Biogeochemical Control Points for Mitigating Metal(loid) Risk in Rice Under Global Change

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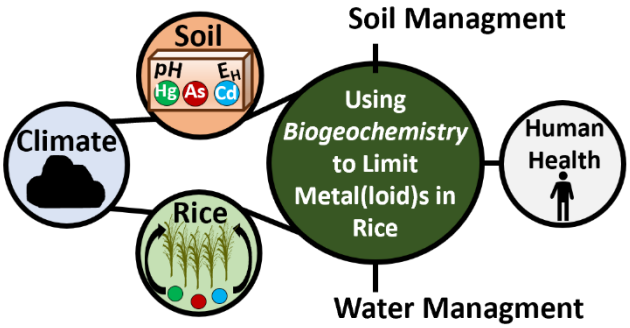
GRAPHICAL ABSTRACT



Abstract

The uptake of arsenic (As), cadmium (Cd) and mercury (Hg) in rice is a global health concern. The variable biogeochemical processes that govern contamination risk, along with increased anthropogenic pressures, present significant obstacles to implementing mitigation strategies. This perspective highlights biogeochemical processes governing the mobility and rice uptake of metal(loid)s across the soil-rice continuum, while providing recent insights for mitigation strategies. An overview of anthropogenic stressors that may further exacerbate rice contamination, particularly climate change, is provided. Finally, in the context of historical knowledge, our perspective for a future roadmap in research priorities and mitigation strategies is identified and discussed including (1) addressing variability in soil-rice systems that alter metal(loid) uptake dynamics, (2) investigating practical application and mitigation strategies for farmers, and (3) expanding research into the myriad effects climate change may have on metal(loid) cycling.

Subjects

- Arsenic
- Cadmium
- Mercury
- Rice
- Land Management
- Human Health Risk

Keywords

- Dietary Exposure
- Plant Metal Uptake
- Biogeochemistry
- Plant-soil interface

Synopsis

This perspective explores the biogeochemical processes of toxic metal(loid) uptake in rice, combining historical insights with recent advances, human health perspectives, a mitigation roadmap, and future research needs.

Introduction

Rice feeds half the world[1]; thus, the accumulation of the toxic elements arsenic (As), cadmium (Cd), and mercury (Hg) in rice grain is a global health concern. Reported concentrations vary widely depending on local conditions and whether the soil suffers from anthropogenic contamination or if the levels occur naturally. A large synthesis of Chinese commercial rice reported province-level mean concentrations of approximately 0.012-0.220 mg kg^{-1} As, 0.007-0.131 mg kg^{-1} Cd, and 0.0015-0.056 mg kg^{-1} Hg[2] but can reach 0.4, 4, and 0.5 mg kg^{-1} for inorganic As, Cd, and Hg, respectively[3,4], compared with international maximum levels of 0.1-0.35 mg kg^{-1} inorganic As and 0.15-0.4 mg kg^{-1} Cd depending on jurisdiction. Currently, only China has a Hg limit of 0.02 mg kg^{-1}, whereas only the E.U. and the U.S. have action levels for inorganic As in infant foods of 0.1 mg kg^{-1}.

Metal(loid) accumulation in grain is governed by often opposing biogeochemical processes that control mobility and transport from soil to grain[3]. For As and Hg, these processes also alter speciation, shifting the balance between inorganic and organic forms, which localize differently in rice grain and have distinct implications for human health. Consequently, the biogeochemical environment and post-processing of rice jointly determine human exposure to toxic metal(loid)s but may also be harnessed to devise mitigation strategies. Approximately 15% of global croplands exceed safe thresholds for soil metal concentrations[5] due to both geogenic and anthropogenic contamination. Moreover, the growing human-caused pressure on paddy soils warrants the development of mitigation strategies that are both practical and resilient, particularly in the face of a changing climate. Here, we argue that managing metal(loid) risk in rice requires a shift from element-specific mitigation toward a unified biogeochemical framework that explicitly accounts for redox dynamics, plant–soil interfaces, and global change pressures. We contend that many current mitigation strategies fail because they neither resolve inherent trade-offs among As, Cd, and Hg nor account for climate-driven shifts in soil processes. We therefore focus on (i) key biogeochemical control points that unify these elements, (ii) how global change perturbs these controls, and (iii) a forward-looking roadmap for mitigation that is both mechanistically grounded and practically deployable. These control points provide a unifying lens through which the often-divergent behaviors of As, Cd, and Hg can be understood and managed.

Biogeochemical Control Points

The biogeochemical environment affects the mobilization, plant-uptake, and grain localization of As and Hg species and Cd in paddies with natural levels of these metal(loid)s, which can be exacerbated by anthropogenic soil contamination[6]. Here, we focus on three biogeochemical control points (Figure 1) that consistently govern metal(loid) behavior in rice systems: (i) redox-driven mobilization, (ii) rhizosphere processes, and (iii) transporter-mediated uptake and grain localization.

Redox-driven mobilization and speciation.

Soil redox potential and pH are master variables that control metal(loid) mobility with fundamentally different—and often opposing—controls on As and Hg vs. Cd[7,8] . This divergence creates an inherent management tradeoff: conditions that mitigate As and Hg may exacerbate Cd risk, and *vice versa*. Under oxic conditions iron(III) oxides, hydroxides, and oxyhydroxides (hereafter, Fe(III) oxides) only weakly adsorb Cd(II) via electrostatic interactions but strongly adsorb inorganic Hg(II) and As species[9–11] through covalent bonding and inorganic Hg(II) also interacts with thiol groups in organic matter. Thus, Cd is readily plant-available under oxic conditions, whereas As and Hg are not. However, the anoxic conditions in flooded rice paddies lead to the microbially mediated reductive dissolution of Fe(III) oxide minerals that mobilize As and Hg to soil solution/porewater[11–14]. Once released to porewater, arsenic is transiently present as arsenate [inorganic As(V); $H_2AsO_4^-$ and $HAsO_4^{2-}$] before being microbially reduced to arsenite [inorganic As(III), $H_3AsO_3^o$, pka = 9.2], which is more soil-mobile than arsenate in rice paddies due to its neutral charge at pH < 9 and lower retention on soil minerals[15,16].

We emphasize that reducing conditions not only mobilize As and Hg but also drive the formation of organic As and Hg species with distinct implications for both yield and human health. Sulfate reduction promotes Hg methylation and formation of methyl-Hg [MeHg][17], which is plant-available and highly toxic. However, MeHg is often found at low levels in rice, except when Hg contamination is high[4]. The microbially mediated methylation of mobilized As to organic forms, including monomethylarsonic acid (MMA), dimethylarsinic acid (DMA), and, under sulfur-rich conditions, dimethyl monothioarsenate (DMMTA)[12,18] in reducing soil can increase grain organic-As concentrations and cause straighthead disorder, thereby reducing yield[19–21]. The assumption that organic As is “safe” because it is less acutely toxic than inorganic As is not universally valid. DMMTA is more acutely toxic than inorganic As, and much less is known about its prevalence in rice compared to inorganic As and DMA. This is partly because the methods previously used to characterize As species in rice led to the conversion of DMMTA to DMA and poor DMMTA elution[22,23], thereby overlooking the importance of this species in many older published studies[24]. Newer work has shown that DMMTA ranges from 0-20% of total As in rice[25]. Moreover, it is chronic exposure, not acute exposure, that is targeted with rice consumption, as concentrations of As high

enough to cause acute human toxicity would kill the plants. The fact that DMA and MMA are Class 2 carcinogens[26], meaning they are probably carcinogenic, suggests that they likely play a role in As-induced chronic disease, yet they remain largely absent from current regulatory frameworks. In addition, metabolism of pentavalent As-bearing DMA or MMA to the trivalent forms during metabolism may increase toxicity within the human gut[27].

The strongly reducing conditions that promote methylated As and Hg formation and higher grain As and Hg decrease Cd mobility and uptake[7]. In acidic soils, this is partially caused by pH increases toward neutral upon soil flooding, resulting in enhanced Cd adsorption onto mineral surfaces, as well as precipitation with sulfides as sparingly soluble CdS[28]. However, oxic conditions and contamination from mining or industrial pollution can increase Cd in rice grain. Although trace levels of metal(loid)s have always been present in rice, the direct linkage of elevated Cd in rice to human disease in the Jinzu River basin established rice as a critical pathway for food-chain contamination. It is now well-established that, except where anthropogenic pollution may lead to elevated Cd and Hg[4,29] , arsenic remains the dominant concern in flooded rice systems due to its ubiquity and enhanced mobilization and plant uptake under typical paddy conditions. Thus, we argue that mitigation strategies must be region-specific and target the metal(loid) of most concern to protect human health; there is no one-size-fits-all solution.

Rhizosphere processes. Once mobilized to the soil solution, rhizosphere processes act as a second control point regulating plant availability. The rhizosphere represents one of the most biochemically reactive zones in rice paddies, where plant activity, microbial metabolism, and mineral transformations converge to regulate uptake. The adaptation of aerenchyma (*i.e.*, hollowed cortical tissue) formation allows gas exchange between roots and the atmosphere, allowing root respiration in an otherwise oxygen-depleted environment[30–32]. Radial oxygen loss from less suberized zones in rice roots into otherwise reduced soils creates steep redox gradients that create a niche for iron(II)-oxidizing bacteria[33,34] and drive the oxidation of Fe(II) and the precipitation of Fe(III) oxide coatings on root surfaces, commonly referred to as Fe plaques[35–39]. Fe plaque forms discontinuous coatings of mixed mineral phases, including ferrihydrite, goethite, lepidocrocite, and sometimes siderite[19,40–42]. These transient mineral phases serve as adsorbents for As[40,43–45], Cd[40,46–48], and Hg[49], through adsorption and coprecipitation processes. The strength of the retention depends on the species, with sorption strength generally following: inorganic As(V) > As(III) > Hg^{2+} >> Cd^{2+} >> organic As and Hg species.

The formation of Fe plaque slows down, but does not prevent, toxic metal(loid) uptake by rice. Most Fe plaque forms under flooded conditions, and more plaque is found where Fe(II) encounters more oxygen in an otherwise flooded soil such as near

the soil surface, at lateral root junctions, and near root tips[43,50–52]. The longer the plant grows, the more Fe plaque develops on roots, but the rate of production is highest during the vegetative phase, peaks at booting, and decreases during heading[40,53]. The formation and dissolution of Fe plaque are highly dynamic, sorbing and releasing As, particularly under repeated cycles of Fe(III) reduction and Fe(II) oxidation[54], but ferrihydrite is always dominant, making up 50% or more of the total Fe phases[40,41,55]. While ferrihydrite has a higher reactive surface area for As retention via strong inner-sphere complexes[10,11,56], and plaque preferentially retains As relative to other elements[40], the increase of ferrihydrite does not counteract the effect of As mobilization due to reducing conditions in bulk soil. Thus, Fe plaque alone cannot minimize the uptake of As. Moreover, Cd and Hg form weaker interactions with Fe plaque minerals. Because of plaque's discontinuous coatings on the roots of rice[43], areas exist where plaque does not cover roots, allowing for plant uptake of metal(oid)s via root transporters.

Beyond mineral sorption processes, recent studies highlight the importance of reactive oxygen species (ROS) generated during Fe redox cycling or microbial activity in shaping metal(loid) transformations within the rhizosphere[57,58]. Oxidative radicals can facilitate the conversion of arsenite to arsenate, thereby limiting As mobility and uptake pathways in rice[59]. In contrast, Cd mobility and uptake increases due to free radicals formed during Fe(II) oxidation as paddies are drained[58] . Emerging evidence suggests that rice plants can indirectly influence Hg transformations through root oxygen release and root exudation, which create dynamic redox gradients in the rhizosphere[60]. These changes can consequently alter the activity and structure of Hg-transforming microorganisms, while simultaneously promoting ROS generation, leading to coupled biotic–abiotic Hg transformations such as (de)methylation[61,62]. The root-associated microbial communities are key drivers of Hg methylation through the activity of anaerobic microorganisms, such as sulfate- and iron(III)-reducing bacteria [60]. Conversely, microbial demethylation pathways can degrade methylmercury[4], thereby reducing its persistence and bioaccumulation potential. The balance between these opposing microbial processes is strongly influenced by rhizosphere redox dynamics and substrate availability.

Transporter-mediated uptake and grain localization. Once metal(loid)s reach rice root apoplasms, their transport toward grain is controlled by physiological barriers that determine root uptake and transport to edible tissues. Toxic elements often exploit nutrient transport pathways,[63,64] with Cd entering rice via Mn transporters[65] and inorganic arsenite and DMA via Si transporters[66,67] given their chemical similarities. In addition, toxic elements may enter plants without transporter regulation at lateral root junctions and root tips[50,51]. Inorganic arsenate can use phosphate transporters[68], but inorganic arsenate is not very abundant in paddy porewaters[69,70], rendering this

mechanism less important. Inorganic Hg(II) is strongly retained with thiol groups on plant cell walls and is not easily transported. The transporters responsible for MeHg and some organic As forms are not well understood. However, once taken up into the symplasm, the organic forms of As and Hg are more readily transported to grain than their inorganic forms via phloem mobility[71,72]. Two key control points occur at stem nodes and at husk, thereby influencing allocation to grain[71,73,74]. At stem nodes, vascular connections regulate partitioning between xylem and phloem[75,76], and deposition into the bran and grain are controlled at the ovular vascular trace[77]. These barriers vary among cultivars due to differences in transporter activity, sequestration capacity, and nodal transfer efficiency, leading to large differences in grain metalloid accumulation.[78]

We emphasize that metal(loid) speciation and plant transport processes ultimately determine not only grain concentrations but also the distribution of elements within the grain and their subsequent bioaccessibility to humans. Inorganic species tend to accumulate in the bran layer, whereas organic species are more readily translocated to and localized within the endosperm[71]. As a result, polishing brown rice to produce white rice can decrease inorganic metal(loid) concentrations but has limited effect on organic species[79]. This distinction introduces an important tradeoff: polishing removes not only inorganic species but also essential micronutrients such as Zn, Fe, and Mn, potentially increasing the relative bioaccessibility of remaining toxic metal(loid)s[80]. Moreover, strongly reducing soil conditions that favor the formation of organic species can shift contaminant burden toward the endosperm, thereby increasing exposure through consumption of polished rice[50,81,82]. Despite these dynamics, current regulatory frameworks focus on inorganic As, and the bioavailability and health effects of organic As species remain poorly constrained. Similar uncertainties exist for Cd and Hg bioavailability in rice. Together, these processes highlight that metal(loid) accumulation in rice and risk to human health is not controlled by a single pathway, but emerges from the interaction of redox dynamics, rhizosphere chemistry, and plant physiology and that grain localization and post-processing are critical, yet underappreciated, links between biogeochemical processes and human exposure.

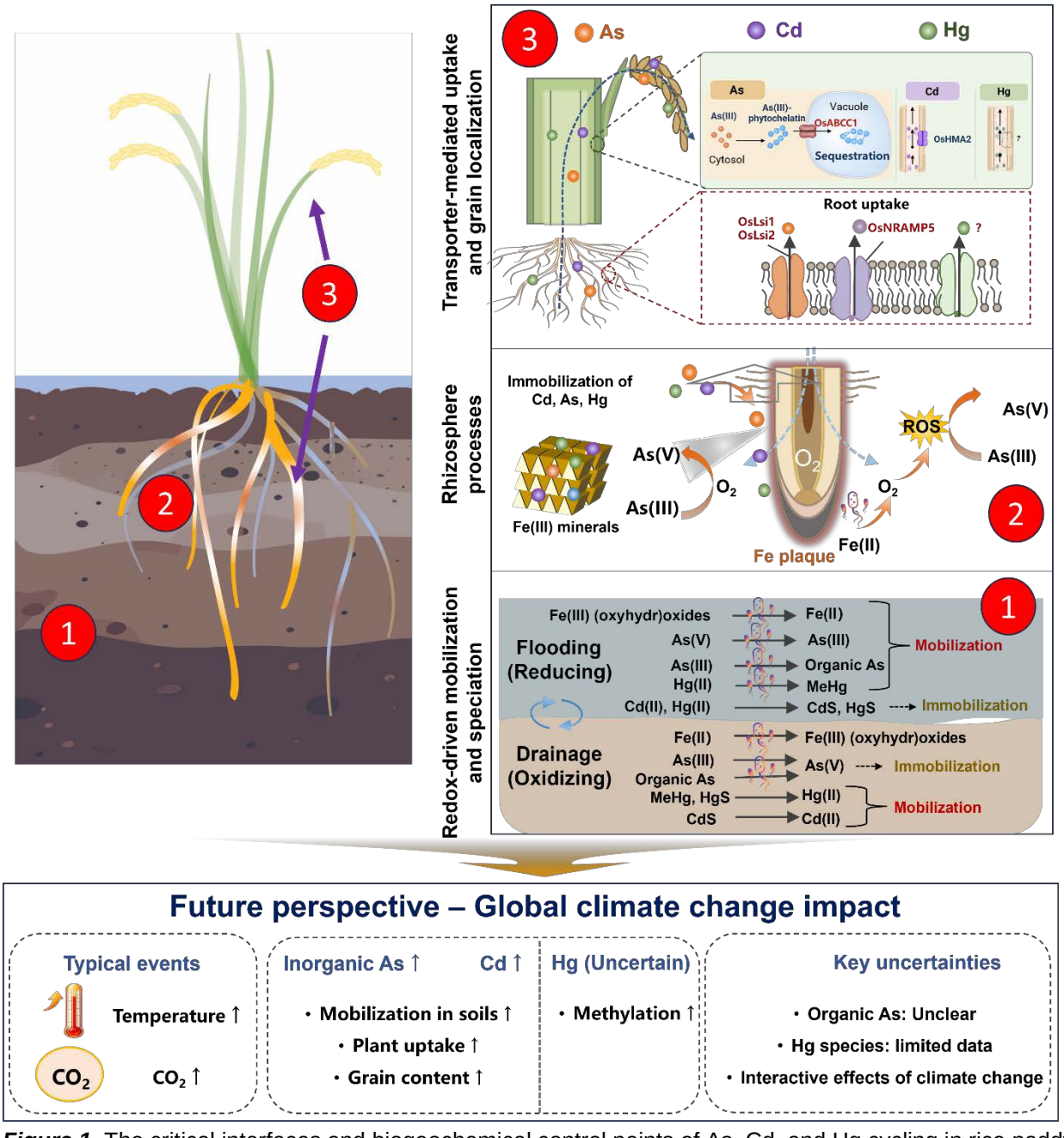


Figure 1. The critical interfaces and biogeochemical control points of As, Cd, and Hg cycling in rice paddy soils. The control points include 1) Redox-driven mobilization and speciation changes in bulk soil, 2) Rhizosphere processes that affect retention, release, and transformation of metal(loid)s, and 3) Plant-mediated controls including transporter uptake into roots, xylem, phloem, and grain. Each of these control points can be targeted for mitigation, but how future climate affects these control-points is just beginning to be explored.

Climate Change as a Perturbation of Biogeochemical Control Points

Future climate conditions will increase inorganic As and often Cd transfer into rice grain, while the picture for Hg is less clear and remains a major research gap[77,83]. Elevated temperature increases As mobilization in flooded paddy soils by enhancing the activity

of Fe and As-reducing microbes that mobilize As from Fe(III) oxide minerals[77,84,85]. Elevated CO_2 can stimulate root and microbial respiration, further driving reduction of Fe(III) oxides and As speciation changes, which in turn increases inorganic but not organic As in grain under future climate scenarios[77,83]. Combined increases in temperature and CO_2 have synergistic effects on inorganic As in rice, with modeled projections indicating higher dietary As exposure and health risks in major rice-consuming countries by mid-century[83]. Although elevated CO_2 may increase plant biomass in rice, a C3 plant, and theoretically dilute metal(loid) concentrations, this effect appears to be outweighed by enhanced soil mobilization and plant uptake driven by increased microbial activity and transpiration fluxes[77]. Meteorological factors such as accumulated temperature, temperature distribution across growth stages, and light duration significantly affect grain Cd when soils are already contaminated; in multi-site experiments, late sowing associated with warmer early growth increased grain Cd 2–5-fold compared with early sowing. Elevated CO_2 and moderately higher temperatures can alter root–shoot distribution of Cd; some seedling studies report changed Cd concentrations and translocation patterns under elevated CO_2, indicating that future climates may modify Cd partitioning even if total uptake does not always rise[86].

Compared with As and Cd, there is little direct experimental work on how elevated temperature, CO_2, and altered water regimes under climate change will affect Hg in rice. Recent work suggests that organic Hg but not inorganic Hg increases under elevated CO_2[87], but at present, we must extrapolate temperature effects from general paddy biogeochemistry. Thus, we expect that warmer temperatures and more reducing, flooded conditions will favor Hg methylation, but how this translates to rice uptake under realistic future climate and management scenarios is still poorly understood. Because of these data gaps, Hg risk projections for rice under climate change remain highly uncertain relative to As and Cd, and this is widely identified as a priority area for further research.

Mitigation Strategies

We emphasize that effective mitigation strategies must be understood as manipulations of the same biogeochemical control points described above, rather than independent interventions. Because climate change alters the same redox, microbial, and transport processes that govern metal(loid) behavior, mitigation strategies must be evaluated not only under current conditions but also under future climate scenarios (Figure 2). The most effective strategies for pre-harvest minimization of exposure risk involve manipulating the biogeochemistry to either enhance toxic metal(loid) retention in soil or compete with them for uptake into the plant[88].

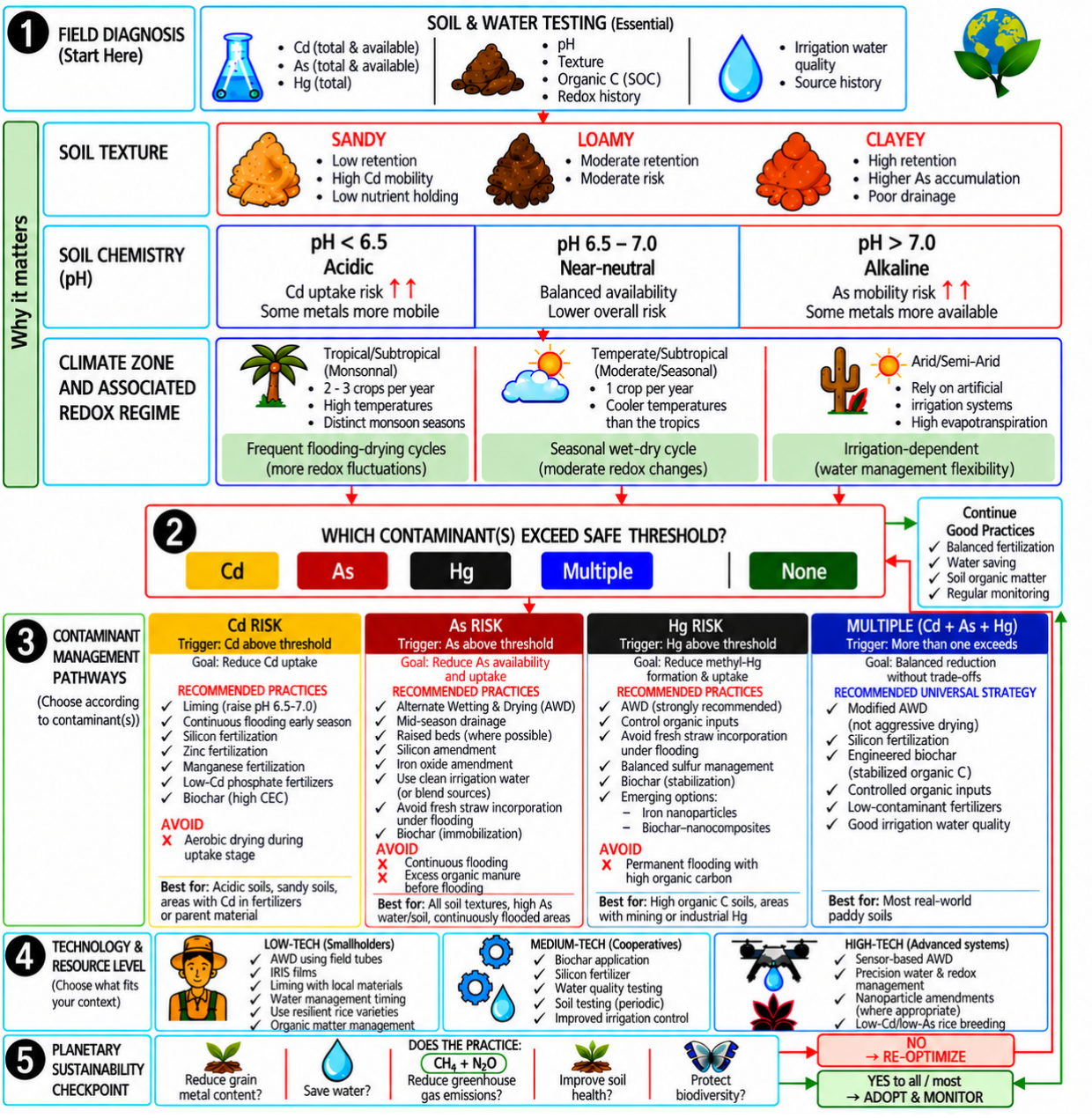


Figure 2. Decision tree for targeted mitigation strategies for Cd, As, and Hg, considering each separately and ensuring resilience in the face of climate change. Rice is grown in different climate zones, including the tropics/subtropics (e.g., India, Thailand, Vietnam, Philippines, Cambodia, Southern China, West Africa, and parts of Brazil and Colombia), subtropic/temperate (e.g., Japan, South Korea, Northern China, Spain, Italy, and the USA), and arid to semi-arid (e.g., Nile delta in Egypt, parts of Pakistan, and expanding across Africa), each of which have different climate pressures and contamination levels as well as infrastructure and technology access for mitigation, thus requiring targeted mitigation. Some areas may have adequate infrastructure for precision high-tech monitoring and mitigation, whereas others may rely on low-tech monitoring, such as field tubes or indicators of reduction in situ (IRIS) films[89] for water level and redox monitoring.

Water management. Because low soil redox conditions strongly increase As and Hg uptake, redox conditions can be increased to lower grain levels through water management, but care must be taken to ensure that Cd levels do not rise to levels of concern. The introduction of one or more drainage cycles during the growing season, termed alternate wetting and drying (AWD) or intermittent flooding (IF), is one of the most effective strategies to lower grain As and Hg concentrations[72,88,90,91]. Mid-season drainage allows oxygen diffusion into soil, oxidation of Fe(II) to Fe(III) oxides, and consequent retention of As via adsorption and co-precipitation, thus lowering As mobility[92]. The timing of the drain is critical, as the conditions of the soil solution or pore water at the time of reproduction are most predictive of grain As concentrations[55,93]. Moreover, aeration also limits the amount of methylated As and Hg by limiting methylation and enhancing demethylation, thus lowering grain levels of total and organic species[90,93–98]. However, implementation of AWD cannot be practiced everywhere, calling for region-specific mitigation, particularly where water is scarce, and yield may be impacted[88]. Furthermore, warming temperatures may necessitate increased frequency in drainage events due to enhanced microbial activity. In areas where row crops rotate with rice, some farmers are planting rice into the existing furrows rather than building levees[99,100], thus creating “row rice” with even more oxic conditions than AWD or IF, which are more effective at lowering grain As and Hg[90,101]. However, because oxic conditions promote Cd mobility, particularly in acidic soils[93,102], this may increase grain Cd to levels that may be higher than regulatory standards in some soils[7,103]. In addition, while AWD is highly effective at minimizing methane emissions, it may increase nitrous oxide emissions, and both are potent greenhouse gases[7,104–107]. Thus, we must ensure that mitigation strategies consider all three of these metal(loid)s and that we do not solve an As or Hg problem by creating a Cd problem, and we must consider impacts on greenhouse gas emissions.

While water management also affects the quantity and mineral composition of Fe plaque[40], Fe plaque should not be viewed as a mitigation strategy. The extent of Fe plaque is often positively associated with grain As, and the extent of plaque formation decreases with water management in the order: flooded > AWD > non-flooded[7,40]. The minerals that comprise Fe plaque retain inorganic As and, to a lesser extent, Cd and Hg, but some may be desorbed by competing ligands and become plant-available, as is the case with phosphorus[108].

Soil amendments. Soil amendments have been widely tested to mitigate toxic metal(loid) accumulation in rice with a primary mode of action of reshaping paddy soil biogeochemistry or transporter competition for uptake[109–112] with varied success[113]. Some soil amendments can immobilize metal(loid)s via complexation and coprecipitation, [114,115] affect microbial activity[115,116] or act as a redox buffer and battery. For example, biochar can suppress microbial Fe(III) reduction and limit As release,[117] or

shift sulfur cycling toward reduction, promoting Cd immobilization as sulfides[118,119], but there has been much less work its effects on Hg. Some amendments, such as nitrate- and sulfur-containing fertilizers and biochar-based materials, were reported to suppress Hg methylation and/or stimulate MeHg demethylation via altering soil redox conditions and regulating Hg-methylating microbial communities such as sulfate- and iron-reducing bacteria[120,121]. The biogeochemical differences in conditions across studies, including different soils, amendment properties, application rates, timing, and edaphic conditions, all contribute to an amendment's efficacy, along with rice variety and microbial activity.

Amendment choice and the target metal(loid) must be considered carefully while balancing tradeoffs. For example, organic-rich amendments can further decrease soil redox potentials, leading to higher As mobilization and organic As (and presumably MeHg) concentrations. However, if Cd is the metal(loid) of concern, then increasing organic matter and raising the soil pH (liming) via amendments will facilitate solid-phase retention and limit uptake[113]. Because many metal(loid)s enter via nutrient transporters, nutrient-based amendments can decrease uptake through competition and regulation of transporter expression[19,122–124]. For example, Mn(II) addition can suppress Cd uptake[65,124], and field-applied MnO_2 decreases grain Cd[125], especially when combined with limestone[65]. Emerging approaches using nanoparticles further regulate transporter activity, cell wall properties, and extracellular redox conditions, providing additional control over contaminant uptake[126–128]. Si addition competes with both inorganic As and DMA for uptake[67,129] and decreases expression of Si transporters[19,130], thus lowering grain As levels. However, Si-rich amendments must provide enough Si to compete for uptake even after Si-induced As mobilization from the solid phase[129,131], and are likely most efficient in well-weathered soils low in plant-available Si.

Targeted mitigation. While research has identified key factors affecting uptake and mitigation of metal(loid)s in rice, we argue that mitigation strategies must be targeted in ways that consider variability in soil-plant biogeochemical factors that affect mobility and plant-uptake. There are more than 10,000 varieties of rice that each respond differently to uptake of metal(loids)s. We cannot breed or genetically-engineer harmful metal(loid)s out of rice without also affecting uptake of key nutrients. Thus, future efforts should focus on growing varieties that minimize toxic metal(loid) accumulation through plant genetics, avoiding pollution, and employing growing practices that minimize metal(loid) mobility and grain storage. However, there is no one-size-fits-all strategy, and not every mitigation strategy will work in every soil or for all toxic elements. For example, soil texture controls oxygen diffusion and water permeability, thereby influencing the reducing conditions that dictate speciation of As and Hg and the mobility of As, Hg, and Cd. There has also been much more research focused on As than on Cd and even less on Hg. While laboratory-based investigations using hydroponics or soil-based pot studies are important, we need more studies that consider how different soils in different

places respond to mitigation efforts and use this data to create models that can identify where mitigation may have the largest impact. Furthermore, rice paddies are becoming more widespread in Africa and South America, with vastly different hydroclimatic and soil conditions[132–134] compared to other well-studied rice-growing regions, making the transferability of concepts uncertain. With new machine-learning tools available, the potential exists to build predictive process-based models with targeted mitigation recommendations, but this requires high-resolution geochemical soil data across geographically diverse locations for accurate model prediction[113,135].

Practical Considerations. Future efforts must consider practical aspects of mitigation, including the cost and type of soil amendment, infrastructure requirements, and impacts on growers[136]. Some soil amendments, such as nanoparticles, may be costly and more experimental, particularly for growers in developing countries. We must also consider how and when the amendment is applied and whether it is feasible. Thus, we argue for more co-creation of mitigation strategies with growers.

Implications for Human Exposure and Policy in a Changing World. We argue that managing human health risks from rice consumption requires moving beyond concentration-based metrics toward a framework that explicitly incorporates speciation, bioaccessibility, and dietary context. Moreover, mitigation should consider how the soil-plant system will respond to and be resilient to future climate scenarios. A recent study showed that AWD is effective at mitigating As under increased CO_2 and temperature[54]. Yet in some areas prone to flooding[137,138], AWD may not be feasible. For Cd, climate-driven changes in temperature patterns and irrigation (including deliberate drying to lower redox) can unintentionally increase Cd bioavailability and grain Cd, especially in already contaminated soils, making management a trade-off between As and Cd risks. For Hg, the directional effects of interacting temperature, CO_2, and hydrology on grain Hg and MeHg are not yet well constrained, so any synopsis must acknowledge substantial uncertainty and the need for targeted experimental and field studies. Taken together, these approaches highlight that mitigation is not a single intervention but a process of managing competing biogeochemical pathways, where reducing risk for one contaminant often requires accepting tradeoffs in another.

Although As, Cd, and Hg in rice are linked to cancer, cardiovascular and kidney disease, and neurodevelopmental impacts, current regulatory approaches remain fragmented and often incomplete. Existing standards primarily target inorganic As, with limits established for specific products such as infant cereals, while Cd regulations are based largely on kidney and bone toxicity thresholds. In contrast, Hg policies focus almost exclusively on fish consumption, despite evidence that rice can be a relevant exposure pathway in certain regions[4,139,140]. As highlighted here, speciation and grain localization strongly influence exposure, yet this is largely unaccounted for in both monitoring and policy frameworks. Similarly, MeHg, although typically present at lower

concentrations than in fish, may contribute meaningfully to dietary exposure in regions with contaminated paddy soils[4,141].

We further emphasize that exposure risk is not uniform across populations. Rice consumption patterns vary widely, and in many rice-dependent cultures—particularly among infants and young children—rice constitutes a major proportion of dietary intake. As a result, even rice that meets existing regulatory standards can pose non-cancer health risks for high-consumption or low-body-weight populations. These disparities highlight an environmental justice dimension, where vulnerable groups may face disproportionately higher exposure due to cultural dietary reliance and limited access to alternative foods.

We therefore argue that future risk management must integrate biogeochemical understanding with policy development. This includes (i) expanding regulatory frameworks to consider metal(loid) speciation and bioavailability, (ii) incorporating variability in dietary exposure and population vulnerability, and (iii) aligning agricultural mitigation strategies with public health goals and the practicality of management strategies. Importantly, climate change adds an additional layer of complexity, as warming temperatures, altered hydrology, and elevated CO_2 are expected to increase mobility and potentially shift metal(loid) speciation, thereby altering exposure pathways, and will impact greenhouse gas emissions.

Together, these challenges underscore the need for a coordinated, systems-level approach that links soil processes, plant physiology, food processing, and human health. Without such integration, efforts to reduce exposure risk may remain incomplete or inadvertently shift risk among toxic elements. We contend that bridging these disciplines is essential for developing effective, equitable, and climate-resilient strategies to protect global populations that rely on rice as a dietary staple and for rice consumers.

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Notes

The authors declare no competing financial interest.

Biographies



Angelia Seyfferth is a Professor of Soil Biogeochemistry in the School of Life Sciences at Arizona State University who has been studying contaminant transport to edible foods for more than 20 years. Most of her research involves understanding trace

metal(loid) uptake into rice and mitigation strategies. She has received the Faculty Early Career Development (CAREER) Award from the National Science Foundation, the SSSA Marion L. and Chrystie M. Jackson Soil Science Award by the Soil Science Society of America for outstanding contributions in the areas of soil chemistry and mineralogy, and the Charles S. Falkenberg Union Award co-sponsored by the American Geophysical Union and the Earth Science Information Partners for her contribution to the quality of life, economic opportunities, and stewardship of the planet through the use of Earth science information and to the public awareness of the importance of understanding our planet.



E. Marie Muehe is a Professor at the Helmholtz Centre for Environmental Research UFZ and the University of Tübingen in Germany. Her and her teams research focus on climate-contaminant interactions in cropping systems and urban settings, with a strong focus on metal/loids. Current works expand to mitigation and management strategies under future climates. Muehe holds an Early Career Helmholtz Young Investigator Grant and is part of the Cluster of Excellence (EXC 3121): TERRA – Terrestrial Geo-Biosphere Interactions in a Changing World, University of Tübingen, Germany.



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