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Post-Fire Resuspension of Metal Toxins at the Wildland–Urban Interface

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ABSTRACT

After wildland–urban interface (WUI) fires, the severe destruction leaves metal-containing materials on the ground. During post-fire recovery, wind, debris removal, vehicle movement, and reconstruction can disturb these residues and release these materials into the air. However, what toxic metals are released into the air from fire-altered materials and what controls their airborne release are only partly understood. Here, we designed and fabricated a windbox to resuspend and collect WUI fire-altered materials under controlled conditions from samples collected after the January 2025 Los Angeles fires. Bulk metal concentrations did not consistently predict the corresponding airborne metal concentrations. Instead, the amount of resuspendable particulate material and its rate of depletion from the air controlled the release of lead (Pb) and chromium (Cr). Vehicle-associated debris produced >50-fold Pb enrichment and >100-fold Cr enrichment in ultrafine particles (< 0.25 μm). Electron microscopy identified Pb-, Cr-, Fe-, Ti-, and Ni-bearing particles < 100 nm that can deposit deep in the lungs, cross the air–blood barrier, and reach other organs, including the brain. Our findings suggest that toxic metal-containing ultrafine particles can be resuspended into the air even at low wind speeds after WUI fires, threatening public health in local communities for extended periods post-fire.

KEYWORDS: Particle resuspension, lead, windbox, airborne nanoparticles, post-fire recovery

INTRODUCTION

After a wildfire has been extinguished, the affected landscape can continue to supply airborne particles whenever wind or recovery activities disturb surface residues.¹⁻⁴ At the wildland–urban interface (WUI), burning leaves mixtures of ash, soil, and damaged structural and vehicle materials that can contain toxic metal-bearing particles.⁵⁻⁹ Wind can remobilize exposed materials, while debris handling, vehicle movement, and reconstruction create further opportunities for their airborne release.^{1, 2} These processes connect a fire to the subsequent recovery period as a combustion–resuspension system: thermal alteration produces a residual metal source, and later aerodynamic or mechanical disturbance transfers part of that source into the air.^{1, 2, 6, 7, 10-12} Understanding this sequence requires examining the materials that remain after burning and their capacity to release particles into the air.

Metal release during resuspension depends on more than the concentration of metal present on the ground.^{13, 14} Laboratory studies^{13, 15} of contaminated soils and mineral dust show that particle generation can selectively transfer metals into airborne fractions, producing compositions that differ from those of the bulk source. For fire debris, this raises two linked questions: how much particulate material can be mobilized, and which metals are carried by the particles that become airborne? A metal-rich residue may release relatively little dust, whereas a smaller pool of readily entrained particles may contribute disproportionately to airborne metal mass.^{13, 15}

Evidence from the 2025 Los Angeles fires demonstrates the relevance of this pathway during recovery.¹⁶ Kleeman et al.² detected airborne hexavalent chromium [Cr (VI)] around debris cleanup zones two months after the fires, with Cr concentrated in particles smaller than 56 nm.² However, these measurements did not identify which burned materials release metals into the air or how much each material contributes. Paired measurements of source materials and the particles they release are needed to determine whether bulk metal concentrations track airborne release and whether particular metals become enriched within specific particle-size fractions.¹⁷

Assessing contamination after WUI fires often involves comparing metal concentrations in materials remaining on the ground with soil screening benchmarks.¹⁸⁻²¹ These benchmarks help identify contamination that warrants further evaluation under defined exposure assumptions.^{19, 20} For Pb, the U.S. EPA uses a residential soil screening level of 200 $\mu\text{g g}^{-1}$ for hazardous waste cleanup soils, while California uses 80 $\mu\text{g g}^{-1}$.^{19, 20} However, fire-altered materials can contain heterogeneous mixtures of ash, soil, and burned debris, and they may release particles to air differently from soil. In addition, bulk metal concentrations alone do not determine how much metal becomes airborne from burned material.¹³ Once particles enter the air, PM_{2.5} mass concentration is commonly used to assess particle pollution,^{22, 23} but this measurement alone does not identify their metal content or the distribution of metals among particle sizes,^{15, 24} which control the deposition and toxicity of particulate material in the respiratory system after inhalation.²⁵⁻²⁸ How bulk metal concentrations in fire-altered materials impact the amount of metal released into the air and its distribution across particle sizes is not currently understood.

Here, we investigate the potential for fire-altered materials to release metal-bearing particles during post-WUI fire resuspension and examine the relationship between source-material and airborne metal composition. We designed and fabricated a windbox with an active air-sampling system to resuspend and characterize particles from selected fire-altered materials collected from properties affected by the January 2025 Los Angeles fires (Palisades and Eaton Fires). The objectives were to examine whether bulk metal concentrations predict airborne release, quantify metal enrichment across aerodynamic size fractions, and evaluate how particle generation and depletion relate to metal release. We also used scanning transmission electron microscopy to characterize the size, morphology, and elemental associations of the resuspended particles. Together, these approaches aim to clarify how materials remaining after WUI fires can contribute to airborne metal exposure during recovery.

Materials and Methods

Fire-altered Material Collection. Seven fire-altered material samples were collected from four Eaton Fire-impacted houses and three Palisades Fire-impacted houses. All selected structures were completely

destroyed, leaving a mixture of burned and unburned materials exposed to air for several months. These exposed materials constitute potential sources of airborne particles during wind-driven resuspension or disturbance associated with recovery activities. Samples were deliberately collected from discrete sources expected to contain elevated concentrations of Pb and other metals, including burned washing machines and vehicles. This targeted sampling strategy was intended to investigate particle resuspension and metal release from potentially highly contaminated materials under controlled windbox conditions. The study therefore focused on seven selected point-source materials to examine release behavior relevant to potentially severe, localized contamination scenarios. The samples were not intended to represent the composition or relative abundance of debris across the affected neighborhoods. The general characteristics of the samples and field photographs are provided in Table S1 and Figure S1, respectively.

Resuspension Experiments. The resuspension experiments were conducted using a custom-built windbox system designed to simulate mechanical disturbance and wind-driven particle entrainment under controlled conditions. The windbox consisted of an enclosed acrylic chamber ($30.5 \times 25.4 \times 20.3$ cm; ~ 15.7 L internal volume), equipped with multiple airflow sources to generate turbulent and multidirectional flow, allowing for reproducible resuspension of particulates from solid substrates. Airflow within the chamber was produced using four calibrated blowers (GDSTIME 75×30 mm) positioned at the corners and TriPole Mini Handheld Fan mounted along the chamber walls to promote mixing and minimize spatial heterogeneity. The blowers were calibrated to the desired wind speed using a handheld digital anemometer (AcuRite 00256M) before each experiment. This configuration creates a well-mixed environment that facilitates consistent particle suspension while avoiding strong directional bias. The chamber was used as a closed system, but no attempt was made to make it fully airtight. As a result, small, non-visible gaps inherent to the box construction allowed limited air exchange, preventing vacuum formation while remaining sufficiently closed to avoid particle loss from the chamber. Pressure differences between the interior and exterior were monitored using a manometer (ARN05M) and remained zero throughout the experiments. All experiments were conducted under a fume hood to ensure that incoming air was free of background particulate contamination. The schematic of the box is shown in Fig. S2. For each experiment, 15 g of

material was evenly distributed across a designated area at the center of the chamber to obtain a thin layer of the samples. Airborne particles were collected using a Flite 4 Area Sample Pump (SKC) coupled with a Sioutas Cascade Impactor (SKC), which segregates particles into aerodynamic diameter fractions of >2.5 μm , $2.5\text{--}1.0$ μm , $1.0\text{--}0.50$ μm , $0.50\text{--}0.25$ μm , and <0.25 μm . The size-separation efficiency of the impactors has been previously characterized and is approximately 90% within the target ranges.²⁹ Three sets of impactors were used for each experiment. In parallel, real-time $\text{PM}_{2.5}$ concentrations were continuously monitored within the chamber using an indoor air quality monitor (Hamos) to track the temporal evolution of the particle suspension during each experiment. Resuspension was initiated by activating the blowers, fans, $\text{PM}_{2.5}$ monitor, and pumps, which entrain particles into the air, maintain them in suspension, and collect them on the filters. A wind speed of 16 km h^{-1} was used to represent a gentle breeze ($13\text{--}19\text{ km h}^{-1}$), which commonly occurs under ambient conditions. Each experiment consisted of repeated resuspension cycles ($n = 5$) using the same sample, with airflow applied until $\text{PM}_{2.5}$ concentrations decayed to near-background levels ($35\text{ }\mu\text{g m}^{-3}$), at which point the cycle was terminated. After each cycle, the remaining material was swept back to the center of the chamber, and the resuspension process was repeated. Following completion of the five cycles, the remaining sample was removed, and the same fresh material was introduced. The experiment was then repeated using a new set of impactors to obtain 6 replicates in total per sample for subsequent analyses. After each experiment, the chamber was fully rinsed with soap and cleaned for the next sample.

Chemical Analysis. The resuspendable particulates and their source materials were digested using a microwave-assisted acid digestion with a mixture of nitric acid (HNO_3 , 67–70%) and hydrochloric acid (HCl , 36–38%). The samples were then measured using an inductively coupled plasma mass spectrometer (ICP-MS).

Electron Microscopy. The morphology, size, and elemental distribution of metal-bearing particles were characterized using scanning transmission electron microscopy (STEM). TEM grids (Ted Pella) were prepared by gently rubbing the grid surface against the particle-loaded filter to transfer collected material directly onto the grid. The grids were then lightly tapped to remove loosely attached particles before

imaging. Imaging was performed in high-angle annular dark-field (HAADF) mode using a Thermo Fisher Spectra 300 monochromated, double-aberration-corrected STEM. Elemental distributions were obtained via energy-dispersive X-ray spectroscopy (EDS) using Super-X detectors. Image brightness and contrast were adjusted linearly across each micrograph to increase the signal-to-noise ratio of the mapped pixels.

RESULTS AND DISCUSSION

Toxic Metals in Resuspended Particulates

Metals were measured across all fire-altered materials. Iron was the most abundant metal, with concentrations ranging from 5089.0 ± 901 to $15115 \pm 1260 \mu\text{g g}^{-1}$ (Fig. S3). Lead was also present at high concentrations, ranging from 332.25 ± 22.1 to $14599 \pm 849 \mu\text{g g}^{-1}$ across the samples. Other metals consistently observed at high concentrations included Cr, Cu, Ti, Zn, Mn, and Ni.

These samples were resuspended in the windbox at speeds consistent with a gentle breeze (16 km h^{-1}). The potential for resuspension of different fire-altered materials varied substantially, ranging from $0.56 \pm 0.06 \text{ mg}$ (sample PFB) to $10.2 \pm 0.7 \text{ mg}$ (sample EFC) (Table S1). The ultrafine particulates ($<0.25 \mu\text{m}$) ranged from only 3.30% (sample PFA) to 34.8% (sample EFD). On average, the generated resuspended material consisted of $45.5 \pm 4.7\%$ coarse ($>2.5 \mu\text{m}$), $39.2 \pm 2.3\%$ fine ($0.25\text{--}2.5 \mu\text{m}$), and $15.3 \pm 4.4\%$ ultrafine particulates (Fig. S4).

Airborne metal concentrations also varied substantially in the resuspended particulates. The absolute concentration of total Pb ranged from $0.666 \pm 0.043 \mu\text{g m}^{-3}$ for sample EFB to $51.04 \pm 6.0 \mu\text{g m}^{-3}$ for sample PFB (Fig. 1). Chromium showed less variability, ranging from $0.261 \pm 0.043 \mu\text{g m}^{-3}$ for sample EFB to $1.99 \pm 0.17 \mu\text{g m}^{-3}$ for sample PFC. Similar sample-to-sample variability was observed for the other metals such as Fe, Zn, Cu, Ni, Mn, and Ti across aerodynamic diameter fractions, with substantial concentrations extending into the fine and ultrafine particles.

To account for differences in the quantity of resuspended particulates, we also evaluated mass-normalized metal concentrations. The distribution of mass-normalized concentrations showed higher median concentrations of Cr, Pb, Cu, and Fe in the ultrafine fraction than in the larger particle diameter fractions (Fig. S5). On average, Fe and Zn were the most abundant metals in the resuspended particulates,

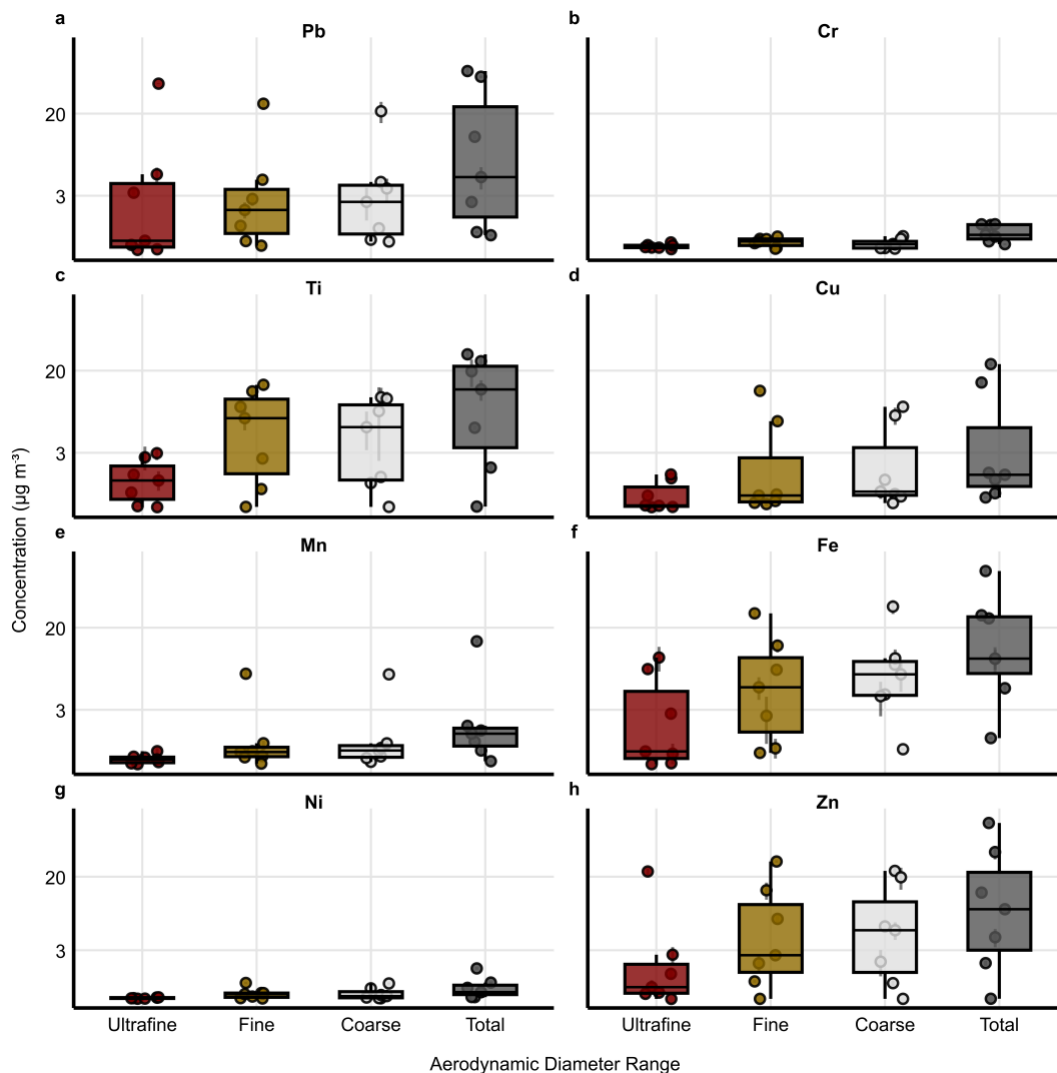
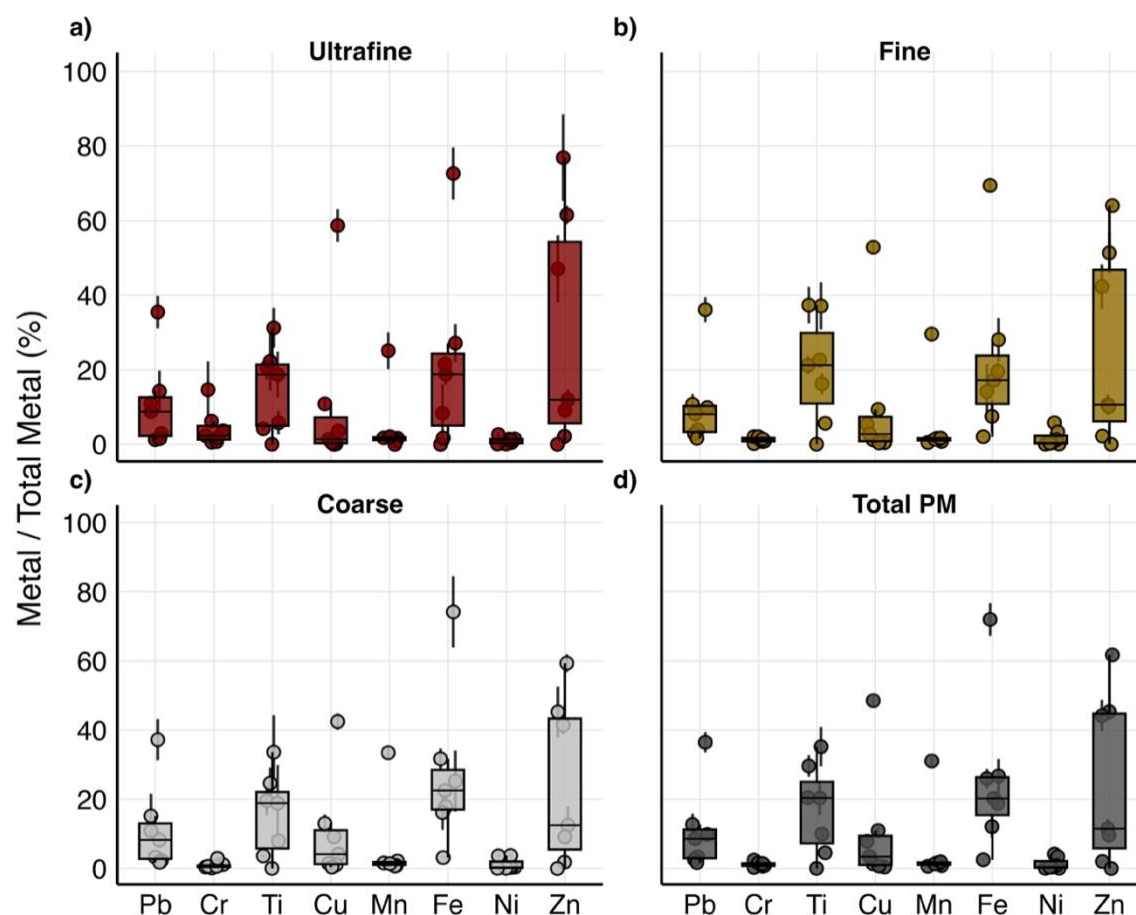


Figure 1. Concentrations of metals in air across aerodynamic diameter fractions generated from the resuspension of burned source materials. Ultrafine, fine, and coarse fractions correspond to <0.25 , 0.25 – 2.5 , and >2.5 μm , respectively; Total sums the five collected impactor fractions ($n = 7$).

each accounting for approximately 25% of the total airborne metal mass. Titanium contributed approximately 17%, followed by Pb (11%), Cu (10%), Mn (6%), and Cr and Ni (1% each) (Fig. 2). The remaining metals each accounted for less than 3% of the total airborne metal mass. These relative abundances were generally consistent across aerodynamic diameter fractions, except for Cr, which showed preferential enrichment in the ultrafine fraction. In this fraction, Cr accounted for approximately 4% of the total metal mass, compared with $\sim 1\%$ of the total resuspended mass.



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185 **Figure 2.** Relative contribution of each metal to the total measured metal mass in resuspended dust across
 186 aerodynamic diameter fractions. Percentages were calculated by dividing the mass of each metal by the
 187 total mass of all analyzed metals within the same aerodynamic diameter fraction.

188 We examined the impact of fire-altered materials on the abundance of airborne metals using both
 189 parametric and non-parametric approaches. Most source–air relationships were weak and nonsignificant
 190 (Figs. S6 and S7). These results indicate that the amount of metal present in the fire-altered materials alone
 191 did not consistently predict the amount released into the airborne fraction. The absence of consistent
 192 source–air relationships suggests that additional physical properties may control post-fire airborne metal
 193 release. We therefore developed an absolute-mass model incorporating the collected resuspendable PM
 194 mass and the PM_{2.5} decay constant, which represent the size of the resuspendable particle pool and its

depletion from the airborne phase, respectively (Text S1 and Figs. S8 and S9). These parameters explained 81% of the variation in total collected Pb mass ($R^2 = 0.81$, $p = 0.04$) (Fig. 3a). The corresponding Cr model explained 87% of the variation ($p = 0.02$) and was also statistically significant (Fig. 3b). These results suggest that airborne Pb and Cr release may be related more to the mass of resuspendable particulates in the source than to their bulk concentrations in fire-altered materials.

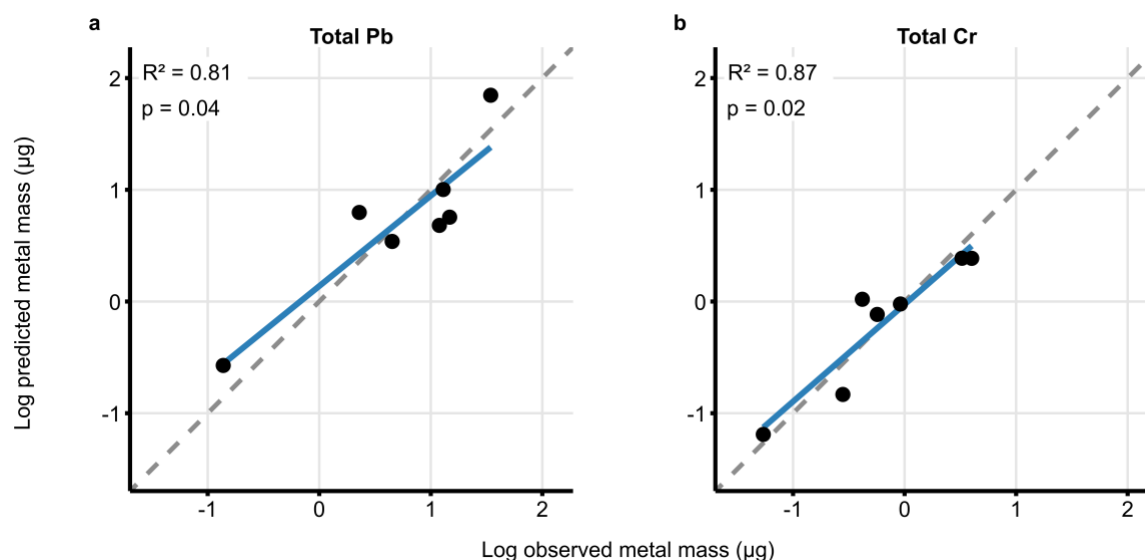


Figure 3. Predicted versus observed airborne mass for (a) Pb and (b) Cr. Log-transformed observed collected metal mass is compared with values predicted by the multivariate model incorporating total PM mass and the slow-phase $PM_{2.5}$ decay constant, K_{slow} (see Text S1). The blue line is the fitted regression line of predicted vs observed values. The dashed line represents the 1:1 relationship between observed and predicted values.

Toxic Metal Enrichment in Air

We calculated metal enrichment factors by comparing metal concentrations in resuspended particulates with their corresponding concentrations in fire-altered materials. Metal enrichment varied substantially among both samples and elements, with some fire-altered materials exhibiting strong enrichment for specific metals while others showed little or no enrichment (Fig. 4 and Table S2). Despite

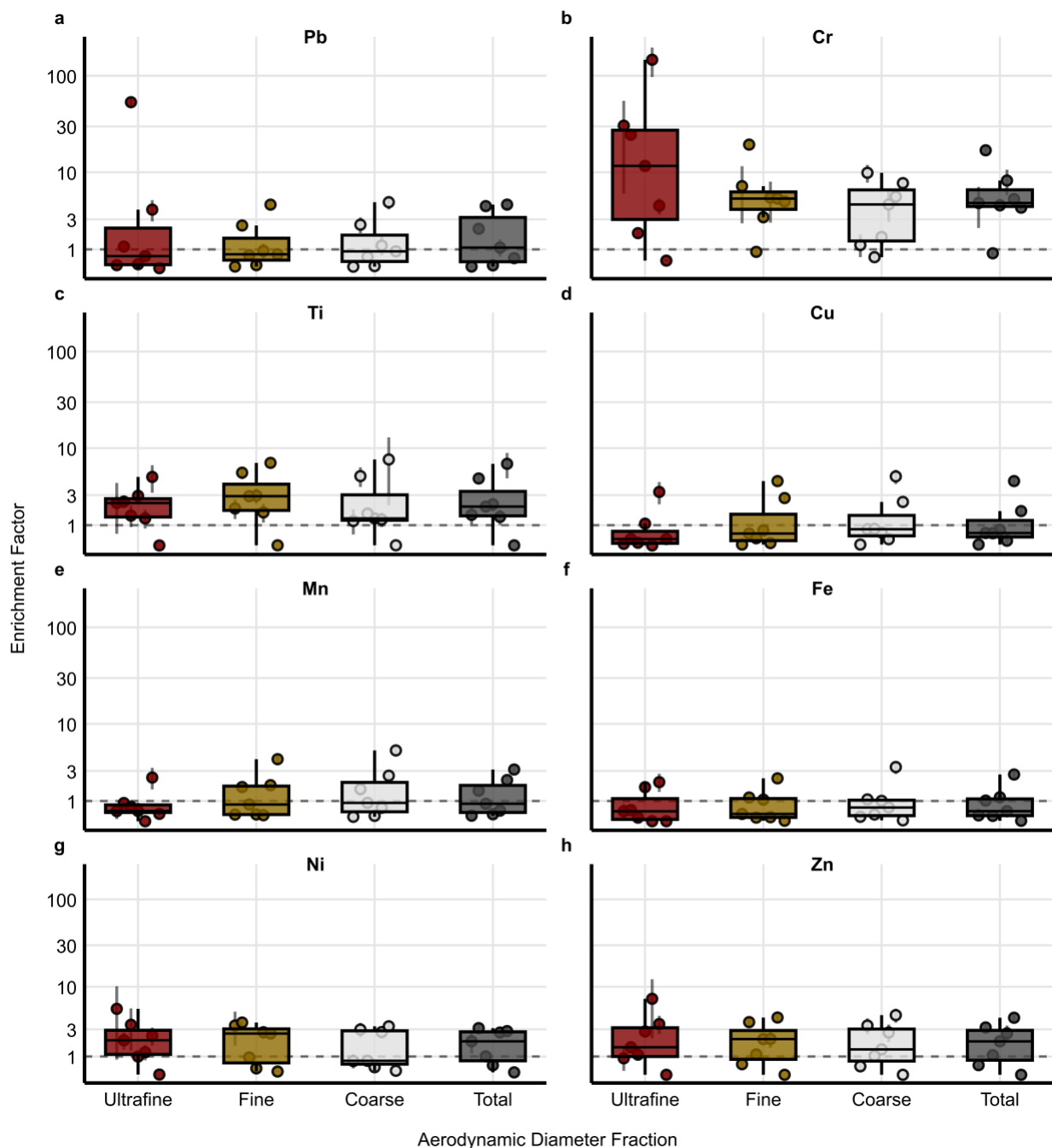


Figure 4. Metal enrichment factors in resuspended dust generated from fire-altered materials across aerodynamic diameter fractions. Enrichment factors were calculated by dividing the mass-normalized metal concentration in resuspended particulates by the corresponding metal concentration in the fire-altered material. The dashed horizontal line indicates an enrichment factor of 1, where values above 1 indicate preferential enrichment in the airborne fraction.

this variability, the median enrichment factors exceeded 1 for Cr, Ti, and Zn across all fractions, and for Ni in the total, fine, and ultrafine fractions but not the coarse fraction. Sample Eaton C, a melted planter ash, was the only sample enriched in all analyzed metals (Table S1 and S2). This sample also generated the greatest mass of resuspended particles (10.2 ± 0.7 mg). In contrast, vehicle-associated materials, including samples PFB, EFB, and PFC, generated lower total resuspendable particulate mass but exhibited substantially greater enrichment of metals. Sample PFB showed strong enrichment of Cr, Ti, Zn, Ni, and Pb, and sample PFC showed pronounced ultrafine Pb enrichment (>50).

Among the analyzed metals, Cr exhibited the greatest enrichment across samples, with an enrichment factor exceeding 100 in the ultrafine fraction of the sample EFB, which represents a residue from a burned Tesla car. Chromium in common vehicle-associated forms, including alloys, is comparatively refractory and cannot volatilize extensively under typical WUI fire conditions.^{5, 30} Instead, thermal stress, oxidation, and structural degradation during burning may fragment or embrittle Cr-bearing materials while leaving a substantial fraction of the Cr in residual materials.^{31, 32} Chromium-bearing material in ash, debris, and surface dust can subsequently become airborne through wind-driven resuspension and activities such as cleanup, debris removal, and reconstruction, as reported by Kleeman et al.²

Here, we used selected samples with potentially high metal concentrations in resuspension experiments to evaluate the potential airborne mobility of metals. Currently, the contributions of different urban materials across the WUI system to airborne Cr and other metals during resuspension remain unclear. Further research, including more field sampling after WUI fires and resuspension experiments using a broader and more representative range of potential sources collected across a larger spatial area and combined with field data, is needed to determine the contributions of different urban sources to airborne metals and their impacts on air quality after WUI fires.

Toxic Metal Nanoparticles in Resuspended Particulates

To characterize the size, morphology, and elemental associations of particles in resuspended particulates, STEM was performed on the fine and ultrafine fractions (Fig. 5). Lead-, Fe-, Cr-, Ti-, and Ni-bearing aggregates contained particles smaller than 100 nm, with particles as small as approximately 30 nm

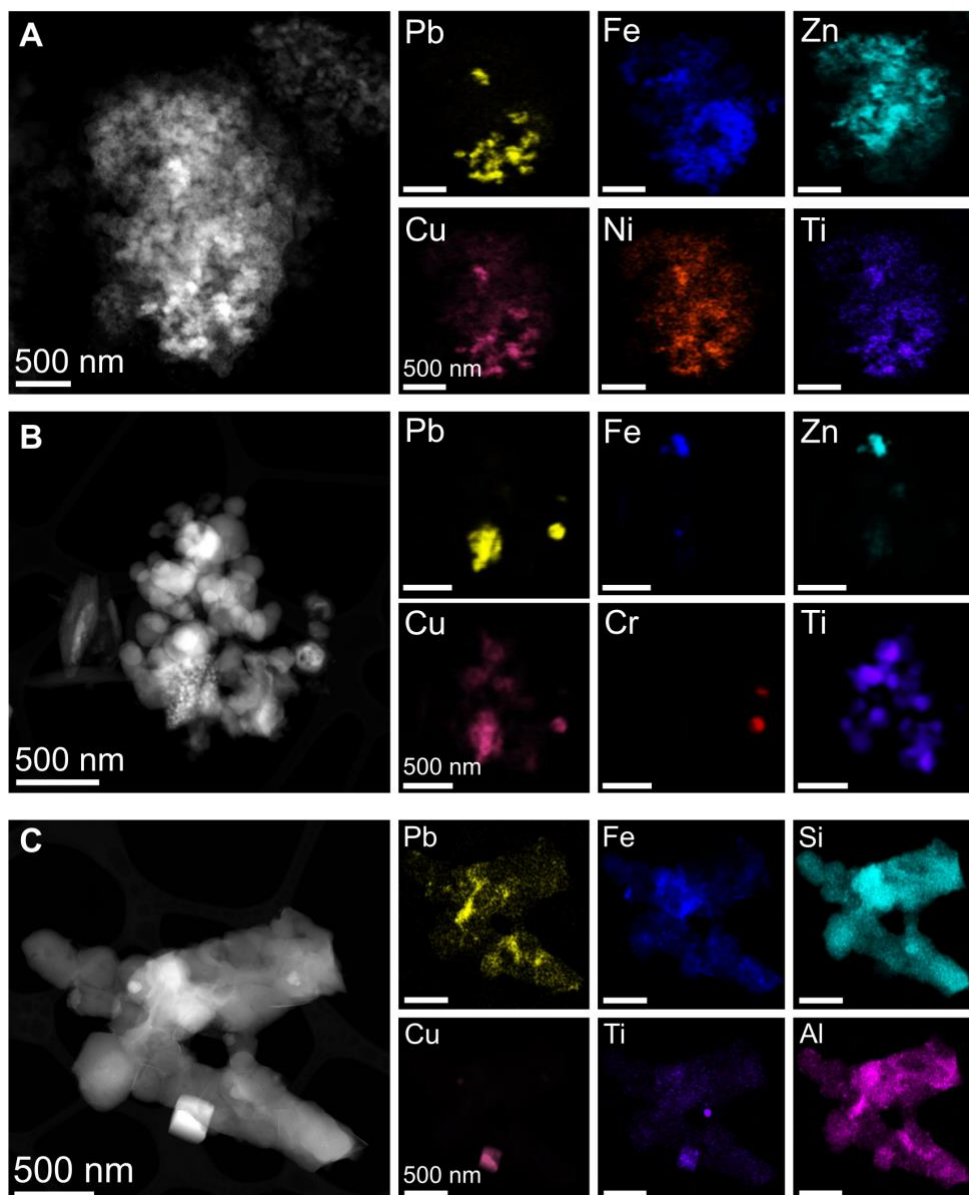


Figure 5. Scanning transmission electron microscopy (STEM) images and corresponding elemental maps. A) Irregular multi-metal particles containing Pb, Cu, Fe, Ni, Zn, and Ti in resuspendable particulates from sample EFA. B) Spherical and irregular Pb-bearing nanoparticle aggregates associated with Cu, Ti, Zn, Fe, and Cr in resuspendable particulates from sample EFC. C) Irregular Pb-bearing particle aggregate associated with Si, Al, Fe, Cu, and Ti in resuspendable particulates from PFB. Elemental maps show that Pb is heterogeneously distributed within the particle, while Si and Al are broadly co-located across the aggregate, consistent with a mineral- or ash-derived particle.

observed within Pb-rich aggregates. These observations indicate that nanoscale metal-bearing constituents can occur within larger aggregates recovered from the resuspended particulates. The nanoscale Pb-associated particles observed in the resuspended particulates are particularly concerning, as previous studies have shown that Pb-bearing nanoparticles may enter the bloodstream or even reach the brain through inhalation.³³

The elemental associations differed among the examined samples. Irregular aggregates from EFA contained Pb together with Cu, Fe, Ni, Zn, and Ti (Fig. 5A). In EFC, spherical and rounded particles occurred alongside irregular structures, with Pb-bearing regions associated with Cu, Ti, Zn, Fe, and Cr (Fig. 5B). The PFB aggregate contained Pb associated with Si, Al, Fe, Cu, and Ti (Fig. 5C). Within this aggregate, Pb was distributed heterogeneously, whereas Si and Al showed broadly overlapping distributions, consistent with association with a mineral-derived matrix. These observations indicate that the airborne metals occurred in chemically heterogeneous particle assemblages. The co-location of Pb with Cu, Zn, Fe, and Ni in aggregates indicates potential exposure to metal mixtures that could produce greater toxicity than Pb alone, depending on their composition and interactions.²⁶

The coexistence of rounded particles, angular fragments, and aggregates suggests that several processes may have contributed to the material available for resuspension. Spherical nanoparticles are consistent with gas-to-particle formation pathways involving high-temperature volatilization followed by nucleation, condensation, and coagulation by cooling,^{34, 35} whereas faceted and irregular particles more likely reflect entrained ash, mineral debris, or structural/vehicle-derived materials.^{5, 36, 37} Previous characterization of WUI fire ashes demonstrates that these residues contain diverse metal-bearing nanomaterials and identifies condensation, oxidation, and reduction as possible formation processes.⁵ This morphological diversity may also influence how the resuspended particulates interact with lung cells and release metals, potentially producing different oxidative-stress responses.³⁸

Implications for Post-WUI fire Air Quality and Exposure Risks

Air-quality index (AQI) and Soil-screening benchmarks provide essential first-order information about post-fire air quality contamination.^{19, 20, 39, 40} Considering airborne metal composition, particle-size

distribution, and resuspension potential alongside these benchmarks can provide a more complete assessment of inhalation exposure during active burning and post-fire recovery.

The resuspension experiments provide a corresponding comparison for post-fire materials. To determine how bulk Pb concentration, particle generation, and airborne Pb enrichment interact, we developed:

$$C_{\text{bulk}} = C_{\text{air}} / (M_{\text{RP}} \times E_{\text{Pb}}),$$

where C_{bulk} is the Pb concentration in the source material ($\mu\text{g g}^{-1}$), C_{air} is the airborne Pb reference concentration ($0.15 \mu\text{g m}^{-3}$), M_{RP} is the resuspendable particulate mass concentration (g m^{-3}), and E_{Pb} is the Pb enrichment factor in resuspendable particulates. Using the EPA Pb air standard of $0.15 \mu\text{g m}^{-3}$ and the measured Pb enrichment factor range of 0.15–4.5 in this study for total Pb, across the range of particle generation observed in our study, the source Pb concentration needed to reach the EPA air threshold ranges from ~ 33 – $1000 \mu\text{g g}^{-1}$ under the lowest particle generation (0.98 mg m^{-3}) in EFB to ~ 5.8 – $174 \mu\text{g g}^{-1}$ under the highest particle generation (5.8 mg m^{-3}) in PFC. Thus, fire-altered materials with bulk Pb concentrations below the U.S. EPA ($200 \mu\text{g g}^{-1}$) and California ($80 \mu\text{g g}^{-1}$) residential soil screening benchmarks^{19,20} could still generate airborne Pb concentrations reaching this reference level during resuspension. This comparison requires careful interpretation, though. The U.S. EPA ambient-air Pb standard of $0.15 \mu\text{g m}^{-3}$ is based on a rolling three-month average; therefore, the exposure time must also be considered. In addition, the residential benchmarks apply to soil, whereas the materials examined here included ash and debris. The comparison therefore illustrates potential resuspension scenarios for the selected materials and does not provide a complete picture of airborne Pb across areas affected by the Los Angeles fires. The results suggest that materials remaining after a fire can release Pb into the air during resuspension, and that bulk soil Pb concentrations alone may not reflect this release potential. Materials with similar bulk Pb concentrations can generate different airborne concentrations depending on their resuspendable particulate content, physical properties, and metal enrichment. These observations suggest the need for future research to better understand how post-fire resuspension contributes to airborne metal exposure.

The windbox experiments demonstrated that fire-altered materials could release metal-bearing fine and ultrafine particles even under air conditions consistent with a gentle breeze. These findings are particularly relevant during cleanup and reconstruction following WUI fires, when disturbance of fire-altered materials may create continuing exposure for residents, workers, and rebuilding communities. The small sizes and metal content of these particles make their release significantly relevant to exposure assessments during cleanup and reconstruction. Field measurements during these activities and broader resuspension experiments could help determine the extent and duration of exposure among residents and workers following WUI fires.

DATA AVAILABILITY

The data supporting this study are available in the article and its Supporting Information.

SUPPORTING INFORMATION

Text S1. Airborne Pb and Cr emission model.

Table S1. General characteristics of the burned source materials and their resuspendable particle content.

Table S2. Metal enrichment across different source materials and aerodynamic diameter fractions.

Figure S1. Collection of burned source materials from fire-impacted structures and vehicles following the 2025 Los Angeles fires. EFD is not shown.

Figure S2. Windbox setup used to generate and collect resuspended dust from burned source materials.

Figure S3. Metal concentrations in source materials collected from fire-impacted properties affected by the Palisades Fire (Palisades A–C) and Eaton Fire (Eaton A–D). Error bars represent standard errors ($n = 3$).

Figure S4. Percentage of airborne PM generated from the resuspension of source materials across different size bins.

Figure S5. Mass-normalized concentrations of metals in resuspended particles generated from burned source materials across aerodynamic diameter fractions.

Figure S6. Relationships between metal concentrations in burned source materials and mass-normalized metal concentrations in airborne resuspended dust.

Figure S7. Non-parametric relationships between metal concentrations in burned source materials and mass-normalized metal concentrations in airborne resuspended dust.

Figure S8. Temporal variation in PM_{2.5} concentrations during windbox experiments. Open circles indicate concentrations reported above the instrument measurement limit ($>999 \mu\text{g m}^{-3}$).

Figure S9. Estimation of the slow-phase PM_{2.5} depletion rate during the experiments.

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NOTES

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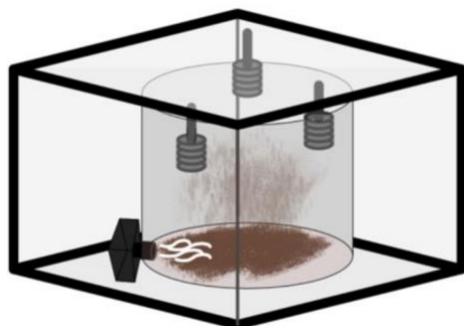
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