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Review

Agricultural Residue Burning and the Fate of Potentially Toxic Elements: From Particulate Emissions to Soil Mobility and Bioavailability

Nuttapon Khongdee [1,2], Kamonchanok Huangmee [3], Raweerat Rukkhun [4], Yupa Chromkaew [2,5], Kanyasorn Prethawin [6], Yao-Tung Lin [7] and Kesinee Iamsaard [6,*]

[1] Department of Highland Agriculture and Natural Resources, Chiang Mai University, Chiang Mai 50200, Thailand; nuttapon.k@cmu.ac.th
[2] Agrobiodiversity in Highland and Sustainable Utilization Research Group, Chiang Mai University, Chiang Mai 50200, Thailand; yupa.c@cmu.ac.th
[3] Department of Agricultural Science, Naresuan University, Phitsanulok 65000, Thailand; kamonchanokh@nu.ac.th
[4] Faculty of Natural Resources, Prince of Songkla University, Songkhla 90110, Thailand; jureerat.ru@psu.ac.th
[5] Department of Plant and Soil Sciences, Chiang Mai University, Chiang Mai 50200, Thailand
[6] Department of Natural Resource and Environment, Naresuan University, Phitsanulok 65000, Thailand; kanyasornp69@nu.ac.th
[7] Department of Soil and Environmental Sciences, National Chung Hsing University, Taichung 402227, Taiwan; yaotung@nchu.edu.tw
* Correspondence: kesineei@nu.ac.th

Abstract

Agricultural residue burning is a significant source of particulate matter (PM)-bound potentially toxic elements (PTEs), which can be transported through the atmosphere and deposited onto agricultural soils, with potential implications for soil quality, food safety, and human health. However, the interactions between deposited PM-bound PTEs and soil properties, as well as their subsequent effects on PTE mobility and bioavailability, remain insufficiently understood. This review examines the environmental pathways of PM-bound PTEs originating from agricultural residue burning, with a focus on atmospheric transport and deposition, soil geochemical transformations, and implications for environmental and human health. The review synthesizes the published literature on PTE emissions from agricultural residue burning, the factors governing their partitioning and transport, and the processes controlling their behavior following deposition in soils. Particular attention is given to the roles of soil pH, organic matter, mineral composition, cation exchange capacity, and redox conditions in regulating PTE speciation, mobility, and bioavailability. The review also evaluates the implications of these processes for crop contamination, ecological risks, and potential human exposure. The reviewed evidence indicates that combustion conditions and the physicochemical characteristics of burned materials influence the partitioning of PTEs among particulate matter, residual ash, and the gas phase. Fine particulate matter, particularly PM2.5, can facilitate the atmospheric transport of PTEs, whereas their subsequent fate in soils is governed by interactions with soil components and by geochemical processes, including sorption, complexation, precipitation–dissolution, and redox reactions. These processes can alter PTE mobility and bioavailability, indicating that total PTE concentrations alone may be insufficient to characterize environmental risks. This review highlights the need to integrate atmospheric emission and deposition processes with soil geochemistry when assessing the environmental impacts of agricultural residue burning. Such an integrated perspective can support more

Academic Editor: Joaquim Esteves Da Silva

Received: 14 October 2026
Revised: 10 November 2026
Accepted: 11 November 2026
Published: 14 November 2026

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comprehensive risk assessment and inform strategies to reduce open burning, monitor contamination, and protect soil health and food safety.

Keywords: particulate matter; PTEs; agricultural residue burning; mobilization; bioavailability

1. Introduction

Agricultural residue burning is widely practiced by farmers worldwide as a rapid and cost-effective method for field clearing and crop residue management. Burning is a quick method to prepare land for the next crop and temporarily increases nutrients such as phosphorus (P) and potassium (K), potentially boosting crop productivity. However, the detrimental effects far exceed the benefits, as burning emits harmful pollutants, posing significant public health risks [1]. Fire is a substantial ecological disturbance that may significantly impact terrestrial ecosystems. It is crucial in determining vegetation patterns, affecting nutrient cycling, and modifying soil physicochemical qualities. The negative effects of residue burning far outnumber the positives, such as the results in the volatilization of organic carbon (OC) and nitrogen (N), leading to significant nutrient depletion and diminished OM contributions. Smoke from agricultural residue burning releases substantial amounts of particulate matter, including fine particles (PM2.5) and particles with aerodynamic diameters of up to 10 μm (PM10), which encompass both fine and coarse fractions. These particles can transport PTEs through the atmosphere and contribute to their deposition in surrounding and downwind agricultural environments [2,3].

Open-field burning of agricultural residues is an important source of PM and gaseous pollutants and can also release PTEs accumulated in plant tissues during crop growth. During combustion, PTEs can become enriched in ash and fine PM2.5, including elements such as Cr, Cd, As, Pb, and Ni, which may pose environmental and health risks even at low concentrations [4,5]. PTEs can disperse and accumulate throughout the environmental cycle, including surface soil, groundwater, and the atmosphere.

Most atmospheric PTEs adhere to PM and are deposited onto surface soil through dry and wet deposition. This contributes to the accumulation of PTEs in the soil's chemical pool. Concentrations of PM-bound PTEs vary considerably depending on emission sources, meteorological conditions, and particle size fractions. Typical concentrations in ambient PM range from a few ng m^{-3} to several hundred ng m^{-3} for elements such as Pb, Cd, Cr, Ni, Cu, and Zn, particularly in urban and industrial areas [6]. Surface soil PTEs can be transported with runoff to low-lying areas and water bodies, influenced by terrain and soil properties. This deposition process increases the availability of labile forms of PTEs that are more easily absorbed by plants and organisms, impacting environmental and human health outcomes [7]. Atmospheric wet deposition of PTEs primarily occurs in the dissolved phase, particularly in association with particles smaller than 3 kDa that can be directly assimilated by organisms. The speciation of PTEs in atmospheric dry deposition demonstrated significant mobility and bioavailability, with a substantial proportion of the acid-exchangeable fraction [8].

While the extensive literature separately reviews independent components of biomass burning, atmospheric haze monitoring, or historical PTE pollution in soils, there remains a critical knowledge gap regarding how these systems interact dynamically. This review provides a distinct conceptual advance by mapping a unified, multi-sphere pathway that directly links atmospheric aerosol physics to soil rhizosphere chemistry. Specifically, this review integrates evidence on how fire-driven alterations in receiving soil properties regulate the post-depositional speciation, mobility, and crop uptake of PTEs

associated with PMs. This cross-scale sequence remains poorly integrated in environmental assessments that typically focus on isolated compartments. To address these knowledge gaps, the review defines a clear analytical scope focused exclusively on the pathway linking agricultural residue burning to crop contamination. The synthesis is organized around three main objectives: (i) to explain the mechanisms by which agricultural residue burning influences the volatilization, transformation, and enrichment of potentially toxic elements (PTEs) in size-segregated particulate matter; (ii) to evaluate the atmospheric transport and dry and wet deposition of PM-bound PTEs and their contributions to contaminant inputs into agricultural soils; and (iii) to examine how soil physicochemical properties and post-depositional biogeochemical processes regulate PTE speciation, mobility, bioavailability, and potential uptake by agricultural crops. Accordingly, this review addresses two key research questions: (1) How does agricultural residue burning influence the release, transformation, and enrichment of PTEs in PM? (2) How do post-deposition processes in agricultural soils regulate the speciation, mobility, and bioavailability of PM-bound PTEs? By synthesizing these processes, this review delineates a mechanistic relationship among crop residue burning, the creation and atmospheric conveyance of PM-bound PTEs, their settling on agricultural soils, and their following biogeochemical alterations. This cohesive viewpoint aids in closing the existing knowledge disparity regarding atmospheric emission mechanisms and the mobility and bioavailability of soil-based potentially toxic elements, thereby enhancing our comprehension of how residue combustion could influence contaminant dynamics in agroecosystems. The compiled evidence can furnish a scientific foundation for evaluating potential hazards to soil integrity, crop safety, and ecological well-being, while facilitating the formulation of specific residue-burning management and soil restoration strategies to mitigate PTE exposure and accumulation in agricultural systems.

2. Literature Search Strategy and Databases

This article uses a structured narrative review approach to synthesize peer-reviewed evidence on agricultural residue burning, the atmospheric transport and deposition of PM-associated potentially toxic elements, and their subsequent behavior in agricultural soils. The literature was identified through searches of Scopus, Web of Science, and Google Scholar, with the final search conducted on 22 July 2026. Publications dated 1 January 2020 to 22 July 2026 were prioritized for the contemporary synthesis, whereas earlier foundational studies were used to explain established mechanisms. Evidence was interpreted according to its direct relevance to field agricultural burning, laboratory residue combustion, or analogous processes. Search keyword strings focused tightly on combinations of "particulate matter", "PTEs", "agricultural residue burning", "bioavailability", and "soil". The literature search used combinations of keywords connected by Boolean operators (AND and OR). A representative search string was ("agricultural residue burning" OR "crop residue burning" OR "biomass burning") AND ("particulate matter" OR "PM2.5" OR "PM10") AND ("potentially toxic elements" OR "heavy metals" OR "trace metals"). Additional search terms, including "atmospheric deposition", "soil mobility", "speciation", and "bioavailability", were used to identify studies addressing specific environmental processes. The relevance of publications was evaluated based on their titles, abstracts, and full texts, where available. Studies were selected according to their relevance to the review objectives rather than through a formal systematic screening procedure. Studies focusing strictly on localized urban vehicular emissions or severe point-source industrial mining operations were excluded to keep our thematic focus locked explicitly on the agroecosystem matrix. Studies published before 2020 were cited selectively for foundational concepts, whereas the contemporary literature emphasized studies published in 2020–2026. These background references are distinct from the papers forming

the central evidence synthesis. To maintain a clear distinction between directly relevant evidence and supporting mechanistic studies, the literature was classified into two categories. Direct evidence comprised studies investigating agricultural residue burning, including emissions of particulate matter and potentially toxic elements, atmospheric deposition, and their effects on agricultural soils. Indirect evidence included studies on forest fires, biomass pyrolysis, industrial thermal processes, and other related emission sources. These studies were used selectively to explain physicochemical mechanisms that have not been sufficiently investigated under agricultural residue-burning conditions. Findings from indirect evidence were interpreted cautiously and were not considered direct confirmation of processes occurring during open-field agricultural burning.

3. Direct Heating of Soil Beneath Burning Residues

3.1. Burning Temperature-Induced Transformation and Speciation of PTEs

Fires affect soil temperatures, leading to changes in soil properties based on the fire's intensity and duration. The temperature during residue burning alters PTE speciation, enhancing their mobility and changing their forms. Lower temperatures promote organic/carbonate binding, while higher temperatures decompose these bonds, increasing environmental risks [9]. In soil, As is bound in Fe-oxyhydroxides, but the burning of OM may trigger defined redox mechanisms involving specific electron donors, acceptors, and relevant mineral or microbial processes, possibly reducing As(V) to the more mobile and toxic As(III), which can be easily transported by water due to easier desorption [10]. In Cr, the stability and toxicity vary based on temperature and soil composition, with Cr(III) potentially converting to the more toxic Cr(VI), impacting food chains and the environment [11].

Previous studies reported that various temperatures have a significant effect on PTE speciation. At certain temperatures (≈200 to 400 °C), changes in PTE speciation occur, such as oxidation of Cr(III) to Cr(VI) [12]. At very high temperatures (>700 °C), the volatilization of Pb, Zn, and Cd becomes significant, particularly during combustion or sintering. Such temperatures are often higher than those found in typical field crop residue fires but can occur in industrial processes or intense wildfires with hot smoldering areas [13].

To avoid overgeneralization, the thermodynamic behavior of PTEs during open-field burning must be strictly qualified by PTEs and combustion temperature regimes.

Pb remains relatively stable at low-to-moderate fire thresholds (200–400 °C), but experiences significant volatilization exceeding 42% when thermal zones breach 800 °C, subsequently condensing onto the highly mobile fine particle fractions (PM2.5). Pb redistribution during field burning depends on fire severity and chemical speciation, with lower-severity fires potentially promoting Pb accumulation in soil and ash, while higher-severity fires may enhance the mobilization of certain Pb species, particularly those of anthropogenic origin [14].

Cr is less volatile and undergoes speciation-specific transformations within the residual ash matrix rather than escaping in gaseous form. In oxygen-rich, moderate-temperature field regimes (200–400 °C), the presence of iron/manganese oxide minerals can accelerate the oxidation of relatively stable trivalent chromium (Cr(III)) into the highly toxic, water-soluble, and carcinogenic hexavalent form (Cr(VI)) [15–17].

When behaving as a redox-sensitive metalloid, As is traditionally bound to Fe-oxyhydroxides in stable agricultural soils. However, under restricted oxygen or intense smoldering conditions where OM burns incompletely, the sudden release of disassociated electrons can reduce arsenate (As(V)) to the much more mobile and highly toxic arsenite (As(III)) state, dramatically increasing groundwater leaching risks post-fire [17,18].

The burning of residues is widely practiced to prepare soil for cultivation, leading to substantial alterations in soil physicochemical properties due to heat transfer from the

combustion of biomass [19]. Residue burning significantly modifies soil physicochemical properties, such as pH, CEC, texture, structure, and OM content, which affects the speciation of PTEs (Figure 1). During fire, OM experiences several changes, such as moisture vaporizing at 100 °C, lignin and hemicellulose degrading in the range of 130 °C to 200 °C, and cellulose undergoing chemical dehydration at 280 °C, impacting soil sorption capacity for PTEs. For example, these changes increase the solubility and harmfulness of Cr, converting it from Cr(III) to Cr(VI), which endangers plants, water, and human health. Other metals (such as Cd, As, Pb, and Hg) transition from organic/acid-soluble forms to more stable oxide-bound or residue fractions, which increase immediate risk [11,20]. Biomass burning results in ash deposition that increases soil pH and levels of Ca, P, K, Mg, and basic cations. However, it may also lead to long-term issues such as decreased soil fertility, lower carbon sequestration, and reduced soil resilience [21].

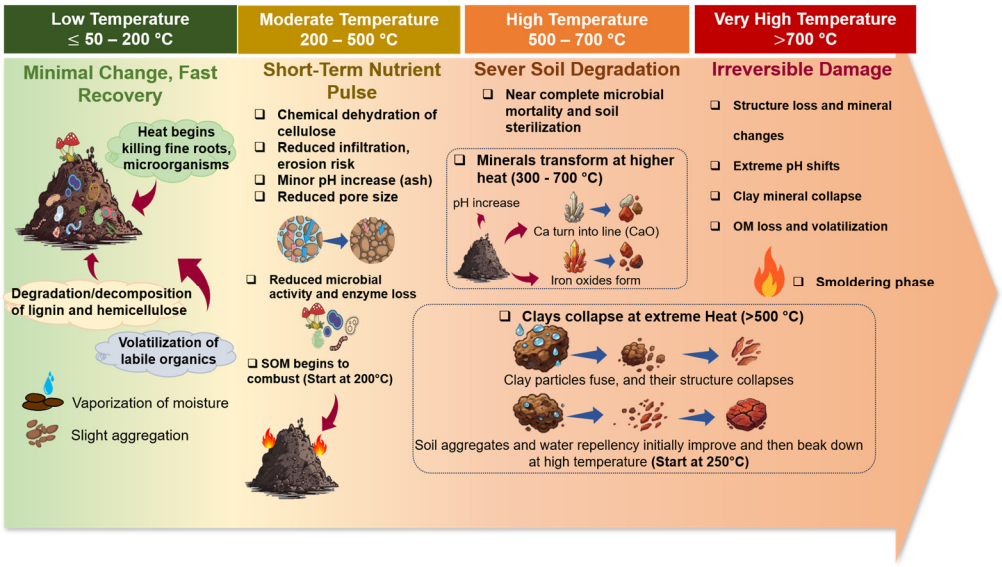


Figure 1. Effect of fire temperature on soil properties.

3.2. Release and Volatilization of PTEs During Combustion

PTE volatilization and transformation are strongly influenced by combustion temperature and atmospheric conditions. This review focuses primarily on open-field agricultural residue burning, which typically occurs at 300–700 °C, though localized hotspots may exceed this range. Due to limited mechanistic studies on open-field burning, relevant findings from biomass pyrolysis, thermal treatment, and burning are integrated to provide insights into PTE behavior under elevated temperatures. PTEs are emitted from sources such as biomass, coal, sludge, or trash during thermal processes (pyrolysis, combustion) by volatilization (gas evaporation) or transformation (speciation alterations, e.g., to oxides or chlorides) (Figure 2), with emission rates escalating at elevated temperatures (>500 to 700 °C for the majority). In addition, metallurgical activities, particularly ore smelting and metal refining processes, constitute major sources of airborne PTE-bearing pollutants due to the release of metal-containing fumes and PM at high temperatures [22]. Emission rates generally increase with temperature, particularly above 500–700 °C for many metals. Volatilization predominates for highly volatile metals (such as Cd, Pb, and Zn) at temperatures over 600 °C. The study of Li et al. [23] found that inorganic particles were initiated in the PM when the temperature was raised to 500–700 °C in a decomposition stage. An increase in pyrolysis temperature resulted in the enrichment of PTEs. The concentrations of Pb and Cd in the PM significantly increased when the temperature reached 700 °C.

At a temperature of 700 °C, the concentrations of Pb and Cd in the PMs were markedly elevated. The temperature ranged from 500 to 900 °C. Increased volatility of alkali and alkaline metals due to salt evaporation, together with enhanced pyrolysis, improves the release ratios of subsequent elements (Cd > Pb > Zn) [24]. Evidence derived from laboratory pyrolysis, controlled combustion, and industrial thermal processes should be distinguished from observations obtained during open-field agricultural residue burning. Although these studies provide valuable mechanistic information on PTE volatilization, oxidation, and condensation, differences in combustion temperature, oxygen availability, heating duration, and feedstock composition may limit their direct applicability to agricultural burning. Therefore, thermal transformation pathways identified under controlled experimental conditions are discussed as potential mechanisms rather than as confirmed processes occurring in agricultural fields.

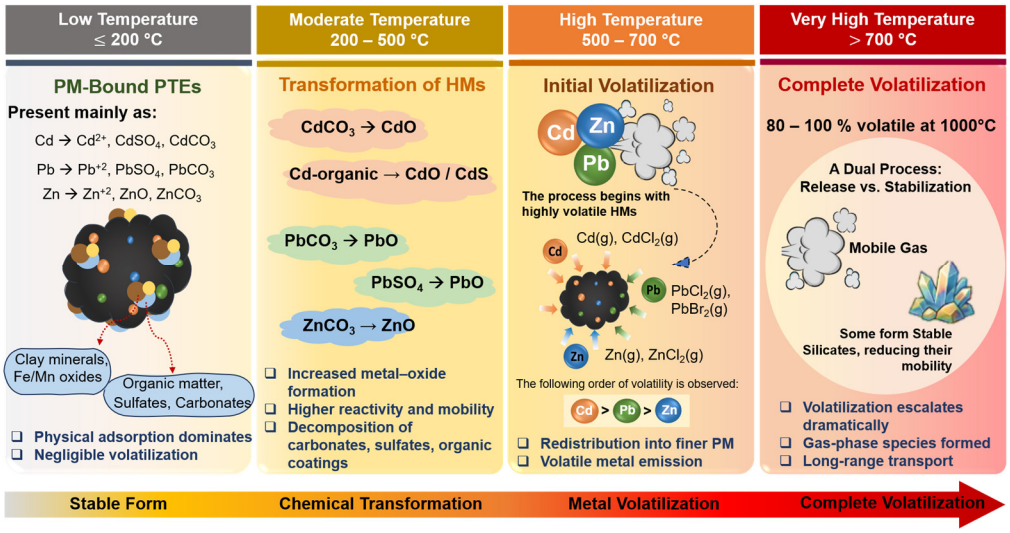


Figure 2. Temperature effect on volatilization and transformation of PTEs.

3.3. Size-Dependent Distribution and Enrichment of PM-Bound PTEs

PM contains various chemical components, including transition PTEs, PAHs, organic compounds, and biological materials. PTEs are significant as they generate reactive oxygen species (ROS) in biological systems, resulting in oxidative stress, which is a key mechanism behind PM-induced toxicity [25,26]. The deposition of particles varies by size, with fine particles migrating more effectively than coarse particles. Coarse particles are mainly deposited through dry deposition, whereas fine particles are removed from the atmosphere during wet deposition, such as during rain or snow, making wet deposition the key method for transporting atmospheric metals [27]. PM-bound PTEs exhibit different behaviors in PM2.5 compared to PM10 particles. The large specific surface area of PM2.5 allows it to effectively carry various chemical components such as PTEs, PAHs, elemental carbon, water-soluble inorganic ions, and OM, resulting in higher enrichment, extended atmospheric transport, and increased lung penetration [28,29]. PTEs such as aluminum (Al), As, Hg, Cd, Cr, Mn, Ni, Pb, selenium (Se), and thallium (Tl) are significant due to their complex sources, high toxicity, biological accumulation, and environmental persistence. PM10 typically deposits locally, but during transport and deposition, metals can transform in form (e.g., solubility, oxidation), affecting their environmental fate, bioavailability, and toxicity. This process often results in the enrichment of anthropogenic metals such as Cd, Pb, and Cr in PM2.5, alongside natural elements found in PM10 [30].

3.4. Emission Factors of PM2.5/PM10 from Residue Burning

Agricultural residues, primarily made of cellulose, hemicellulose, and lignin, are highly flammable. Their incomplete combustion contributes to the release of PM, carbonaceous aerosols, and gases, exposing nearby populations to elevated pollutant levels. Additionally, the uncontrolled burning of these residues results in significant emissions of OC and greenhouse gases (GHGs). Prediction indicates that emissions from residue burning may rise by 45% by 2050 if current practices continue unchanged [2,3].

Emission factors (EFs) are crucial for creating emissions inventories and control strategies in air quality management. EFs for PM2.5 and PM10 vary significantly based on biomass type and burning conditions (pile vs. non-pile). An investigation of PM2.5 emissions from rice residue burning in Vietnam reported EFs of 16.9 g kg^{-1} (±6.9) for pile burning and 8.8 g kg^{-1} (±3.5) for non-pile burning [31]. The EFs for fine PM2.5 and PM10 associated with the open burning of various agricultural and forest residues under different burning conditions are shown in Table 1. The EFs are quantified in grams of pollutant released per kilogram of dry biomass combusted (g kg^{-1}) and are derived from previously published research. The non-pile combustion of agricultural leftovers, including maize, barley straw, sorghum, alfalfa, soybean, sugarcane, and grass, often demonstrates lower to moderate PM2.5 and PM10 emission factors in comparison to the pile burning of forest residues and coconut husk. Pile burning of damp forest waste has the greatest emission factors for both PM2.5 and PM10, indicating incomplete combustion and increased moisture content. In contrast, the burning of dry forest leftovers in piles produces somewhat reduced particle emissions.

Although this review primarily focuses on agricultural residue burning, selected studies on forest-derived biomass combustion are included to provide complementary evidence on particulate emissions, atmospheric transport, and potential deposition of contaminants onto agricultural soils. These studies are clearly distinguished from agricultural residue burning to account for differences in biomass composition and combustion conditions.

Table 1. Emission factors for PM2.5/PM10 from burning residue.

Biomass Category	Residue Type	Burning Method	EFs (g kg^{-1})		References
			PM2.5	PM10	
Agricultural residues	Corn (maize)	Non-pile	10.5	11.7	[32]
Agricultural residues	Barley straw	Non-pile	1.1–4.1	1.7–6.0	[33]
Agricultural residues	Sorghum	Non-pile	3.5–5.5	5.0–8.0	[33]
Agricultural residues	Alfalfa	Non-pile	2.8–4.2	4.0–6.5	[33]
Agricultural residues	Soybean	Non-pile	4.1	4.5	[33]
Forest-derived biomass	Forest leaf litter	Non-pile	2.1–4.3	-	[34]
Agricultural residues	Sugarcane	Non-pile	6.5–7.1	7.2	[35]
Forest-derived biomass	Forest residue	Pile (wet)	18.0	20.2	[36]
Forest-derived biomass	Forest residue	Pile (dry)	4.9	5.5	

4. Transport and Deposition of PM-Bound PTEs

4.1. Atmospheric Transport of PM-Bound PTEs

Burning agricultural waste releases PTEs from plant materials and soil into the atmosphere, where some PTEs volatilize during combustion and subsequently condense onto fine PM, which serve as carriers, resulting in increased air pollution. The PTEs derive from plant absorption, soil, fertilizers, and feed additives, becoming airborne and adhering to PM during burning, thus generating harmful aerosols. Fly ash facilitates the environmental dispersion of PTEs and metalloids due to its small particle size, high specific

surface area, and long atmospheric residence [37]. Notably, toxic elements like As and Se are highly enriched in fly ash compared to biomass and bottom ash. Following atmospheric transport, dry and wet deposition deliver these contaminants to agricultural soils, increasing metal bioavailability and subsequent crop uptake. Thus, PM and fly ash serve as primary vectors transferring PTEs from biomass burning to soil ecosystems. PM can be emitted directly into the atmosphere as primary particles from combustion, dust, and soot, while others are formed in the atmosphere as secondary particles from the emissions of SO_2, NO_x, and NH_3 [38]. PTE concentrations in agricultural wastes vary widely depending on the type of waste, the crop variety, and the level of soil contamination in the region. Studies indicate that plants tend to accumulate PTEs such as Cd, As, Cr, Cu, Pb, and Fe when grown in polluted environments, such as industrial or smelting areas. Furthermore, greenhouse vegetables generally show elevated PTE contamination (Cu, Zn, Mn, Pb, and Cd) compared to open-field vegetables, which is likely attributed to lower light exposure.

PM-bound PTEs have potential long-range transport and significant health risks, including respiratory issues and potential carcinogenicity, as these metals deposit in the lungs and accumulate in soil, impacting crops and ecosystems [39]. The accumulation of PTEs via atmospheric deposition can result in their concentration in soil, water, and crops, leading to detrimental effects and modifications of physiological functioning in animals and humans through the food chain [40–42].

The previous section focused on the emission characteristics of PTEs linked to PM generated from burning agricultural waste and their initial association with PM. The atmospheric transport and deposition mechanisms that control the transfer of these pollutants from the atmosphere to agricultural soils are particularly examined in this section. Studies were screened based on their relevance to agricultural residue burning, PM-bound PTEs, atmospheric transport, deposition, soil processes, and bioavailability (Figure 2). Information extracted from each study included emission characteristics, transport pathways, deposition mechanisms, PTE speciation, soil physicochemical responses, and mitigation strategies. After emission from agricultural residue burning, PTEs are predominantly associated with fine particulate matter (PM2.5) because of its high specific surface area and abundance of reactive functional groups. During atmospheric transport, the residence time and dispersal distance of PM-bound PTEs are governed by particle size, meteorological conditions, and atmospheric stability. Finer particulates can linger in suspension for numerous days and experience extensive travel before being eliminated by dry settling or wet scavenging, while coarser particles (PM10) often settle close to the emission origin due to gravity forces [43]. Atmospheric aging further modifies the physicochemical properties of PM by promoting oxidation, secondary aerosol formation, and interactions with sulfate, nitrate, and organic compounds, thereby altering metal solubility and chemical speciation prior to deposition. Subsequent to deposition, the environmental behavior of PTEs is influenced by soil pH, OM, cation exchange capacity, and redox conditions, which together dictate the distribution of metals among exchangeable, carbonate-bound, oxide-bound, organic-bound, and residual fractions [44]. Consequently, the coupled processes of atmospheric transport, deposition, and post-deposition transformation govern the mobility, bioavailability, and ecological risks of PM-bound PTEs in agricultural soils [45].

In addition to wind speed, relative humidity, and precipitation, atmospheric transport and deposition of PM-bound PTEs are influenced by the vertical structure and stability of the atmospheric boundary layer. Atmospheric stability regulates turbulent mixing and vertical pollutant dispersion, whereas low-level jets can modify horizontal transport and vertical exchange through changes in wind shear and turbulence. These processes may influence the residence time, transport distance, and spatial distribution of particulate deposition in downwind agricultural environments [46]. Recent observations

over Lake Baikal demonstrated that smoke aerosols originating from regional wildfires contributed to elevated PM2.5 concentrations in areas distant from combustion sources. The study also reported increasing PM2.5 dry deposition fluxes, highlighting the potential for combustion-derived particles to be transported and subsequently deposited in downwind environments [47]. Although evidence from wildfire smoke and non-agricultural emission sources helps explain the atmospheric transport and deposition of PM-bound PTEs, these findings should not be directly generalized to agricultural residue burning. Differences in fuel composition, combustion intensity, particle size distribution, and emission characteristics may influence the chemical composition, atmospheric behavior, and deposition of PTE-bearing particles. Therefore, studies from non-agricultural sources are considered supporting mechanistic evidence rather than direct evidence of PTE emissions and deposition resulting from agricultural residue burning.

4.2. Dry and Wet Deposition onto Agricultural Soils

The transportation of HMs between soil and the atmosphere involves two key processes: the release of soil HMs into the atmosphere through fugitive dust and the deposition of atmospheric HMs into the soil via both dry and wet deposition. PAHs and PTEs are significant air pollutants linked to PM. PM aids in the transport of PAHs and PTEs, promoting their spread in the atmosphere and subsequent deposition on land via wet and dry processes. Once deposited, PAHs and HMs can accumulate in soils, exhibiting different environmental behaviors than in the atmosphere. Soil can function as both sinks and sources of pollutants, influenced by pollutant characteristics, land use, and environmental conditions [48,49]. The pathways of PM deposition involve both sources that emit particles into the atmosphere and natural mechanisms responsible for their removal. The transition from high to lower PM concentrations is facilitated by natural sinks, including dry deposition (settling on surfaces), moist deposition (removal by precipitation), and vegetation uptake by trees. PM is emitted from both human and natural sources and is distributed by the wind. Plant leaves capture PM through dry or wet deposition, while wind can resuspend it, and raindrops wash PM off leaves, depositing it into the soil. The presence of a plant belt can reduce suspended PM as indicated by the wind direction [50].

Atmospheric deposition significantly contributes to the global distribution and long-range transport of pollutants, particularly through dry and wet deposition processes, which facilitate the migration of metals from the atmosphere to ecosystems. The toxicity of deposited chemical species is influenced by their properties, such as inherent toxicity, environmental persistence, interactions with other chemicals, bioaccumulation, and post-deposition transformation in water [51,52]. In England and Wales, atmospheric deposition was found to be the predominant source of trace metals in agricultural regions, accounting for 78% of Pb, 56% of As, and 53% of Cd in agricultural soils annually [53]. Additionally, the distance metals can travel in the atmosphere is primarily affected by meteorological conditions, particle size, and aerosol characteristics, leading to varying exposure levels in urban versus rural environments.

Deposition velocity depends on particle size, gravitational settling, atmospheric turbulence, surface roughness, and meteorological conditions. For time-varying conditions, cumulative deposition can be estimated by integrating the deposition flux over the exposure period.

Atmospheric transport and deposition are also influenced by the vertical structure and stability of the atmospheric boundary layer. Stable atmospheric conditions may suppress vertical mixing, whereas enhanced turbulent exchange can redistribute pollutants vertically and influence their removal at the surface. Low-level jets may contribute to horizontal transport and modify pollutant dispersion through changes in wind shear and turbulence. Consequently, smoke plumes generated by agricultural residue burning may

transport PM-bound PTEs beyond the immediate combustion area, potentially contributing to deposition in downwind agricultural ecosystems. The extent of transport and deposition depends on emission characteristics, plume injection height, atmospheric circulation, particle size, precipitation, and surface conditions.

However, an important limitation is that most published studies investigate individual components of the emission–transport–deposition–soil continuum rather than the complete environmental pathway. As a result, the causal connections between PM-associated PTE emissions, atmospheric deposition, alterations in soil physicochemical properties, metal speciation, and crop absorption are inadequately substantiated in long-term field settings [54,55]. Future investigations that combine air surveillance, soil analysis, and plant absorption evaluations during various seasons would significantly enhance mechanistic comprehension and environmental risk evaluation.

4.3. Factors Controlling Atmospheric Transport and Deposition

Atmospheric deposition is a major pathway through which PTEs released from agricultural residue burning are transferred to agricultural soils. This process is influenced by several interrelated factors, including burning intensity, meteorological conditions, and landscape characteristics, which collectively regulate the atmospheric transport, dispersion, and deposition of PM-bound PTEs. Specifically, burning intensity and combustion conditions influence the initial release and partitioning of PTEs among PM, residual ash, and the gas phase. Meteorological conditions, particularly wind speed and precipitation, govern atmospheric dispersion as well as wet and dry deposition, whereas landscape characteristics affect the spatial distribution and deposition patterns of PM-bound PTEs. In addition, irrigation water and soil hydrological conditions may influence the redistribution and accumulation of PTEs following deposition. Although researchers have recognized these factors individually, their relative contributions and interactions in determining the transport, deposition, and subsequent accumulation of PM-bound PTEs from agricultural residue burning remain poorly understood. A better understanding of these interconnected factors is therefore essential for evaluating the environmental fate of PTEs in agricultural ecosystems.

(1) Meteorology

The concentration of PTEs in the atmosphere is strongly influenced by meteorological factors, including wind speed, temperature, humidity, and precipitation, which affect the dispersion, accumulation, and transformation of air pollutants. These factors determine both wet depositions, through processes such as rain washout, and dry depositions through direct settling [56,57]. Precipitation plays a role in washout processes, whereas humidity influences particle properties, thereby altering metal deposition on soil surfaces. Stable atmospheric conditions, including temperature inversions, can confine pollutants on the surface, leading to higher pollution levels. Furthermore, PTEs in the atmosphere exhibit the ability for extensive diffusion pushed by wind. Increased wind speeds generally facilitate movement and dispersion, hence presenting considerable threats to distant and vulnerable ecosystems [58,59].

(2) Landscape

The landscape is characterized as a mosaic of land use elements in terms of composition, structure, and function that may affect pollutant deposition and, consequently, local environmental pollution [60]. Landscape factors affecting the deposition of PM-bound PTEs to soils encompass atmospheric conditions (e.g., wind and precipitation), proximity to pollution sources (such as roadways and industrial sites), land use (vegetation and urban surfaces), and soil characteristics (including pH, OM content, and texture). These factors influence both the aerial transport and the retention/mobility of PTE post-deposition

[61,62]. Pollutants released into the environment can be transported over long distances and distributed across various ecological spheres, including the atmosphere, hydrosphere, lithosphere, and biosphere. Understanding the landscape factors that affect the transport, fate, and bioavailability of PTEs in ecosystems is crucial for predicting and mitigating their impacts on human and environmental health [5].

(3) Burning intensity

Biomass burning has become a global issue due to its significant emissions of fine PM, leading to atmospheric pollution, disrupting the radiation energy budget, and contributing to climate change. Burning intensity influences PTE soil deposition by modifying particle size and atmospheric transport. Higher intensity, from sources such as intense fires and industrial processes, results in increased fine PM carrying metals such as Pb, Cd, As, and Cr. The PM released from intense burning impacts not only the local atmosphere of burn areas but also affects distant regions via atmospheric transport and circulation [63,64]. This not only raises local deposition but can also mobilize these metals, with weather factors such as wind and humidity determining their final settlement. The burning releases nitrogen oxides (NO and NO_2, collectively referred to as NO_x), VOCs, and aerosols that impact tropospheric oxidants. Research indicates that the simultaneous release of NO_x and VOCs significantly boosts global tropospheric O_3 levels, rivaling or surpassing the contributions from urban pollution [65,66].

(4) Irrigation water, soil pore water and groundwater

Apart from atmospheric deposition, contaminated water sources are major pathways for the accumulation of PTEs in agricultural soils. Irrigation using water from rivers, reservoirs, reclaimed wastewater, or groundwater can introduce high concentrations of metals from industrial, mining, agricultural, and urban wastes. Repetitive irrigation causes these metals to accumulate in the topsoil, enhancing plant uptake. Concurrently, pore water regulates metal mobility and redistribution within the soil profile, with dissolution, adsorption, and transport driven by changes in soil moisture, redox conditions, pH, and dissolved OM [67]. Groundwater further contributes to soil pollution through capillary rise or irrigation in anthropogenically impacted areas. Ultimately, the interaction between atmospheric deposition and water-mediated transport determines the distribution, retention, and bioavailability of PTEs in agricultural soils [68,69].

5. Post-Deposition Transformation, Mobility, and Bioavailability of PTEs in Agricultural Soils

5.1. Adsorption–Desorption Processes

Additional PTEs can be added to agricultural soils through atmospheric deposition; these elements then split between the solid and soil solution phases. Both the amount retained by soil solids and the amount in operationally defined bioavailable or extractable pools may rise in tandem with an increase in total PTE loading. Therefore, while analyzing post-depositional PTE behavior, it is important to distinguish between sorption capacity, dissolved concentration, bioavailable amount, and the bioavailable proportion of the overall PTE pool. PM-bound PTEs that reach agricultural soils after atmospheric deposition experience dynamic partitioning between the soil solution and solid phase. Adsorption–desorption processes are therefore critical controllers of the mobility, persistence, and bioavailability of deposited PTEs [70]. PTEs may be retained by clay minerals, soil OM, Fe/Mn (oxyhydr)oxides, carbonates, and other reactive mineral surfaces via electrostatic attraction, ion exchange, surface complexation, and co-precipitation after being released from deposited particles by dissolution, weathering, or surface reactions [71]. Moderately mobile elements such as Cd and Zn can be readily extracted from exchange sites

in acidic conditions, whereas strongly sorbing cations like Pb may remain associated with OM, clay minerals, or Fe/Mn oxides. On the other hand, because their interactions with soil surfaces depend on both surface charge and aqueous speciation, oxyanion-forming PTEs like As and Cr(VI) display distinct pH-dependent behavior [72].

Deprotonation of pH-dependent surface functional groups on clay minerals, organic matter, and Fe/Mn (oxyhydr)oxides can raise the net negative surface charge as soil pH rises, thereby enhancing the sorption of several cationic PTE species. In addition, species like $M(OH)^{+}$ that can interact strongly with reactive soil surfaces may be produced by the hydrolysis of metal cations [67].

Oxyanion-forming PTEs, on the other hand, might show distinct pH-dependent sorption behavior. Because of variations in surface protonation and aqueous solutions, raising pH can decrease sorption and increase desorption from mineral surfaces. As a result of changes in speciation, the concentrations of dissolved As in the soil solution may rise [73]. As a result, the impact of raising pH on PTE retention varies depending on the element and species.

5.2. Redox-Induced Transformations

Redox conditions are a major control on the post-deposition fate of PM-bound PTEs in agricultural soils because changes in oxidation–reduction potential (Eh) can modify both the oxidation state of redox-sensitive elements and the stability of mineral phases that retain them [73]. PTEs linked to particulate matter may be redistributed among dissolved, exchangeable, oxide-bound, organic-associated, sulfide-bound, and residual pools after deposition. These changes are especially dynamic in agricultural soils that experience cyclical wetting and drying, irrigation, flooding, and drainage, where microbial activity and oxygen availability lead to significant variations in soil redox conditions. Therefore, depending on the element, its chemical form, and the surrounding soil environment, redox reactions can either mobilize or immobilize deposited PTEs. Eh interacts with pH, natural OM, mineral phases, and microbes to regulate metal (loid) transformation and availability, according to recent findings. One particularly significant connection between soil redox conditions and PTE mobility is iron and manganese (oxyhydr)oxides. Fe(III) and Mn(III/IV) oxides offer extremely reactive surfaces that can adsorb PTEs, form surface complexes, and promote their co-precipitation in oxic conditions. When soils become waterlogged and oxygen is depleted, microorganisms can use Fe(III) and Mn(IV) as alternative electron acceptors. The resulting reductive dissolution of Fe/Mn (oxyhydr)oxides can destroy important sorption phases and release previously associated PTEs into soil porewater. Conversely, re-oxidation during drainage promotes the formation of new Fe/Mn oxide phases, creating fresh sorption surfaces that can re-sequester dissolved PTEs. Thus, repeated wetting–drying cycles can continuously redistribute PTEs between solid and aqueous phases rather than producing a permanently immobilized contaminant pool [74]. This mechanism is particularly important for arsenic (As) because its mobility is closely coupled with Fe redox cycling. Under aerobic conditions, As(V) generally predominates and can be strongly retained by Fe(III) (oxyhydr)oxides. As soil conditions become reducing, microbial reduction of As(V) to As(III), together with reductive dissolution of Fe(III) minerals, can increase dissolved As concentrations in porewater. Further research reveals that Fe redox cycling includes secondary mineral production and recrystallization in addition to dissolution, which can redistribute As among newly created solid phases [75]. Therefore, As mobilization under reducing conditions reflects the combined effects of As speciation transformation and Fe-mineral transformation, rather than oxidation-state changes alone.

5.3. Influence of Soil pH and OM

Atmospheric metals attached to PM are co-transported with PM and subsequently deposited into ecosystems through wet and dry deposition mechanisms. Interfacial chemical processes at soil phase interfaces significantly influence the fate and removal of pollutants, determining their release into groundwater or retention in soil. Retention and transformation of PM-bound metals in soil matrices describe how airborne metals, carried by PM, get trapped (retained) in soil and change (transform) chemically. This interaction impacts the metal's harmfulness to plants or potential leaching into water, influenced by minerals, OM, and microorganisms within heterogeneous soil ecosystems that create organic–mineral complexes and aggregates [11,76]. The speciation and bioavailability of HMs in soils are affected by various factors such as binding to solids, ionic states in soil solutions, and adsorption processes. These processes are influenced by precipitation, chelation, and mineral–organic interactions that can alter surface properties and affect the retention of HMs at soil interfaces. Key processes involved in retention and transformation include adsorption–desorption, precipitation–dissolution, oxidation–reduction, complexation, ion exchange, and redox transformations, altering metals' speciation and mobility within the soil system. The interactions are affected by soil physicochemical properties such as pH, redox potential, OM content, and mineralogical composition [77,78].

Comparison of burned and unburned soil reveals that fire significantly affects soil properties (Table 2). Residue burning in agriculture continues due to factors such as quick crop turnover, lack of access to alternative technologies, and economic pressures on smallholder farmers. Despite government incentives for practices like mechanized zero tillage, many farmers opt for burning because it is easy, inexpensive, and time-efficient [79]. For instance, almost 80% of rice straw is burned as the quickest method of disposal before planting the next crop [80]. Fire affects the mobility and bioavailability of PTEs by changing soil properties, such as OM decomposition and mineral weathering. These processes can release these elements, notably Cr, posing risks to agriculture and human health. Variations in soil redox conditions and OM can convert Cr into its more toxic Cr(VI) form [81].

Because ash contains alkaline components like Ca, Mg, and K, keeping it in the source field after burning crop residues might momentarily raise the pH of the surface soil. Rainfall or subsequent management, initial soil pH and buffering capacity, ash composition and loading, and other factors all affect the strength and duration of this reaction. Certain cationic PTEs, such as Cd and Pb, may be better retained as soil pH rises because it can change the protonation state and surface charge of soil colloids and reactive mineral surfaces. Where geochemical conditions encourage the development of sparingly soluble carbonate phases, carbonate precipitation may also aid in metal retention.

Conversely, the direct heating of the soil beneath incinerated remnants may modify OM and the corresponding sites for binding PTEs when adequate heat is delivered to the soil. However, these effects depend on the actual soil temperature reached at a given soil depth, the heating period, soil moisture, oxygen availability, and initial soil properties rather than on heating temperature alone. Reduced organic complexation and possibly increased mobility of previously bound Cd and Pb might result from loss or modification of soil OM. However, because short-duration field burns, especially in moist soils, may result in far less soil heating than indicated by combustion-zone temperatures, this reaction should not be applied to all residue-burning events [82].

Table 2. Comparison of burned and unburned residues on soil properties.

Soil Property/Process	Burned Soils	Timescale/Controlling Conditions	Reference
SOM	Short-term increase due to ash content	Immediate to long term; influenced by combustion severity, soil heating, initial SOM, charred residues, and sampling time	[83]
Total N and available P	Short-term increase due to ash content	Immediate to short term; dependent on temperature, duration, initial N, and combustion completeness	[83]
Exchangeable K, Ca, Mg	Increase after burning due to ash content	Usually immediate to short term; dependent on ash retention, soil pH, mineral reactions, and plant uptake/leaching	[83]
CEC	Decrease	Short to longer term; dependent on OM, clay mineralogy, pH, and degree of soil heating	[84]
pH	Increases	Immediate to short term, potentially transient; dependent on initial pH, buffering capacity, ash loading, rainfall, and leaching	[83]
Infiltration and water retention	Decreases	Short to longer term; dependent on aggregate stability, pore structure, hydrophobicity, SOM alteration, burn conditions, and recovery	[85]
Microbial biomass	Decreases	Immediate to long term; dependent on actual soil temperature, heating duration, moisture, substrate availability, and recolonization	[83]
PTEs mobility	May increase	Immediate to long term; controlled by pH, OM, redox conditions, mineral phases, ash inputs, and post-burning transformations	[86]
Long-term soil health	Degradation risk	Long term/repeated burning; dependent on burn frequency, management, climate, soil type, and recovery period	[83]

6. Evidence for Changes in PTE Accumulation and Bioavailability Following Residue Burning

6.1. Changes in Bioavailable and Mobile PTE Fractions

Fire can be considered a boon or a bane, depending on the changes it brings to different materials when subjected to it. However, the process of residue burning may increase the bioavailable fractions of PTEs under certain conditions, making them more mobile and toxic, which can heighten environmental risks despite stable total concentrations. High heat from fire releases metals and transforms inert forms into mobile ones, posing immediate health and ecological threats through food chains and water contamination [87]. In northern Thailand, fire is commonly employed to eradicate vegetative remnants before the planting of highland rice (*Oryza sativa*). After burning residue in the rotational shifting cultivation of upland rice soil, the highest concentrations of Pb, Cr, and Hg were found in the ash [88]. In addition, the levels of Cu, Cr, As, Pb, and Hg were analyzed in rice grains from rotational shifting cultivation soils where residue burning was practiced. The findings indicated that the concentration order was Pb > Cu > Cr > As, with Hg and Cd undetectable. Notably, Pb levels in the rice grains surpassed safety thresholds [1].

Temperature influences the availability of potentially hazardous elements in soil. Elevated temperatures can enhance the mobilization of harmful substances by accelerating OM decomposition and mineral weathering, which usually occurs over extended periods. Moreover, rapid temperature changes, such as those during fires, can quickly alter soil properties, affecting the availability of PTEs [81]. Burning mobilizes and releases metals associated with residue and soil into PM and ash. Consequently, metal-laden particles and

ash can redeposit in soils, altering overall concentrations and possibly the ratio of readily exchangeable and residual forms [82]. Available PTEs predicted metal transfer from soil to crops better than the total amount of PTEs. Land use could influence both the total amount and availability of PTEs through human activities and soil properties. PTE concentrations, soil properties, and land use patterns were reported to integrally cause high human health risks [76].

6.2. Changes in Total PTE Concentrations and Stocks

Table 3 summarizes the literature-derived ranges of PTE concentrations, atmospheric deposition fluxes, and soil-to-crop transfer factors reported in independent studies. These values originate from different geographical locations, environmental conditions, and experimental settings and therefore do not represent measurements from the same agricultural residue-burning event. The table provides an illustrative comparison of PTE behavior across environmental compartments rather than a quantitatively linked exposure pathway or risk assessment model. Consequently, the compiled values should not be used to calculate cumulative exposure or establish direct quantitative relationships between atmospheric emissions, soil contamination, and crop uptake.

Table 3. Synthesized environmental ranges of PM-bound PTEs across agricultural pathways.

PTE Type	Typical Concentration Range in Biomass Ash/PM (mg kg^{-1})	Estimated Atmospheric Deposition Flux Range (mg m^{-2} $year^{-1}$)	Average Crop Transfer Factor (TF) Range (Soil-to-Grain/Leaf)	Primary Environmental Speciation Control Framework	References
Pb	45.0–201.0	12.5–78.0	0.01–0.15	Managed primarily by post-fire pH surges; high cationic affinity for binding to alkaline ash cores.	[1,89,90]
Cd	2.1–18.5	0.8–5.4	0.20–1.50	Highly mobile; exhibits elevated TF values in acidic, low-OM soils post-combustion.	[89,91]
Cr	15.0–95.0	4.2–22.1	0.02–0.08	Strongly dependent on temperature-driven oxidation shifts (Cr(III) to Cr(VI))	[12]
As	8.4–42.0	1.5–11.2	0.05–0.45	Governed closely by post-fire microbial redox kinetics and soil moisture variations.	[81]
Cu	38.0–155.0	18.0–64.5	0.10–0.50	Essential element showing dose-dependent toxicity; strongly bound to residual OM.	[5]

Note: The reported values are illustrative ranges compiled from independent published studies and do not represent measurements from the same geographical location, experimental system, or agricultural residue-burning event. The concentration, deposition flux, and crop transfer factor values are not quantitatively linked and should not be interpreted as a validated exposure pathway or risk assessment model. Differences in environmental conditions, soil properties, crop species, and analytical methods may contribute to the variability among reported values.

6.3. PTE Uptake and Translocation in Crops

PM-associated PTEs deposited onto agricultural soils not only increase the total soil metal burden but also enhance the bioavailable metal fractions that are readily taken up and translocated by plants. Due to easy accumulation, non-biodegradability, and low volatility, PTE pollutants persist in the environment in the long term, leading to significant

implications for plant health and food safety. PM, including combustion-derived fly ash, serves as an important carrier of PTEs, facilitating their atmospheric transport and subsequent deposition onto agricultural soils. The toxicity of metal pollutants is reportedly dependent on chemical speciation and concentration. Speciation has a significant impact on environmental mobility [10]. The water-soluble fraction of a metal is characterized by rapid diffusivity and high solubility, allowing metal ions to be easily absorbed by plant roots through cation exchange. However, different crop types exhibit varying capacities to uptake, transfer, and accumulate metals from soil due to physiological metabolic differences [91]. For instance, previous studies indicated that leafy crops have a higher concentration of metals compared to green beans, tomatoes, potatoes, kohlrabi, and carrots. This is attributed to their large leaf area, proximity to the ground, and elevated transpiration rates, facilitating easier transfer of metals from the soil to leaves via the xylem. Additionally, soil properties, such as pH, dissolved OM, N, P, and S contents, are closely relevant to metal speciation and their bioavailability, and subsequently determine the amounts of metal being transferred to the edible tissues of crops, thereby modulating root absorption and xylem/phloem translocation dynamics [92].

6.4. Risks to Soil Biota and Microbial Communities

Soil serves as an essential habitat for soil biota and microbial communities, forming an important part of the food chain. These communities comprise microorganisms such as bacteria and fungi, as well as microbivores (like nematodes) that feed on them, and micropredators such as omnivorous and carnivorous nematodes. These organisms are vital for soil functions such as OM cycling, carbon sequestration, biogeochemical cycle maintenance, soil structure stability, and plant disease control. Due to their extensive presence and mobility, soil organisms are often the first to interact with PTEs in the soil. They are sensitive to environmental changes, particularly PTEs, and serve as effective indicators of soil toxicity and the effects of harmful compounds [93].

PTEs bound to PM can impair microbial growth by damaging proteins, cell membranes, enzymes, and DNA, especially in low-OM conditions that enhance metal mobility. These metals are often found in labile, exchangeable forms in the surface layer, which are easily accessible to soil organisms [94,95]. Moreover, environmental conditions substantially influence microbial responses to metal exposure. In acidic environments, PTEs can form free ionic species, increasing their bioavailability and toxicity to soil organisms. This reduces the attraction between the adsorbent and metal cations, resulting in greater bioavailability of PTEs, thus amplifying their toxicity to them [96]. Accumulation of PTEs correlates with organism populations and metabolic functions, particularly in heavily contaminated areas, where diversity and abundance vary significantly with different heavy metal types and concentrations. The effects of PTEs on microbes are complex; low contamination levels may stimulate certain microbiological indicators, while higher levels generally have a negative impact [97].

7. Remediation and Risk Mitigation Strategies

Soil contamination represents a critical environmental challenge with significant implications for ecosystem function, agricultural productivity, and public health. Numerous in situ and ex situ remediation technologies have been developed to mitigate PTE contamination in soils (Table 4). These approaches function to immobilize, extract, or restore contaminated soil systems. Soil remediation strategies are classified into four principal categories: physical, chemical, phytoremediation, and biological techniques. Each remediation methodology operates through distinct mechanistic pathways, presenting characteristic tradeoffs between performance efficacy and economic feasibility, with selection depending on site-specific conditions and remediation objectives [98,99]. Remediation

with sulfuric and phosphoric acids affects soil bacteria and earthworms due to changes in pH, electrical conductivity, and nutrient levels. High acid concentrations lead to nutrient loss, structural changes, and harm to microorganisms [100]. Consequently, when selecting a soil remediation technique, it is essential to evaluate the polluted site to choose the most suitable treatment strategy for each location. Soil stabilization methods for PTEs involve immobilizing contaminants through chemical binding or physical encapsulation, thus reducing mobility and toxicity.

Table 4. The alternative remediation technologies to treat contaminated soils.

Remediation Process	Technique	Target Contaminants	Description	References
Physical	Vitrification	PTEs, VOCs	Heats contaminated soil (1400–2000 °C), melting it to a glass-like solid, effectively immobilizing PTEs	[101]
	Thermal	Semi-volatile organic compounds (SVOCs), Hg, petroleum	Heats soil to remove or immobilize soil contaminants, which are fast and effective tools for the remediation of contaminated soils.	[102]
	Electrochemical bleaching	PTEs, VOCs, SVOCs, radionuclides	Direct current moves ions, often combined with other methods such as soil washing or stabilization for enhanced results.	[103]
	Hydrodynamic	SVOCs, PTEs, PAHs, pesticides	Separates soil particles by size/density using water flow (e.g., screening, flotation).	[104]
	Encapsulation	Hydrocarbons, halogenated organics	Uses polymers to physically bind contaminants, creating a solidified mass with reduced mobility.	[105]
Chemical	Oxidation	PTEs, PAHs	Injects oxidants (e.g., permanganate, peroxide) with water to break down contaminants.	[106]
	Soil washing	PTEs	Flushes soil with water or solutions to extract contaminants.	[107]
Biological	Bioreactors	PTEs, PAHs, hydrocarbons, pesticides, petroleum, organic pollutants	Mixes soil with water/nutrients in tanks, often with aeration, for intense microbial action.	[108]
	Flushing		Increases bioavailability of hydrophobic pollutants (PAHs, oils) for microbial degradation.	[107]
	Bioventing		Enhances oxygen supply to subsurface microbes via air injection, promoting aerobic degradation.	[109]

8. Research Gaps and Future Directions

Although substantial progress has been made in understanding particulate matter-bound PTEs emitted from agricultural residue burning, major knowledge gaps remain regarding how seasonal and climatic variability influence their emission, atmospheric transport, soil fate, and bioavailability. Existing studies have largely emphasized short-term observations or site-specific assessments, with limited attention to long-term seasonal patterns and climate-driven changes. Insufficient research has examined how shifts in temperature, precipitation, and other climatic factors affect burning-related emissions, atmospheric transformation processes, and the subsequent mobility and availability of metals in soil–crop systems.

To address these gaps, future studies should prioritize comprehensive field investigations that capture seasonal and climate-related variability in the emissions, size distribution, and chemical composition of PM generated by burning different agricultural residues under contrasting combustion conditions. Particular attention is needed to determine how these source-specific factors influence the partitioning of PTEs between fine PM, coarse ash, and the gas phase, and how PM-bound PTEs are transported and deposited in surrounding agricultural soils. These investigations should be complemented by validation approaches that link the characteristics of deposited PM and the chemical speciation of its associated PTEs to soil properties, crop contamination, and potential human health risks. At the same time, greater attention should be given to the post-deposition transformations of PM-bound PTEs, including changes in speciation, mobility, bioavailability, and uptake by crops, as these processes may differ from those of PTEs originating from other sources. Research should also assess bioaccumulation and trophic transfer within food webs under different seasonal and climatic conditions. By explicitly tracing PTEs from agricultural residue combustion to atmospheric transport and deposition to soil–plant systems, advancing research on PM-bound PTEs from agricultural residue burning will support a more climate-responsive understanding of their environmental and health implications. Such knowledge is essential for improving risk assessment frameworks and guiding sustainable agricultural management strategies under changing climatic conditions.

9. Limitations of the Current Literature

While this review integrates multi-sphere pathways connecting residue fires to crop contamination, several inherent limitations in the existing body of literature must be acknowledged. The majority of current datasets rely on short-term, localized field tracking or simulated laboratory pyrolysis experiments. These setups often fail to capture real-world, long-term multi-seasonal variations, wind-driven deposition shifts, and chronic metal accumulation trends under actual field management conditions.

A significant barrier to cross-study comparison is the lack of standardization across solid-phase sequential extraction protocols. Variations in operational parameters, such as solution pH, extraction times, and chemical selectivity boundaries between Tessier and BCR matrices, introduce high uncertainty when comparing metal liabilities across different soil contexts. In multi-use rural landscapes, it remains methodologically difficult to purely decouple the diffuse PTE inputs of agricultural residue burning from baseline volcanic background geochemistry, commercial phosphate fertilizer overprints, or historical point-source industrial fallout. Therefore, current exposure frameworks frequently rely on static soil-to-plant transfer factors, omitting the highly dynamic, temporal variations in rhizosphere biochemistry, plant transpiration spikes, and root–

microbe exudate interactions that actively fluctuate during the crop growth cycle post-burning.

10. Conclusions

Agricultural residue burning is a significant source of particulate matter (PM)-bound PTEs, including As, Cd, Pb, Cr, and Ni, which can be transported through the atmosphere and subsequently deposited onto agricultural soils. The partitioning of PTEs among residual ash, particulate matter, and the gas phase is governed by combustion conditions and the physicochemical characteristics of the burned materials. Fine particulate matter, particularly PM2.5, can act as a carrier of PTEs and facilitate their long-range atmospheric transport. Following deposition, soil properties and geochemical processes, including pH-dependent sorption, complexation, precipitation–dissolution, and redox reactions, regulate PTE speciation, mobility, and bioavailability. These processes have potential consequences for soil quality, crop contamination, ecological integrity, and human health.

Collectively, these findings indicate that assessing the impacts of agricultural residue burning requires consideration of both atmospheric emissions and the subsequent fate of PTEs in soils. Total PTE concentrations alone may not adequately characterize environmental risks, because mobility and bioavailability depend on chemical speciation and on interactions with soil components. Effective management should therefore combine the reduction in open burning with coordinated air and soil monitoring and, where necessary, site-specific measures to limit contaminant mobility. Such an integrated approach would help safeguard soil health, food safety, and the long-term sustainability of agricultural systems.

Author Contributions: N.K.: formal analysis, investigation, data curation, writing—original draft, writing—review and editing. K.H.: writing—original draft, data curation, writing—review and editing. R.R. and Y.C.: methodology, writing—review and editing. K.P.: writing—review and editing, visualization. Y.-T.L.: writing—review and editing. K.I.: conceptualization, writing—review and editing, visualization, supervision, funding acquisition. All authors have read and agreed to the published version of the manuscript.

Funding: This research received no external funding.

Data Availability Statement: No new data were created or analyzed in this study. Data sharing is not applicable to this article.

Acknowledgments: The authors would like to express their gratitude to the Appropriate Technologies for Waste Reuse and Management Research Unit, Naresuan University, Phitsanulok 65000, Thailand for their support and contributions. This research was partially supported by Chiang Mai University.

Conflicts of Interest: The authors declare no conflicts of interest.

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