg/ft² over all sampled homes, respectively. Zn and Cu are widely present in household infrastructure and vehicles, such as plumbing, wiring, roofing, and brake components, and are known to be released during combustion of these materials in WUI fires.[56] Holder et al. (2023) found that Cu and Zn emission factors from vehicle combustion were over 60 and 400 times higher, respectively, than those from biomass burning.[12] Mn, while naturally occurring in soils and vegetation, is also used as an additive in treated wood, steel, and metal coatings, which can volatilize at the high temperatures generated in structural fires. [56,58]

Pb also emerged as a key anthropogenic contaminant in our surface wipe samples. As shown in Figure 5 (floors and surface samples) and S2 (window sill samples), Pb concentrations exceeded the EPA's dust lead clearance levels (DLCL) of 5 µg/ft² for floors and surfaces and 40 µg/ft² for window sills in multiple households, with values at least four times greater than the standard. Given that most homes in the fire-affected regions were likely built before 1978 (the year that the U.S. banned the residential use of Pb-based paints), as evidenced by all 19 sampled homes in this study, the probable use of such paints in these buildings could be a significant contributor to the elevated Pb levels detected on indoor surfaces.[59–61]. Additionally, the combustion of electric vehicles and internal combustion engine vehicles can also serve as a contributor to the observed Pb contamination.[62]

Comparisons across surface types further reveal distinct patterns of metal deposition, reflecting how material characteristics influence indoor contaminant distribution. Window sills frequently exhibited higher metal concentrations than adjacent surfaces, as illustrated by Sample 4308_2 versus 4308_1 and 2310_2 versus 2310_1, likely due to direct deposition from outdoor air infiltration (Figures 5 and S2). Similarly, entrance floor surfaces (e.g., Samples 2309_2, 1310_1, and M3310_1) exhibited elevated metal concentrations, likely due to particulate intrusion through door gaps or door opening during the fire. In contrast, wardrobes and closets (e.g., Samples 2310_2 and M3310_1) had lower concentrations, likely due to their enclosed and protected positions. Carpeted surfaces (e.g., Samples 1306_1, 3306_1, and 4308_1) consistently exhibited low metal concentrations compared to other indoor surfaces. This may be attributed to their fibrous and porous structure, which both limits surface deposition and reduces recovery efficiency during wipe sampling. Prior research findings note that while carpets can act as long-term particle reservoirs, they are less effective for surface sampling compared to smooth, hard materials.[63,64]

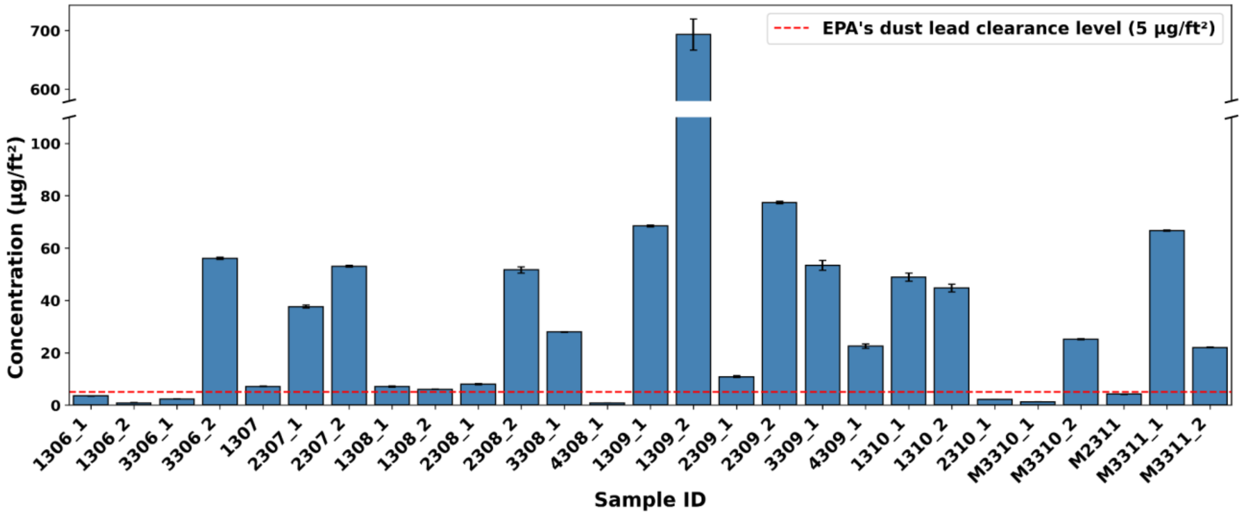

Figure 5. Pb concentrations in wipe samples taken from floors and surfaces across sampled homes. Note that the y-axis is broken from 110 to 580. A similar figure of Pb concentration taken from window sill samples can be found in Figure S2.

4. Discussion

4.1. Possible Causes of I/O Ratios >1 Two Months Post-Fire

To understand the drivers of PM I/O ratios > 1 in many of our sampled homes after two months of the fire (Figure 3), it is important to assess whether they are due to changes in indoor concentrations, outdoor concentrations, or both. In particular, an I/O ratio > 1 may reflect low outdoor PM levels rather than high indoor levels during the sampling period. To investigate this, Table S5 presents the average and standard deviation of $PM_{2.5}$ and PM_{10} concentrations measured at both indoor and outdoor locations across all homes. The variation in outdoor PM concentrations was relatively small during our sampling periods, with a total mean and standard deviation of 4.31 ± 2.71 µg/m^3 for $PM_{2.5}$ and 13.12 ± 5.11 µg/m^3 for PM_{10}. This consistency of outdoor PM levels across homes suggests that the higher I/O ratios observed in some homes were primarily driven by elevated indoor PM concentrations there, rather than by fluctuations in outdoor conditions during the sampling period.

As mentioned in Section 3.2, most homes were unoccupied and had no active indoor sources, which suggests that the elevated indoor PM concentrations were due to the lingering presence of fire debris, possibly driven by several pathways. First is the resuspension of settled fire residues.[22] Walked-on floors and lightly disturbed surfaces can re-aerosolize particles even under minimal activity, such as the movement of our sampling team. Second, limited ventilation in window and door-sealed homes suppresses dilution, allowing particles to accumulate, where low air exchange traps resuspended PM indoors and prevents them from leaving.[65] Third, the re-emission of SVOCs from indoor surfaces that became contaminated during the fire may also contribute. Compounds that sorbed onto these surfaces can slowly desorb, repartition into the gas phase, and re-adsorb onto airborne particles, sustaining indoor PM concentrations even in the absence of new emissions.[24,66] Finally, procedural factors, such as the sampling team not removing shoes upon entry, may have inadvertently introduced outdoor dust or disturbed indoor deposits, further elevating indoor PM levels.[67]

The pronounced elevation of PM_{10} I/O ratios compared to $PM_{2.5}$ arises primarily from size-dependent resuspension dynamics. PM_{10} particles, owing to their greater mass and weaker adhesive forces, detach far more easily than fine particles under similar mechanical forces. In a controlled chamber study, PM_{10} resuspension rates were found to be an order of magnitude higher than $PM_{2.5}$ ($\sim 0.25\ h^{-1}$ (PM_{10}) vs. $\sim 0.02\ h^{-1}$ ($PM_{2.5}$)) during identical walking simulations.[68]

4.2. Key Factors Driving PM Variation Across Homes

Our individual parametric tests suggested six variables (including proximity to the fire zone, presence of persistent smoke odor, burned external structures, closed windows or doors during the fire, indoor air purifier usage, and cleaning activity) were each associated with significant variations in PM concentrations between their respective categories across the sampled homes ($p < 0.05$) (Table S2). However, in the multiple linear regression model, we excluded burned external structures (due to overlapping with fire proximity) and persistent smoke odor (due to its subjective nature) as independent variables. The final model was built upon the remaining four variables, which explain 73% and 86% of the variance in indoor PM_{10} and $PM_{2.5}$ concentrations, respectively, across all sampled homes (Table 2 and Table S6).

Table 2. Multiple linear regression results on average indoor PM_{10} concentrations.

Variables	Reference Condition	Regression coefficients [a]	p-value [b]
Model intercept	-	59.3	***
Distance from fire zone <100 m	Backyard on fire	-35.4	***
Distance from fire zone >100 m		-28.1	***
Professional cleaning	Not cleaned	-14.6	0.38
Vacuum cleaning (Non-HEPA)		40.3	***
Active air purifiers	w/o air purifier	-34.9	***
Windows or doors were fully closed during the fire	Windows or doors were opened during the fire	-11.5	0.33
		Number of samples = 20	**$R^2 = 0.73$**

[a] Regression coefficients represent the change in indoor PM_{10} ($\mu g/m^3$) across homes associated with each condition, relative to the reference category. A negative coefficient indicates a reduction in PM_{10} compared to the reference.

[b] Statistically significant p-values (i.e., $p < 0.05$) are indicated as ***.

Proximity to the fire zone emerged as one of the major predictors of indoor PM_{10} levels. Compared to homes with their backyard directly on fire, homes located less than 100 meters away from the fire had an average reduction of 35.4 µg/m³ ($p < 0.05$), and those more than 100 meters away showed a reduction of 28.1 µg/m³ ($p < 0.05$). Although both distance categories were associated with significantly lower PM_{10} levels relative to the reference group, the comparable reductions in PM_{10} levels suggest that additional site-specific factors can influence the difference of post-fire PM levels in different homes, such as wind direction during the fire, the orientation of windows and doors relative to prevailing winds, and household infiltration characteristics.[51]

The use of air purifiers after the fire was associated with a substantial reduction of 34.9 µg/m³ in PM_{10} ($p < 0.05$) compared to homes without them. This finding reinforces the effectiveness of active filtration systems in mitigating PM exposure in fire-affected homes, consistent with recent studies that have demonstrated the efficacy of HEPA purifiers in reducing indoor PM levels during and after high-pollution events, including wildfire episodes and dust storms.[69–71]

The impact of cleaning activities on indoor PM_{10} concentrations varied substantially between homes. While professional cleaning was associated with a reduction of 14.6 µg/m³ relative to uncleaned homes, the reducing effect was not significant ($p = 0.38$). In contrast, non-HEPA vacuum cleaning was associated with a significant increase of 40.3 µg/m³ in indoor PM_{10} concentration ($p < 0.05$), suggesting that standard household vacuums may resuspend ash and soot particles, worsening the IAQ. More specifically, Homes 3307 and 2310 underwent professional cleaning, which included the use of wet cloths, mops, and HEPA-filter vacuums, methods known to limit particle resuspension.[72,73] These homes were associated not only with lower indoor PM_{10} levels but also with surface lead dust concentrations below the EPA standards, as mentioned in section 3.3. In contrast, Home 4308 was cleaned using a common household vacuum without a HEPA filter, along with wet cloth wiping. Previous studies have shown that such vacuums are ineffective at capturing fine particles and may recirculate contaminants into the indoor air.[74,75] Additionally, residents at Home 4308 reported using a leaf blower to clear ash and soot from the front lawn, which could further contribute to indoor particle loading and surface lead dust by transporting outdoor contaminants indoors.

Keeping windows and doors fully closed during the fire resulted in a modest reduction of 11.5 µg/m³ in PM_{10} compared to homes with openings, though this effect is not a significant predictor of PM_{10} ($p = 0.33$). The limited association of this variable with different PM_{10} levels across homes may be explained by uncontrolled variability in building leakage and the timing of window closure after the fire started, both of which require further investigation that was not systematically documented in this study. Nonetheless, previous research supports our findings that maintaining a well-sealed building envelope (by keeping windows and doors closed and sealing cracks) can significantly reduce indoor PM levels during wildfire events, especially compared to homes with higher infiltration rates.[13]

The $PM_{2.5}$ regression model shown in Table S6, which included the same four predictor variables, exhibited coefficient signs consistent with those in the PM_{10} model (i.e., $PM_{2.5}$ levels tended to be lower in homes equipped with air purifiers, located farther from the fire, and with windows and doors closed during the fire; and higher in homes using non-HEPA vacuums or situated closer to the fire). However, only the categories of <100 m distance, non-HEPA vacuuming, and closed windows and doors during the fire were statistically significant in explaining the variation in average $PM_{2.5}$ concentrations across homes.

4.3. Hazardous Air Pollutants in Surface Wipes

Initial air quality reports from the early days of the Eaton and Palisades fires documented acute spikes in airborne Pb concentrations. At the Pico Rivera ASCENT station (roughly 30 miles from the Eaton fire region), for instance, Pb levels exceeded 100 times the typical background average on January 9th, indicating substantial short-term emissions from the combustion of manmade materials.[7] However, these spikes subsided within approximately 48 hours of fire onset. Subsequent continuous ambient air monitoring conducted by the SCAQMD beginning in February showed that concentrations of the seven metals classified as hazardous air pollutants (HAPs) by the U.S. EPA (including As, Cd, Cr, Co, Mn, Ni, and Pb) had returned to typical background levels at the Los Angeles Basin [9] (Figure S3 and Table S3). Similarly, our indoor air measurements, conducted approximately two months after the fires, detected no elevated concentrations of these metals; most were below TARTA's LODs or comparable to concurrent outdoor levels (see section 3.1).

However, the absence of elevated airborne metal concentrations does not imply the absence of indoor residual contamination and exposure risks. Our surface wipe samples revealed that all seven HAP metals were detected in multiple homes (Figure S3). These metals likely infiltrated when ambient concentrations were high and settled indoors along with ash and soot during or shortly after the fire. Among the HAP metals detected in surface wipe samples, Pb and Mn were the most abundant metals and exhibited elevated levels in a subset of homes (including 16 out of 30 wipe samples) compared to the benchmark home (Home 3307). This pattern aligns with SCAQMD ambient air measurements during the post-fire monitoring period, where Pb, Mn, and Cr were the most elevated metals among the HAPs (Figure S3). A notable example is Sample 1309_2, collected from a garage floor, which shows one of the highest concentrations of all seven HAP metals, particularly Pb and Mn, highlighting how semi-exposed or outdoor areas can serve as metal accumulation zones.

These findings suggest that even after airborne levels of HAP metals have declined, indoor surfaces may act as persistent reservoirs, capable of resuspending metal particles into the air through certain environmental disturbances, such as walking, sweeping, and vacuuming. This risk is especially pronounced in homes that have not undergone thorough post-fire remediation and poses particular concerns for vulnerable populations, including elders and children, who are more susceptible to the harm through inhalation and dermal exposure. In addition to indoor sources, post-fire soil and surrounding ground surfaces represent another potential source of re-exposure. The accumulated ash and soot in these areas can be remobilized by strong winds, debris removal, or the use of leaf blowers, contributing not only to outdoor air contamination but also to subsequent indoor exposure through infiltration.

5. Summary and Implications

This study presents a comprehensive evaluation of indoor air quality and surface contamination in 19 homes affected by the Eaton and Palisades fires, approximately two months after fire events. Indoor PM concentrations were generally below U.S. EPA ambient air standards across homes, with overall average $PM_{2.5}$ and PM_{10} concentrations of 3.45 and 31.66 µg/m³, respectively. BC levels were found to be below typical values reported for occupied homes, reflecting the unoccupied status of most sampled homes. Notably, I/O ratios > 1 were observed in many homes, particularly for PM_{10}, suggesting that limited ventilation may have trapped fire-related debris indoors and coarse particles had a greater resuspension potential due to indoor activities. Regression analysis of key environmental and behavioral factors revealed that proximity to the fire zone, closed windows or doors during the fire, indoor air purifier usage, and cleaning activity significantly influenced indoor PM levels, explaining 73% of PM_{10} and 86% of $PM_{2.5}$ variations across all homes. Although indoor airborne metal concentrations during the study period were within typical background ranges, surface wipe samples revealed widespread, and in some cases elevated, levels of metals on indoor surfaces. Detected metals appear to originate from different WUI fire sources: K, Mg, and Fe were consistent with vegetation combustion, while Pb, Zn, Cu, and Cr likely originated from infrastructure materials. Presence of HAP metals, including Pb and Mn, on indoor floors and surfaces two months post-fire, exhibits that these metals can persist on surfaces and act as potential reservoirs for re-exposure, especially through resuspension, posing long-term health risks even when air quality appears improved. Of note, surface metal concentrations varied by location, with window sills and entry floors showing the highest levels, and carpeted or enclosed interior areas the lowest.

This study is subject to several limitations, including the relatively small sample size, the limited number of professionally cleaned homes as of sampling period, and the absence of sampling at homes located much farther from the fires, which may have constrained our interpretation. Nonetheless, consistent with previous studies, our results emphasize the importance of comprehensive post-fire cleaning and remediation practices, as homes that underwent wet wiping and HEPA vacuuming showed reduced indoor PM and metal levels.

As we move into the second phase of post-fire recovery, and with ambient air quality expected to continue improving, future work will address community concerns regarding elevated indoor air pollution during hot weather due to higher ventilation rates and increased off-gassing from fire-affected surfaces and furniture. Additionally, with more residents returning and a greater number of homes having undergone professional cleaning, upcoming efforts will focus on evaluating post-remediation indoor air quality and surface contamination. These assessments will be compared to baseline conditions reported in this manuscript and will also be used to evaluate the effectiveness of different cleaning strategies across households.

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HL and EG designed the research. EG, JC, and LF conducted the field sampling. EG analyzed data. EG and HL led the manuscript writing. All authors approved the final version.

Note

During the preparation of this work, the authors used ChatGPT to enhance the clarity, conciseness, and academic tone of the manuscript. After using this tool, the authors reviewed and edited the content as needed.

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Supplementary Information

SUPPLEMENTARY TEXT

- **Questionnaire to recruit volunteers**

Post-fire indoor air monitoring

Coordinating research activities for the research team from San Diego State University (Hanyang Li, hli6@sdsu.edu). The team will spend approximately four hours at your home, using battery-powered, portable air quality monitors to measure particulate matter (PM), volatile organic compounds (VOCs), and toxic metals inside and outside your home. They will also test air purifiers to evaluate their effectiveness in reducing pollution. Their presence will cause minimal disruption, and you can continue your normal activities while monitoring takes place. Your participation will help provide valuable data to improve post-fire recovery strategies and protect community health. The information provided below will be shared with the research team. Monitoring is expected to start as early as 3/3/25.

* Indicates required question

1. **Your Name** *
2. **Phone Number** *
3. **Email** *
4. **Address of home in/near the Eaton Fire impacted area** *
5. Can you smell wood smoke or other burning odors inside your home? *
    - Yes
    - No
    - Other:
6. What is your best estimate of your home's fire damage status? *
    - Destroyed (>50%)
    - Major (26% – 50%)
    - Minor (10% – 25%)
    - Affected (1% – 9%).
    - No structural damage, but impacted by smoke and ash
    - Other:
7. When did you first return to your fire-affected home? *
    - Before evacuation orders were lifted
    - Within a day after the evacuation orders were lifted
    - Within a week after the evacuation orders were lifted
    - More than a week after the evacuation orders were lifted
8. Do you currently have power in your home? *
    - Yes, electricity was not damaged during the fire
    - Yes, grid electricity has been restored after the fire
    - Yes, we rely on a generator provided by the county
    - No, power is still unavailable
    - Other:
9. Have you already cleaned the ash inside your home? *
    - Yes, by wiping surfaces
    - Yes, using air purifiers
    - Yes, by vacuuming or washing
    - Not yet

- Other:

10. Do you have plans to do any reconstruction or repairs for structural damage? *
    - Yes
    - No
    - Other:

11. I have the authority and give consent for researchers to access and take/remove samples of soil and ash from my property at the address given above. (If no, no samples will be collected.) *
    - Yes
    - No

12. Is there anything else you would like us to know about your home's condition or your concerns about indoor air quality?

- **Sampling procedure of the dust wipes**

Our surface sampling followed the U.S. EPA protocol for lead dust wipe collection, using a 3D-printed template with a 10 × 10 cm^2 opening to define the sampling area. For each sample, the template was secured to the target surface using painter's tape. To prevent cross-contamination, a new pair of gloves and a Kimtech delicate task wiper, pre-wetted with deionized water in a petri dish, were used for each sample. To collect the dust sample, we placed the wipe at the corner of the template and started a sideways (i.e., horizontal) motion. Then, we folded the wipe and used the clean side to wipe the surface in a forward and backward (i.e., vertical) motion, starting from the same corner. Finally, we folded the wipe again and used another clean side to clean the interior edges of the sampling area. Once the sample collection was done, the wipe was folded inward with the sample side enclosed, placed in a sanitized centrifuge tube, and sealed. All the tubes were labeled with date, house ID, location of sample collection, and stored in a transport container. To assess background contamination, we collected field blanks at the beginning and end of each sampling day. These wipes were handled identically to sample wipes but were not used to contact any surfaces. All samples were stored in a freezer upon return to the laboratory prior to analysis.

SUPPLEMENTARY TABLES

Table S1. TARTA 2.0 LODs based on 30-minute sampling duration. All the units are in ng/m³.

Metal	**Al**	**As**	**Be**	**Cd**	**Co**	**Cr**	**Cu**	**Fe**	**Hg**	**Li**	**Mg**	**Mn**	**Ni**	**Pb**	**V**	**Zn**
LOD (1σ)	14.8	53.4	10.4	20	3.4	4.2	2.4	11.8	16.8	4	7.6	2.1	20.8	14	4.1	2.8

Table S2. Categorical variables and t-test results for indoor PM concentration differences across homes.

Home ID	**Type of windows**	**Burned external properties**	**Proximity to the fire**	**Reoccupation**	**Persistent smoke odor**	**Air purifier usage**	**HVAC use after the fire**	**Closed windows and doors during the fire**	**Cleaning activity**
3311	Single-pane	External structure	Backyard on fire	No	Yes	No	No	No	Not cleaned
1306	Double-pane	No	<100m	No	Yes	No	No	Yes	Not cleaned
2306	Double-pane	No	>100m	No	No	No	No	Yes	Not cleaned
1307	Single-pane	No	>100m	No	Yes	No	No	Yes	Not cleaned
2307	Single-pane	External structure	Backyard on fire	No	Yes	No	No	No	Not cleaned
3307	Double-pane	No	>100m	Yes	No	Yes	Yes	Yes	Professional cleaning
1308	Mixed	No	<100m	No	Yes	No	No	Yes	Not cleaned
2308	Single-pane	No	<100m	No	Yes	No	Yes	Yes	Not cleaned
Home ID	**Type of windows**	**Burned external properties**	**Proximity to the fire**	**Reoccupation**	**Persistent smoke odor**	**Air purifier usage**	**HVAC use after the fire**	**Closed windows and doors during the fire**	**Cleaning activity**

3308	Mixed	No	>100m	No	Yes	No	No	Yes	Not cleaned
4308	Double-pane	No	<100m	Yes	No	No	Yes	No	Vacuum cleaning
1309	Single-pane	No	<100m	No	Yes	No	No	Yes	Not cleaned
2309	Single-pane	No	<100m	No	Yes	No	No	No	Not cleaned
3309	Single-pane	No	>100m	No	No	Yes	No	Yes	Not cleaned
4309	Mixed	No	<100m	No	Yes	No	No	No	Not cleaned
1310	Mixed	No	>100m	No	No	No	No	Yes	Not cleaned
2310	Single-pane	No	>100m	Yes	No	Yes	No	Yes	Professional cleaning
3310	Double-pane	External structure	Backyard on fire	No	Yes	Yes	No	Yes	Not cleaned
2311	Mixed	No	Backyard on fire	No	No	Yes	Yes	Yes	Not cleaned
3306	Mixed	External structure	Backyard on fire	No	Yes	No	No	Yes	Not cleaned
$PM_{2.5}$ p-value[1]	***	***	***	***	***	***	***	***	***
PM_{10} p-value[1]	0.35	***	***	0.92	***	***	0.91	***	***

[1] Statistically significant p-values (i.e., $p < 0.05$) are indicated as ***.

Table S3. Elemental concentration detected at the stationary air monitoring stations of South Coast AQMD. Note: Only metals detectable by the TARTA are included.

LEGEND NAME	SAMPLE DATE	Al	As	Be	Cd	Co	Cr	Cu	Fe	Li	Mg	Mn	Ni	Pb	V	Zn
Eaton Site #3	**3/8/2025**	351.59	0.21	ND	ND	0.16	3.04	6.32	385.49	ND	244.4	6.53	1.08	2.45	0.69	19.67

Eaton Site #2	**3/8/2025**	486.52	0.28	ND	ND	0.24	2.42	15.45	477.11	0.67	260.7	8.64	1.18	3.88	0.94	24.73
Eaton Site #1	**3/8/2025**	568.02	0.2	ND	ND	0.2	3.8	26.57	375.02	ND	312.37	7.78	2.11	1.56	0.72	28.61
Eaton Site #3	**3/11/2025**	468	0.16	ND	ND	0.19	2.9	5.1	410.41	ND	147.1	8.14	0.98	2.26	0.83	19.7
Eaton Site #2	**3/11/2025**	700.72	0.26	ND	ND	0.27	3.83	6.57	586.01	ND	245.23	11.56	1.49	3.58	1.17	28.21
Palisades Site #1	**3/8/2025**	2331.4	0.62	0.09	0.15	1.01	5.63	11.47	2102	1.59	980.07	39.05	3.1	3.47	4.31	64.16
Palisades Site #2	**3/8/2025**	906.39	0.39	ND	0.09	0.43	4.09	12.02	819.89	0.73	567.24	15.43	2.21	2.51	1.62	33.33
Palisades Site #1	**3/11/2025**	1514.1	0.51	ND	ND	0.63	5.89	13.72	1616.2	1.15	516.82	26.29	2.9	3.32	2.92	45.86
Average (Eaton)		515 ± 129	0.2 ± 0.05	ND	ND	0.2 ± 0.04	3.2 ± 0.6	12 ± 9	447 ± 87	ND	242 ± 60	8.5 ± 1.9	1.4 ± 0.5	2.7 ± 1	0.9 ± 0.2	24 ± 4
Average (Palisades)		1584 ± 715	0.5 ± 0.1	ND	0.12 ± 0.04	0.7 ± 0.3	5.2 ± 1	12 ± 1	1513 ± 647	1.2 ± 0.4	688 ± 254	27 ± 12	2.7 ± 0.5	3 ± 0.5	3 ± 1	48 ± 16

Table S4. Metal concentrations in 30 wipe samples across sampled homes. Values represent mean ± standard deviation based on five repeated analyses of each sample.

Sample ID	Al	As	Cd	Co	Cr	Cu	Fe	K	Mg	Mn	Ni	Pb	V	Zn
1306_1	99.28 ± 7.09	0.87 ± 0.07	0.16 ± 0.03	0.12 ± 0.02	0.17 ± 0.01	9.96 ± 0.06	22.42 ± 2.28	385.73 ± 11.03	400.17 ± 14.40	9.80 ± 0.29	1.66 ± 0.13	3.46 ± 0.04	0.66 ± 0.07	28.00 ± 0.27
1306_2	22.41 ± 2.48	0.60 ± 0.01	0.12 ± 0.02	0.03 ± 0.00	0.06 ± 0.03	5.19 ± 0.11	13.94 ± 1.24	174.74 ± 7.33	326.59 ± 3.96	3.26 ± 0.17	1.23 ± 0.12	0.86 ± 0.01	0.26 ± 0.06	11.32 ± 0.12
3306_1	55.59 ± 2.08	0.38 ± 0.03	0.04 ± 0.02	0.03 ± 0.00	0.09 ± 0.03	4.43 ± 0.04	26.13 ± 2.41	205.30 ± 8.68	371.69 ± 7.93	6.94 ± 0.19	0.13 ± 0.06	2.31 ± 0.08	0.12 ± 0.02	10.58 ± 0.55
3306_2	1960.58 ± 32.18	2.44 ± 0.08	0.57 ± 0.04	1.10 ± 0.03	1.19 ± 0.04	21.66 ± 0.42	332.50 ± 7.84	1849.62 ± 24.66	1960.60 ± 42.31	183.86 ± 2.07	2.06 ± 0.14	56.06 ± 0.41	3.42 ± 0.12	172.82 ± 1.75
1307	558.50 ± 21.21	0.44 ± 0.07	0.13 ± 0.03	0.31 ± 0.02	0.17 ± 0.03	4.98 ± 0.10	95.27 ± 1.56	491.12 ± 8.56	507.00 ± 16.74	32.42 ± 0.62	2.02 ± 0.11	7.12 ± 0.07	0.48 ± 0.05	55.84 ± 0.38
2307_1	3683.77 ± 39.42	1.83 ± 0.07	0.38 ± 0.05	1.48 ± 0.04	0.76 ± 0.02	17.41 ± 0.37	383.60 ± 8.77	1676.00 ± 29.44	1473.05 ± 33.05	239.23 ± 3.00	3.30 ± 0.22	37.62 ± 0.57	3.83 ± 0.12	124.30 ± 1.42
2307_2	708.02 ± 22.40	1.91 ± 0.11	1.03 ± 0.08	0.38 ± 0.04	0.91 ± 0.06	22.25 ± 0.42	346.28 ± 6.74	1595.81 ± 28.05	778.65 ± 27.35	55.52 ± 0.45	3.27 ± 0.12	53.01 ± 0.36	0.84 ± 0.11	210.85 ± 1.02
3307	170.22 ± 5.85	0.61 ± 0.08	0.09 ± 0.04	0.12 ± 0.01	0.23 ± 0.02	9.98 ± 0.22	71.48 ± 3.33	302.06 ± 9.71	437.08 ± 14.64	13.41 ± 0.16	0.67 ± 0.04	5.34 ± 0.17	0.25 ± 0.06	85.73 ± 1.26
1308_1	40.28 ± 2.82	0.65 ± 0.05	0.51 ± 0.05	0.07 ± 0.01	0.18 ± 0.03	7.40 ± 0.22	31.09 ± 2.59	816.64 ± 32.70	371.30 ± 8.83	3.96 ± 0.10	1.92 ± 0.15	7.08 ± 0.26	0.17 ± 0.03	53.05 ± 1.06
1308_2	37.68 ± 1.58	0.59 ± 0.06	0.44 ± 0.06	0.06 ± 0.02	0.11 ± 0.04	5.50 ± 0.10	27.87 ± 1.39	689.48 ± 13.16	350.33 ± 5.49	4.36 ± 0.17	1.41 ± 0.03	6.03 ± 0.05	0.25 ± 0.03	177.13 ± 2.08
2308_1	130.86 ± 6.22	0.62 ± 0.06	0.29 ± 0.05	0.09 ± 0.01	0.17 ± 0.03	10.25 ± 0.19	61.10 ± 2.60	395.44 ± 9.42	447.79 ± 15.79	12.97 ± 0.41	1.34 ± 0.07	7.98 ± 0.27	0.30 ± 0.04	41.60 ± 1.07

Sample ID	Al	As	Cd	Co	Cr	Cu	Fe	K	Mg	Mn	Ni	Pb	V	Zn
2308_2	2376.29 ± 23.85	2.18 ± 0.11	0.53 ± 0.05	1.58 ± 0.05	1.48 ± 0.06	23.92 ± 0.54	366.66 ± 10.10	2458.25 ± 56.37	1672.58 ± 22.38	305.44 ± 4.20	4.16 ± 0.12	51.65 ± 1.18	4.57 ± 0.19	247.10 ± 2.62
3308	1192.60 ± 9.13	0.93 ± 0.03	0.54 ± 0.04	0.77 ± 0.03	0.62 ± 0.04	17.62 ± 0.19	315.02 ± 6.46	1264.67 ± 9.50	1031.42 ± 19.73	94.65 ± 0.50	1.94 ± 0.07	27.88 ± 0.14	2.38 ± 0.09	165.57 ± 1.96
4308_1	21.49 ± 1.61	0.41 ± 0.05	0.04 ± 0.01	0.01 ± 0.01	0.07 ± 0.04	4.19 ± 0.07	20.78 ± 1.26	541.66 ± 6.11	279.91 ± 2.67	2.64 ± 0.11	0.12 ± 0.03	0.83 ± 0.00	0.03 ± 0.01	14.18 ± 0.17
4308_2	2099.81 ± 50.50	2.34 ± 0.11	0.71 ± 0.05	1.00 ± 0.03	1.69 ± 0.07	48.12 ± 0.09	797.68 ± 4.72	1495.61 ± 34.06	1702.59 ± 48.88	132.31 ± 0.96	5.27 ± 0.14	60.47 ± 0.35	4.64 ± 0.17	752.65 ± 8.11
1309_1	3299.98 ± 66.08	2.30 ± 0.09	0.43 ± 0.01	1.53 ± 0.07	1.51 ± 0.05	36.35 ± 0.36	597.82 ± 5.60	2552.15 ± 9.90	1971.17 ± 44.96	371.89 ± 5.21	5.29 ± 0.12	68.42 ± 0.43	4.43 ± 0.13	195.38 ± 1.86
1309_2	6262.73 ± 89.29	7.09 ± 0.12	2.66 ± 0.16	6.03 ± 0.10	4.99 ± 0.07	48.39 ± 0.49	3006.76 ± 34.09	6902.40 ± 234.50	6265.91 ± 92.59	953.32 ± 9.83	16.13 ± 0.23	693.28 ± 26.52	13.40 ± 0.67	1970.54 ± 21.43
2309_1	448.11 ± 25.60	1.64 ± 0.03	0.15 ± 0.04	0.27 ± 0.02	0.41 ± 0.05	13.48 ± 0.31	145.69 ± 4.02	1055.96 ± 20.09	605.57 ± 21.48	48.83 ± 0.76	2.37 ± 0.12	10.85 ± 0.38	0.88 ± 0.04	64.25 ± 0.75
2309_2	3690.30 ± 77.38	11.75 ± 0.37	0.83 ± 0.06	5.45 ± 0.08	2.72 ± 0.08	39.20 ± 0.50	738.63 ± 3.01	3665.97 ± 37.60	2790.82 ± 80.22	472.09 ± 3.78	5.90 ± 0.10	77.36 ± 0.51	6.75 ± 0.10	448.73 ± 1.48
3309	2083.41 ± 61.23	1.83 ± 0.03	0.38 ± 0.04	1.15 ± 0.04	0.67 ± 0.07	23.69 ± 0.54	384.36 ± 11.69	2277.60 ± 85.06	1228.28 ± 29.89	186.17 ± 2.93	2.09 ± 0.11	53.40 ± 1.94	3.31 ± 0.26	175.30 ± 3.64
4309	1101.71 ± 19.41	2.96 ± 0.03	0.37 ± 0.04	0.93 ± 0.04	0.93 ± 0.11	20.81 ± 0.27	270.11 ± 1.79	1537.75 ± 60.69	975.90 ± 20.61	78.50 ± 1.05	3.26 ± 0.09	22.48 ± 0.87	2.82 ± 0.17	120.57 ± 1.89
1310_1	2403.79 ± 53.28	1.56 ± 0.08	84.85 ± 3.64	1.73 ± 0.03	2.63 ± 0.06	47.63 ± 1.04	875.60 ± 20.09	3426.77 ± 105.80	2199.65 ± 33.93	225.67 ± 3.64	4.11 ± 0.16	48.86 ± 1.54	5.87 ± 0.48	644.57 ± 10.12
1310_2	2698.24 ± 46.47	1.32 ± 0.08	56.10 ± 2.56	1.92 ± 0.04	2.97 ± 0.08	32.35 ± 0.42	850.44 ± 9.86	3003.06 ± 95.34	2112.46 ± 53.80	202.92 ± 3.30	4.62 ± 0.09	44.68 ± 1.44	4.55 ± 0.24	764.81 ± 7.97

Sample ID	**Al**	**As**	**Cd**	**Co**	**Cr**	**Cu**	**Fe**	**K**	**Mg**	**Mn**	**Ni**	**Pb**	**V**	**Zn**
2310_1	22.66 ± 3.49	0.52 ± 0.04	0.06 ± 0.01	0.14 ± 0.03	0.09 ± 0.02	5.33 ± 0.15	16.63 ± 1.00	567.85 ± 6.52	365.57 ± 8.56	3.36 ± 0.16	0.27 ± 0.03	2.17 ± 0.03	-0.02 ± 0.03	25.23 ± 0.11
2310_2	1318.01 ± 31.92	1.00 ± 0.08	0.84 ± 0.03	0.55 ± 0.02	0.42 ± 0.02	13.96 ± 0.26	318.63 ± 3.17	767.80 ± 9.56	702.87 ± 13.14	74.25 ± 0.48	1.22 ± 0.04	37.16 ± 0.27	1.43 ± 0.07	121.35 ± 0.33
M3310_1	56.98 ± 3.30	0.44 ± 0.03	0.09 ± 0.00	0.16 ± 0.01	0.22 ± 0.03	7.51 ± 0.24	36.70 ± 1.51	358.96 ± 5.71	536.01 ± 12.73	4.41 ± 0.11	3.40 ± 0.11	1.26 ± 0.01	0.13 ± 0.02	28.99 ± 0.39
M2311	534.53 ± 12.17	2.47 ± 0.13	0.11 ± 0.03	1.09 ± 0.02	0.32 ± 0.03	39.11 ± 0.49	228.39 ± 3.21	852.50 ± 22.03	1002.66 ± 32.01	79.27 ± 0.45	1.37 ± 0.10	4.13 ± 0.04	1.60 ± 0.10	75.14 ± 0.61
M3310_2	3572.15 ± 68.54	4.24 ± 0.05	1.29 ± 0.05	7.13 ± 0.10	3.34 ± 0.18	49.17 ± 0.57	1178.49 ± 12.15	7838.62 ± 88.14	6976.88 ± 174.42	650.75 ± 5.56	10.60 ± 0.30	25.11 ± 0.24	15.27 ± 0.29	1100.77 ± 11.31
M3311_1	11979.19 ± 325.59	5.65 ± 0.16	4.25 ± 0.17	28.35 ± 0.43	8.78 ± 0.05	146.19 ± 1.61	2494.58 ± 33.17	15239.59 ± 261.57	17418.27 ± 425.88	2323.36 ± 26.44	50.42 ± 0.51	66.65 ± 0.24	35.07 ± 0.09	7030.91 ± 51.30
M3311_2	2077.95 ± 43.23	2.74 ± 0.08	1.83 ± 0.04	3.47 ± 0.07	2.24 ± 0.04	62.77 ± 0.47	777.98 ± 4.28	2675.04 ± 29.68	2590.51 ± 57.73	303.76 ± 3.37	7.79 ± 0.26	22.00 ± 0.17	10.51 ± 0.30	248.11 ± 1.72

Table S5. Indoor and outdoor $PM_{2.5}$ and PM_{10} summary with I/O Ratios.

Home ID	$PM_{2.5}$ Indoor (μg/m³)	$PM_{2.5}$ Outdoor (μg/m³)	PM_{10} Indoor (μg/m³)	PM_{10} Outdoor (μg/m³)	$PM_{2.5}$ I/O Ratio	PM_{10} I/O Ratio
2310	2.47 ± 0.72	4.12 ± 1.69	8.07 ± 5.31	13.05 ± 3.04	0.60	0.62
1310	4.53 ± 0.44	1.99 ± 0.28	23.37 ± 11.04	14.15 ± 5.71	2.28	1.65
4309	3.67 ± 0.76	9.23 ± 1.54	26.75 ± 15.85	16.39 ± 4.43	0.40	1.63
3309	1.45 ± 0.28	1.85 ± 3.38	9.94 ± 8.75	6.21 ± 5.49	0.78	1.60
2309	3.05 ± 0.58	1.93 ± 0.25	26.15 ± 13.55	7.21 ± 3.52	1.58	3.63
1309	2.54 ± 0.83	2.67 ± 0.32	16.64 ± 9.54	9.35 ± 5.04	0.95	1.78
4308	10.74 ± 1.26	8.58 ± 1.38	75.71 ± 31.65	24.03 ± 5.97	1.25	3.15
3308	3.01 ± 0.51	9.12 ± 2.08	35.12 ± 17.76	17.89 ± 7.14	0.33	1.96
2308	2.24 ± 0.40	3.45 ± 0.62	23.91 ± 22.00	9.73 ± 4.76	0.65	2.46
1308	2.43 ± 0.62	4.53 ± 1.06	15.23 ± 8.23	15.43 ± 7.19	0.54	0.99
3307	1.28 ± 0.37	5.46 ± 45.11	11.33 ± 8.43	12.70 ± 45.70	0.23	0.89
2307-2	6.48 ± 1.08	2.54 ± 0.65	72.88 ± 34.85	9.65 ± 3.25	2.55	7.55
2307-1	5.44 ± 1.05	2.61 ± 0.50	54.70 ± 15.55	10.43 ± 4.12	2.08	5.24
1307	2.43 ± 0.34	7.72 ± 0.79	12.28 ± 6.77	12.67 ± 2.40	0.31	0.97
3306	2.87 ± 0.62	1.14 ± 0.55	82.19 ± 42.63	5.69 ± 4.70	2.51	14.44
2306	1.66 ± 0.26	3.30 ± 0.59	13.51 ± 7.51	20.17 ± 8.27	0.50	0.67
1306	2.37 ± 0.45	3.04 ± 0.46	30.39 ± 17.18	18.27 ± 8.08	0.78	1.66
Average	3.45 ± 2.33	4.31 ± 2.71	31.66 ± 24.45	13.12 ± 5.11	1.08 ± 0.81	2.99 ± 3.46

Table S6. Multiple linear regression results on average indoor $PM_{2.5}$ concentrations.

Variables	**Reference Condition**	**Regression coefficients** [1]	**p-value** [2]
Model intercept	-	5.1	***
Distance from fire zone <100 m	Backyard on fire	-1.2	***
Distance from fire zone >100 m		-0.5	0.38
Professional cleaning	Not cleaned	-0.4	0.63
Vacuum cleaning (Non-HEPA)		6.8	***
Active air purifiers	w/o air purifier	-1.2	0.12
Windows or doors were fully closed during the fire	Windows or doors were opened during the fire	-1.5	***
		Number of samples = 20	**R^2 = 0.73**

[1] Regression coefficients represent the change in indoor $PM_{2.5}$ ($\mu g/m^3$) associated with each condition, relative to the reference category. A negative coefficient indicates a reduction in $PM_{2.5}$ compared to the reference.

[2] Statistically significant p-values (i.e., $p < 0.05$) are indicated as ***.

SUPPLEMENTARY FIGURES

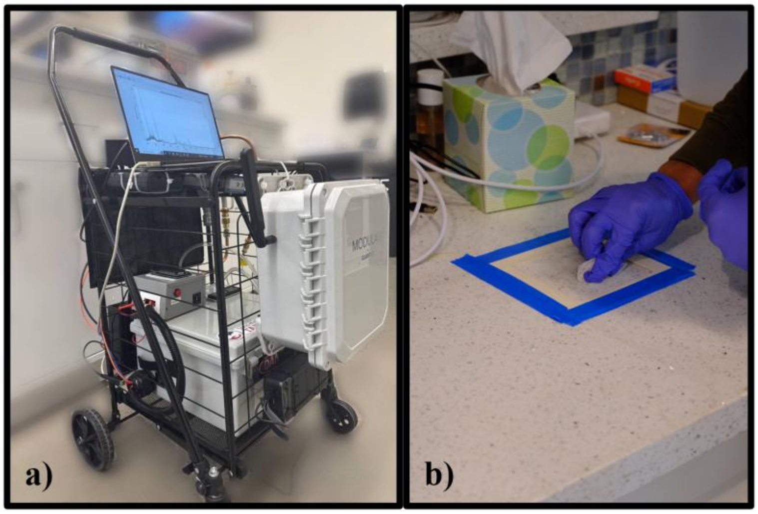

Figure S1. a) Sampling cart containing all the instruments with required batteries and b) Dust sample collection based on the EPA's method for lead dust wipe collection using a 10 × 10 cm^2 sampling template

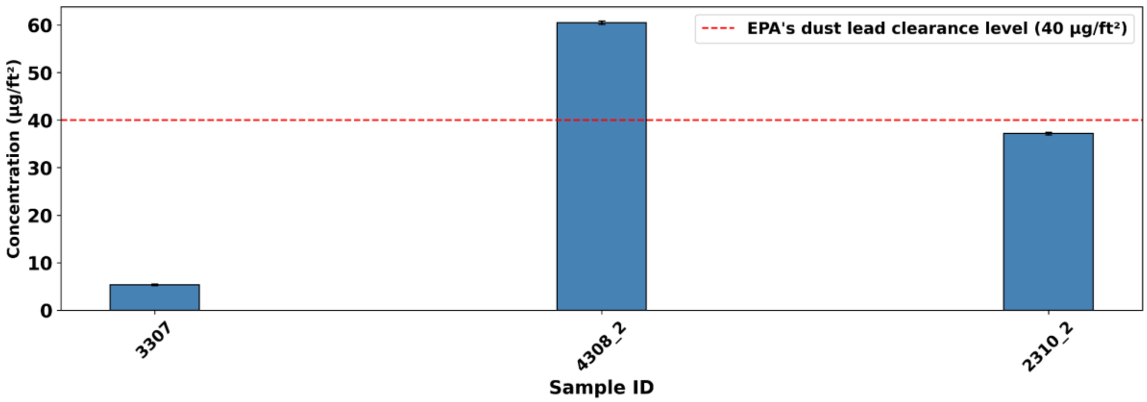

Figure S2. Lead concentrations in wipe samples taken from window sills.

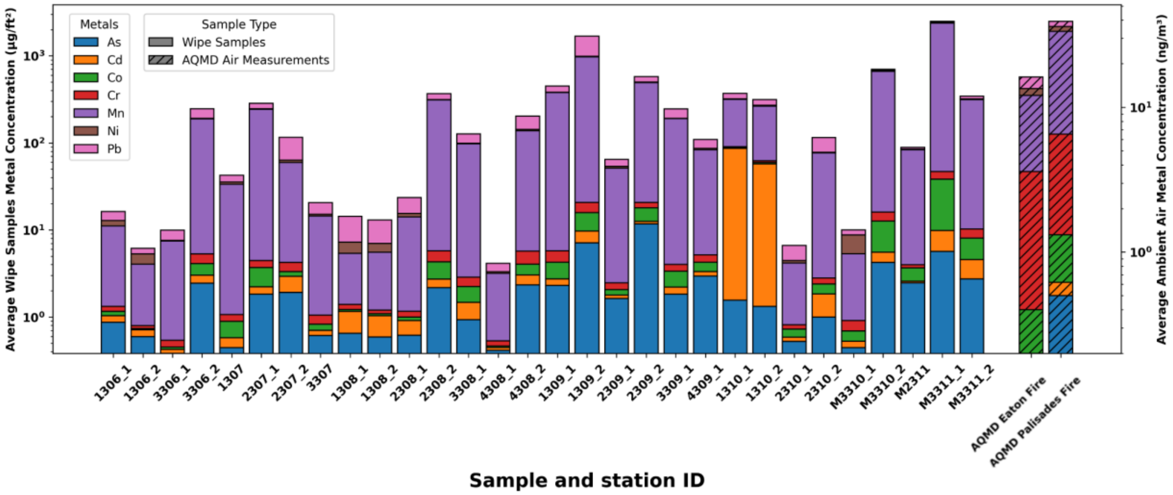

Figure S3. Stacked plot of average HAP metals concentrations in collected wipe samples from homes (left axis) and AQMD average HAP metals concentrations measured in ambient air (right axis). Note that the y-axis is on a log10 scale.