

# Determination of Soil Contamination at the Wildland-Urban Interface after the 2021 Marshall Fire in Colorado, USA

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**ABSTRACT:** Wildfires at the wildland-urban interface (WUI) are increasingly common. The impacts of such events are likely distinct from those that occur strictly in wildland areas, as we would expect an elevated likelihood of soil contamination due to the combustion of anthropogenic materials. We evaluated the impacts of a wildfire at the WUI on soil contamination, sampling soils from residential and nonresidential areas located inside and outside the perimeter of the 2021 Marshall Fire in Colorado, USA. We found that fire-affected residential properties had elevated concentrations of some heavy metals (including Zn, Cu, Cr, and Pb), but the concentrations were still below levels of likely concern, and we observed no corresponding increases in concentrations of polycyclic aromatic hydrocarbons (PAHs). The postfire increases in metal concentrations were not generally observed in the nonresidential soils, highlighting the importance of combustion of anthropogenic materials for potential soil contamination from wildfires at the WUI. While soil contamination from the 2021 Marshall Fire was lower than expected, and likely below the threshold of concern for human health, our study highlights some of the challenges that need to be considered when assessing soil contamination after such fires.



**KEYWORDS:** soil contamination, wildfire, 2021 Marshall Fire, wildland-urban interface, PAH, heavy metals, mercury

## INTRODUCTION

Wildfire risk at the wildland-urban interface (WUI) is expected to increase in the future due to expansion of the WUI and increased wildfire activity.<sup>1,2</sup> The WUI, defined as the area where houses meet or intermingle with undeveloped wildland vegetation,<sup>3</sup> is expanding in many regions across the globe. This expansion is particularly evident in the United States, where home construction from 1990 to 2010 increased the WUI area from 581,000 km<sup>2</sup> to 770,000 km<sup>2</sup> and increased the number of houses at the WUI from 30.8 million to 43.4 million.<sup>4</sup> The frequency and size of WUI wildfires are expected to increase as many wildland-urban regions become hotter, drier, and experience more frequent drought events due to climate change.<sup>2,5–7</sup> The increased risk of catastrophic WUI wildfire events is exacerbated by the elevated potential for human-driven ignition, barriers to fire prevention and suppression,<sup>8</sup> and the high flammability of structures.<sup>2,9</sup> Compared to wildland fires, WUI fires can be particularly damaging to public health and costly for local economies, with potentially larger societal repercussions. Unfortunately, there are numerous global examples of the devastation wrought by WUI wildfires, including multiple fires in Greece and Portugal in 2023, the 2018 Camp Fire that destroyed Paradise, California, and the 2023 Lāhaina fire in Maui, one of the deadliest wildfires in US history.

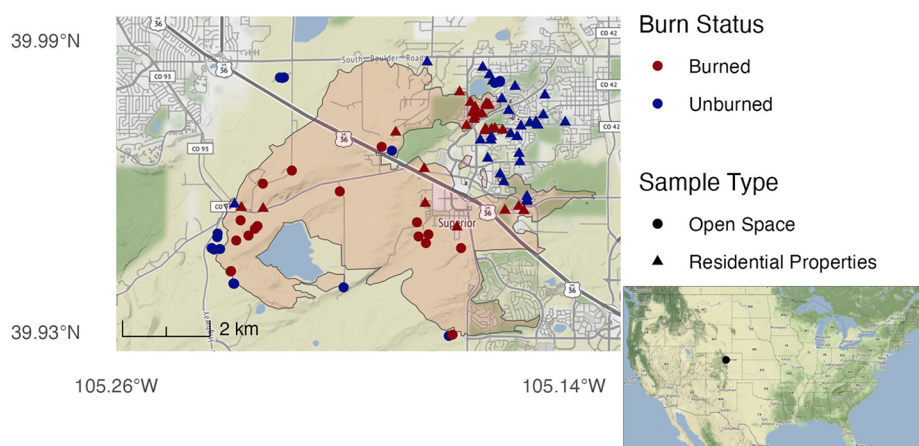
The impacts of wildland fires on air, water, and soil quality in nonresidential areas have been relatively well-studied.<sup>10–13</sup> In

contrast, the environmental impacts of wildfires that occur at the WUI have received far less attention, a point highlighted in a recent U.S. National Academies report.<sup>14</sup> Addressing this knowledge gap is crucial because the impacts of WUI wildfires on soil quality are likely distinct from those of wildland fires, including a greater potential for soil contamination. Whereas wildland fires are typically fueled solely by vegetation combustion, WUI fires can also be fueled by the partial or complete combustion of structures, vehicles, and other components of urban infrastructure. These anthropogenic fuels often contain materials with high concentrations of toxic compounds. These compounds include toxic metals and polycyclic aromatic hydrocarbons (PAHs) that can be released into soil after a fire event.<sup>15–21</sup> In fact, studies that have investigated soil contamination after fires fueled entirely, or in part, by structures have often found evidence for significant postfire soil contamination.<sup>16,22–25</sup> This potential for postwildfire soil contamination could have long-term impacts on the health of people, animals, and plants living in or near burned areas of the WUI.<sup>14</sup>

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**Figure 1.** Map of the sampling area. Residential properties (triangles) and nonresidential grassland (circles) sampling locations are plotted with the outline of the Marshall Fire (shaded in light orange). Burn status of sampling locations indicated with red (burned) and blue (unburned) symbols.

The primary objective of this study was to determine the existence and extent of soil contamination following the 2021 Marshall Fire in Colorado, USA, a particularly catastrophic example of the types of wildfire events that can occur at the WUI. The fire started on December 30, 2021, and quickly spread into suburban neighborhoods due to unseasonably warm, dry, and exceptionally windy conditions. This resulted in the most destructive wildfire in terms of structure loss in Colorado history with over 1000 homes being destroyed across a ~2500 ha area. In fact, more homes were destroyed in the Marshall Fire than in the two largest wildfires in recorded Colorado history, both of which were ~80,000 ha in size.<sup>7</sup>

We conducted an extensive and systematic sampling of soils from burned and unburned residential properties that were within or adjacent to the perimeter of the 2021 Marshall Fire. In addition, we collected soils from nonresidential sites (grasslands) inside and outside of the burn perimeter to provide samples representative of a wildland wildfire. We hypothesized that, given the presence of anthropogenic materials in residential areas, burned residential soils would have significantly higher concentrations of metals and PAHs than unburned residential and nonresidential soils. Ultimately, we sought to use this study of the 2021 Marshall Fire to better assess how wildfires at the WUI may uniquely impact soil quality, building the knowledge base needed to safeguard the reestablishment and repopulation of impacted and adjacent residential areas after WUI wildfire events.

## MATERIALS AND METHODS

**Study Area.** The Marshall Fire began on 30 December 2021 in the semiarid Colorado Front Range urban corridor situated in the foothills east of the Rocky Mountains at an elevation of ~1650 m (Figure 1). The climate in the region is characterized by mean monthly temperatures typically ranging from 1 to 23 °C (NOAA 2023a) and mean monthly precipitation amounts between 20 and 80 mm (NOAA 2023b). The vegetation in the study area is mainly grasslands interspersed with patches of Ponderosa pine (*Pinus ponderosa*) forests. This land cover is mostly intact at the western to central part of the study area, but an urban landscape dominates the eastern area. Prior to the wildfire event, conditions were unusually dry and warm for that time period, and the fire moved rapidly from west to east with wind speeds often exceeding 160 km h<sup>-1</sup>. The fire was contained on 31

December 2021 due to fire-fighting activities, a decrease in wind speeds, and a snowfall event that occurred. For additional details on the fire event and associated conditions, see Fovell et al.<sup>26</sup> and the NOAA Boulder Web site<sup>27</sup> which also includes images and videos taken from the area during and after the fire event.

**Site Selection.** In March 2022, we developed a survey to identify homeowners within the wildfire-affected area who were willing to participate in this project. The survey was distributed by the Cooperative Institute for Research in Environmental Sciences (CIRES, University of Colorado Boulder) and received nearly 300 responses. We selected 58 properties within or adjacent to the fire perimeter for sampling (Figure 1), 29 properties that were completely burned and 29 properties that were completely unburned (noting that these “unburned” properties may have received some wind-deposited ash during the fire event, but were outside the fire perimeter and were not directly damaged by the fire). We selected properties that minimized geographic distances and other potentially confounding landscape characteristics (including topography and vegetation cover). We also did not select properties where there was evidence of substantial postfire debris removal or surface soil scraping. In addition to sampling from these residential locations, we also selected burned and unburned locations distributed across the Marshall Fire footprint in neighboring, undeveloped (nonresidential) grasslands for comparable sampling of wildland conditions (Figure 1).

**Field Sampling.** Sampling sites were visited over a three-week period in April 2022, four months after the Marshall Fire with multiple rain and snow events occurring during this period between the fire event and the soil sampling. Soil samples were collected from four different locations at each of the 59 residential properties selected for this study. Specifically, lawns and landscaped areas were sampled at these properties, avoiding any structural debris, nonsoil surfaces, or areas where any alteration to the soil surface (including mulching or scraping) was evident. For each soil sample, rocks and loose plant material were scraped off the soil surface, and eight soil cores (0–5 cm depth) were collected from four separate ~1 m<sup>2</sup> areas per property to obtain sufficient material for chemical analyses (total of 32 soil cores from each property). These sampling areas were selected without considering whether they were on the windward or leeward side of the main structure

during the fire event as this was often difficult to determine and many properties did not have soil on all sides of the main structure. In nearly all cases, the sampling areas were within 20 m of the main structure on each property. The eight soil cores collected within each individual  $\sim 1$  m<sup>2</sup> area were combined, yielding four composited soil samples per residential property. Each composited sample was sieved to  $\leq 2$  mm to remove rocks and larger debris and then stored at 4 °C until further analysis. In total, we collected 118 soil samples from the residential properties for chemical analyses: 50% from burned properties and 50% from unburned properties. We note that there was no visible ash on the surface of the sampled soils. We presume that any ash had been transported away from the sampling site via wind or transported deeper into the soil profile via rainfall or melting snow over the four-month period between the fire event and the soil sampling.

Soil cores from 33 nonresidential, grassland locations were also collected inside and outside of the fire perimeter (Figure 1). Soils from these grassland sampling sites were collected in the same manner as described above with eight soil cores (0–5 cm) collected and composited per sampling location. This effort yielded a total of 17 fire-affected nonresidential, grassland soil samples collected from sites within the fire perimeter and a corresponding 16 grassland soil samples collected from neighboring sites outside the fire perimeter. No ash was visible at the surface of these soils but we did note the presence of partially combusted vegetation (shrubs, grass, trees) in close proximity to all of the grassland sampling sites located within the fire perimeter.

**Bulk Soil Chemical Analyses.** For measuring pH, total nitrogen (TN), total carbon (TC), total inorganic carbon (TIC), and total organic carbon (TOC), soils were air-dried for 1 week and then ground using a roller mill for 24 h. A subsample of each soil ( $100 \pm 0.01$  mg) was combusted at  $>1000$  °C within a CN 802 Carbon Nitrogen Elemental Analyzer (VELP Scientific, Deer Park, NY) to determine TC and TN content (LOD = 0.5% TC and 0.001 mg TN). TIC was calculated using a modified pressure-calorimeter method.<sup>28</sup> Total TOC was calculated by subtracting TIC from the TC. Soil pH was measured in a 1:1 soil/water slurry.<sup>29</sup>

**Soil Metal Analyses.** EPA Method 6200<sup>30</sup> was used for X-ray fluorescence (XRF) measurements of bulk soil concentrations of common metals. Air-dried, ground soil samples were packed tightly into plastic XRF sampling cups (SC-4331-32 mm) and covered with a 4  $\mu$ m thick plastic film (SpectroMembrane, Chemplex Industries). A hand-held XRF analyzer (Olympus Delta) was used in a secure tabletop setup to maximize accuracy of total metal concentrations with the analyzer calibrated using a certified XRF reference material before conducting analyses on the soil samples. We focused our analyses only on elements that are potentially toxic (Zn, Pb, Cu, Ni, Cr, As) or generally nontoxic elements that are important soil nutrients (Ca, Fe, K, P, S, Mn, Cl).

A subset of the soil samples collected from the burned and unburned residential properties (43 samples in total) were also analyzed via inductively coupled plasma-mass spectrometry (ICP-MS) to provide a better indication of metal bioavailability. This subset was selected from 19 soils for which the XRF results indicated high concentrations of metals of human health concern and an additional 24 soils collected from six properties (4 samples per property) which were heavily impacted by the fire event. The air-dried and ground soil samples were extracted using the Mehlich 3 method: a

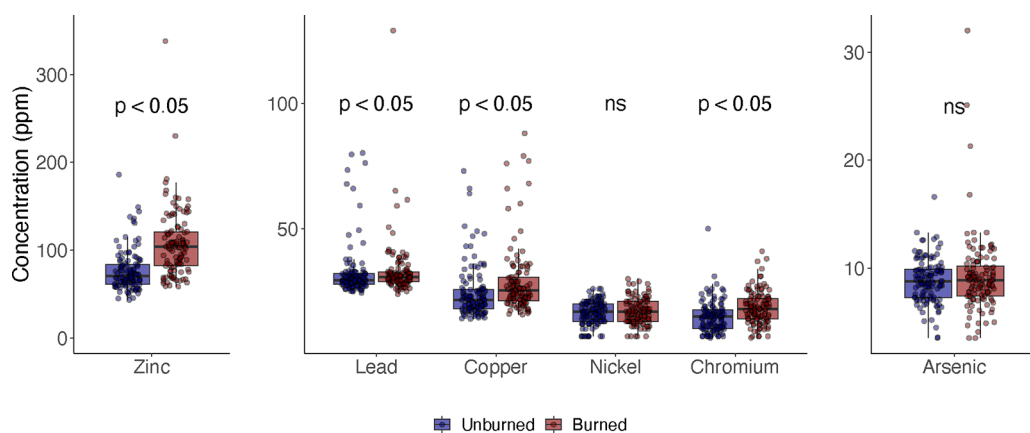
nondigestion method that is more representative of metal bioavailability than total digestion.<sup>31</sup> The extractant solution utilized trace metal-grade quality chemicals and all clean glassware was thoroughly washed with pH 2 HCl. The extracts were diluted with 2% nitric acid and then analyzed with an ICP-MS instrument (NexION 350D, PerkinElmer, Waltham, MA) following EPA Method 6020<sup>32</sup> with 6 calibration standards and multiple internal standards run alongside the soil samples.

**Soil Polycyclic Aromatic Hydrocarbon (PAH) Analyses.** We selected a subset of 20 soil samples for PAH analyses with one randomly chosen sample from each of 10 burned and 10 unburned properties. The ten burned samples were selected from residential properties located in the center of neighborhoods with extensive fire damage. The ten unburned samples were selected from nearby residential properties that were unaffected by the fire. PAH extraction was conducted following a cyclohexane/acetone extraction using a Soxhlet distillation extractor following a protocol described previously.<sup>33</sup> A mixture of standards containing 5 ppm of phenanthrene, fluoranthene, anthracene, naphthalene, chrysene, acenaphthylene, pyrene, perylene, benzo[a]pyrene, benzo[a,h]anthracene, 9-methylanthracene, 9,10-dimethylanthracene, 1-phenylnaphthalene, 2-ethylnaphthalene, and 1-methylnaphthalene, and a nitrobenzene-d5 (15 ppm) internal standard was prepared for accurate quantification. This mixture was serially diluted to 2.5, 1, 0.5, 0.1, and 0.05 ppm. Singular standards were also made to confirm retention times. These PAH compounds were selected since they are all known to be toxic and are likely present in the WUI fire-impacted soils.

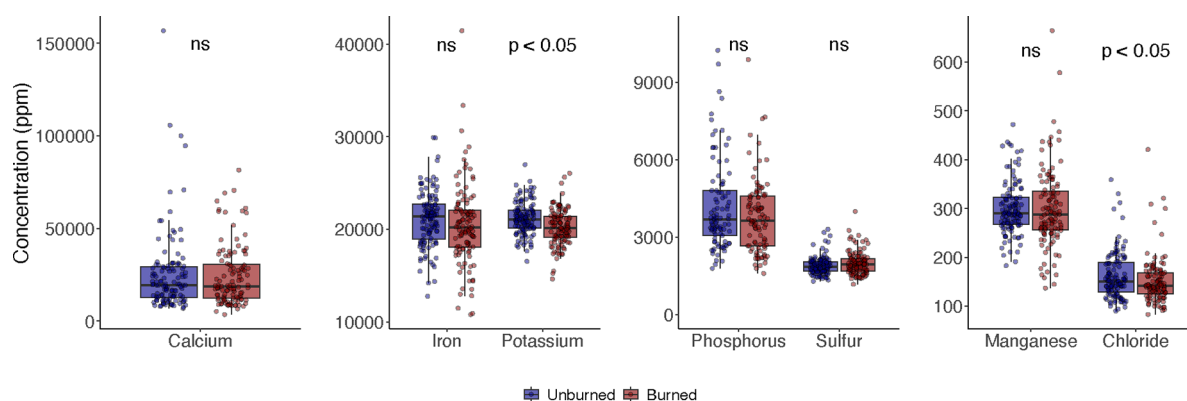
PAH concentrations in the Soxhlet extracts were quantified using atmospheric pressure gas chromatography (APGC) coupled with Waters Xevo G2 quadrupole time-of-flight (QTOF) mass spectrometer (Milford, MA). One microliter of each sample was injected and compounds separated using an Agilent DB-5MS (30m  $\times$  250  $\mu$ m  $\times$  0.25  $\mu$ m) column. The chromatography temperature gradient was set to 18 °C/min with a final temperature of 330 °C resulting in a run time of 16.89 min. The inlet was split/splitless using a pulsed splitless mode. The pulse time was 0.50 min. The mass spectrometer was run in positive mode with 0.1 s of scan time at a collision energy of 4 V. The analyzer was set to sensitivity mode with an API source. The scan range was 50–1200 Da.

**Soil Mercury Analyses.** A subset of 31 soil samples from both burned and unburned areas were analyzed for their mercury concentrations on a Milestone Direct Mercury Analyzer (DMA) at AIRIE, Fort Collins, Colorado. The detection limit is about 5 ppb, and the external precision is  $<5\%$  at one standard deviation. AIRIE routinely verifies external precision through analysis of an in-house organic-rich garden soil and the NIST SRM 1632e. Standard data and analytical details including stepwise temperature tests are available at <https://www.airieprogram.org>. Whole soil samples (full organic plus clastic components) were used for this study. Sample sizes were between 50 and 200 mg of soil, optimized to yield 3–12 ng total Hg for DMA analysis. Samples were combusted at 650 °C to release all Hg.

**Statistical Analyses.** All statistical analyses were performed in R (4.2.1). The XRF data and soil chemistry data were based on a hierarchical experimental design with four sampling locations per residential property. Thus, we used linear mixed effects models to determine differences in metal concentrations and soil characteristics between burned and



**Figure 2.** Comparison of metal concentrations in unburned and burned soils from residential properties as measured by XRF, with the statistical significance of the differences indicated for each element (ns = not significant). Note different scales on y-axes. These soil metal concentrations, regardless of burn status, are all well below the concentrations typically considered to pose a risk to human health (see Discussion).



**Figure 3.** Comparison of the concentrations of generally nontoxic elements in soils from burned and unburned residential properties as measured by XRF with the statistical significance of the differences indicated for each element (ns = not significant). Note different scales on y-axes.

unburned residential soils with property as a random effect in the *lme4* package.<sup>34</sup> For each response variable, we implemented a mixed effects model using the Gaussian error distribution. When necessary, we used a log transformation of the response variable to meet assumptions of the linear model. Values that were beyond three absolute deviations from the median value were considered outliers and were removed.<sup>35,36</sup> Elemental concentrations below detection limits were adjusted to the limit of detection divided by the square root of two. For analyses where a hierarchical experimental design was not used (including soil metal concentrations as determined by ICP-MS, soil Hg and PAH concentrations, and analyses of the nonresidential, grassland soils from burned and unburned locations) we used parametric *t* tests and Wilcoxon rank-based tests, depending on the distribution of the data.

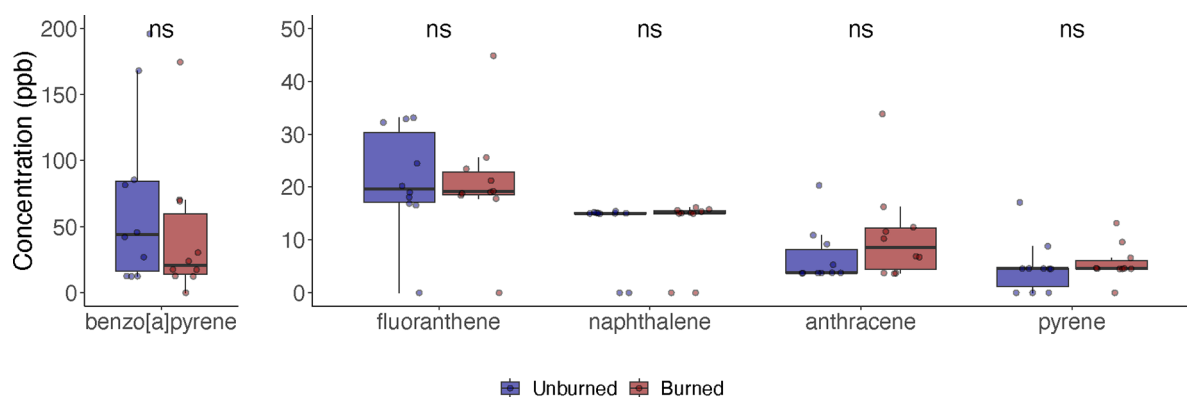
## RESULTS

Measured total soil nitrogen (TN), total inorganic carbon (TIC), total organic carbon (TOC), and soil pH were not statistically different between burned and unburned residential properties (linear mixed effects models, *p*-value >0.05) (Supplementary Table 1). However, TN concentrations in burned nonresidential grassland soils ( $0.31 \pm 0.13\%$ ) were significantly lower as compared to unburned grassland soils ( $0.36 \pm 0.19\%$ , linear mixed effect model,  $t = 2.25$ , *p*-value = 0.02) (Supplementary Table 2).

The results of the XRF analyses show that the fire-impacted residential properties had significantly higher concentrations of total soil Cr, Cu, Pb, and Zn (Figure 2, Supplementary Table 3) which are typically considered toxic contaminants.<sup>37,38</sup> Fire-impacted properties had significantly lower concentrations of Cl and K (Figure 3, Supplementary Table 3). For the nonresidential grassland soils, total metal concentrations were not significantly different between burned and unburned soils (Supplementary Figure 1, Supplementary Table 4) with the exception of Cl concentrations which were significantly lower in the fire-impacted nonresidential grassland soils (Supplementary Figure 1, linear mixed effect model,  $t = 3.017$ , *p*-value = 0.003).

ICP-MS analyses of the bioavailable fraction demonstrated that Zn, As, and P concentrations were significantly higher in fire-impacted residential properties (Supplementary Figure 2, Supplementary Table 5). We also measured mercury concentrations in a subset of samples and found that mercury concentrations do not appear to be significantly different between soils from burned and unburned residential properties (Supplementary Figure 3, Supplementary Table 7).

Concentrations of the 12 targeted PAHs were not statistically different between burned and unburned soils from the residential properties (Supplementary Table 6). Importantly, the concentrations of the five PAHs of most concern for human health (anthracene, pyrene, benzo[a]-



**Figure 4.** Concentrations of five PAHs of human health concern in residential soils. Note different scales on y-axes. We found no significant differences in PAH concentrations between burned and unburned soil samples from the residential properties (Wilcoxon rank-based test,  $p > 0.05$ ). ns = not significant. For the concentrations of the other measured PAHs, see Supplemental Table 6.

pyrene, fluoranthene, and naphthalene, shown in Figure 4) were not different between burned and unburned properties.

## DISCUSSION

Some of the toxic metals measured, namely Cr, Cu, Pb, and Zn were more concentrated in soils from burned residential properties compared to soils from unburned properties (Figure 2). However, the concentrations of these metals were not elevated in nonresidential grassland areas (Supplemental Figure 1), highlighting the likely importance of anthropogenic material combustion (including structures and vehicles) to postfire soil contamination. These results are consistent with previous studies which have also documented increases in soil metal concentrations after fire events driven in part by the burning of structures and other urban infrastructure.<sup>15,18,21–23</sup> While other studies of *wildland* fires have observed increases in soil PAH concentrations postfire,<sup>17,20</sup> such increases were not observed in this study (Figure 4, Supplemental Table 6). However, it is difficult to place our PAH results in the context of the broader literature given the lack of studies examining WUI fire-driven soil PAH concentrations.

Given the elevated concentrations of some toxic metals in burned residential properties, we sought to determine if soil contamination from WUI wildfires could pose a threat to human health. To determine the human health risk of the quantified concentrations of the PAHs and metals, we used the threshold concentration for dermal exposure to children in residential soils using a Target Hazard Quotient (THQ) = 0.1, a conservative approach to estimating toxicity/carcinogenicity risk when exposed to multiple contaminants.<sup>39</sup> We determined that dermal exposures were the most plausible exposure pathway for metal contaminants since it is unlikely that residential soil would be ingested or inhaled in amounts of any concern to human health. Although concentrations of toxic metals did increase in fire-impacted residential soils (Figure 2), their concentrations were well below the estimated thresholds of concern. More specifically, we note that the concentrations of Zn, Cr, As, Cu, and Pb measured in the collected soils, even soils from burned residential properties, are all well below thresholds commonly considered to pose a risk to human health through dermal exposure.<sup>39</sup>

We acknowledge there are still unquantified risks associated with these postfire changes in soil metal concentrations since our analysis only quantified elemental concentrations. Thus, we cannot exclude the possibility that the fire event mobilized

or transformed metals to increase their bioavailability/toxicity and, perhaps, elevate their risk to human health. For instance, even at low concentrations, methylmercury and hexavalent chromium can threaten human health.<sup>40,41</sup> Finally, we are hesitant to define risk with specific threshold values for soil metal concentrations given that the risk is context dependent. Health risks associated with exposure to metals in wind-blown soil (dust) could be very different from risks associated with growing and consuming crops in fire-impacted soils or risks associated with the movement of metals from soils into aquifers.<sup>42</sup> We need additional studies of soil contamination after WUI wildfires to specifically examine how the bioavailability of metals or other contaminants are affected by wildfires and the potential for the transport of soil contaminants into the atmosphere and water sources.

All wildfire events are unique, and the effects of wildfires on soils will always be context dependent. The specific effects of fire events on soil chemistry will vary depending on the intensity and duration of the fire event, soil properties, and the characteristics of the combusted materials.<sup>18,21</sup> Thus, we need to be careful not to extrapolate the results from this single wildfire event to all wildfire events, including those in other WUIs. The high winds associated with this fire event may have dispersed contaminants away from the fire perimeter, reducing the potential for local soil contamination. Likewise, the dominant grassland vegetation type in the sampled area could lead to distinct fire characteristics (faster moving, lower intensity) than wildfire events that may occur in more forested WUI areas. Also, we sampled soils four months after the fire event, making it possible that contaminants were transported away from surface soils by wind, surface water runoff,<sup>43</sup> postfire debris removal, or movement of contaminants deeper into the soil profile with melting snow or rainfall. Similarly, we note that we only collected soils from a small fraction of the residential properties located within or adjacent to the fire perimeter and we only collected soil from four locations per property. Clearly there is a high degree of spatial variation in soil PAH and metal concentrations (Figures 2–4). This variation, which is likely a product of soil heterogeneity, landscape position, ash transport and deposition, land-use practices, proximity to combusted structures or vehicles, and other factors, needs to be accounted for in future studies, particularly in highly heterogeneous residential areas. Sampling soils from fire-impacted residential areas is not trivial and

Careful design of such sampling efforts is essential when trying to quantify postfire soil contamination.

Soil contamination results from wildland fire studies may not necessarily apply to WUI fires. The nature and distribution of the contaminants are likely affected by the unique source of combusted materials. WUI regions can be highly dissimilar, and the wildfire events that may occur in these areas can be highly variable. There is a clear need for more studies specifically investigating the impacts of wildfires at the WUI on potential soil contamination. WUI wildfires are likely to become more common and pose greater risks to human populations in coming years.<sup>2</sup> Thus, additional information is needed on the impacts of WUI fires on soil quality to guide policy and provide informed recommendations for the resettlement of areas affected by WUI wildfires.

## ■ ASSOCIATED CONTENT

### SI Supporting Information

The Supporting Information is available free of charge at <https://pubs.acs.org/doi/10.1021/acs.est.3c08508>.

Supplemental Table 1: Summary of basic soil chemical characteristics for soils collected from burned and unburned residential properties. Supplemental Table 2: Summary of basic soil chemical characteristics for soils collected from burned and unburned nonresidential grassland sites. Supplemental Table 3: Summary of soil metal and metalloid concentrations as measured via XRF for the samples collected from the residential properties. Supplemental Table 4: Summary of soil metal and metalloid concentrations as measured via XRF for the samples collected from the nonresidential grassland locations. Supplemental Table 5: Summary of soil metal and metalloid concentrations as measured via ICP-MS for the subset of samples collected from residential properties. Supplemental Table 6: Summary of measured PAH concentrations for the subset of soil samples collected from residential properties. Supplemental Table 7: Subset of samples with measured Hg concentrations in soils from burned and unburned residential sites. Supplemental Figure 1: Comparison of the concentrations of metals (as measured via XRF) in soils collected from the nonresidential grassland locations. Supplemental Figure 2: Comparison of metal and metalloid concentrations in burned and unburned residential properties as measured by ICP-MS. Supplemental Figure 3: Concentrations of mercury in a subset of soils collected from burned and unburned residential properties. (PDF)

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## Notes

The authors declare no competing financial interest.

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