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Review article

Safety of African grown rice: Comparative review of As, Cd, and Pb contamination in African rice and paddy fields

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A R T I C L E I N F O

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A B S T R A C T

This review aimed to investigate the reported concentrations of arsenic (As), cadmium (Cd), and lead (Pb) in rice cultivated in Africa and African rice paddies compared to other regions. It also aimed to explore the factors influencing these concentrations and evaluate the associated health risks of elevated As, Cd, and Pb exposure. Relevant data were obtained from electronic databases such as PubMed, Scopus, and Google Scholar using specific keywords related to arsenic, cadmium, lead, rice, Africa, paddy, and grain. While the number of studies reporting the concentrations of As, Cd, and Pb in rice and rice paddies in Africa is relatively low compared to other regions, this review revealed that most of the African rice and paddy soils have low concentrations of these metals. However, some studies have reported elevated concentrations of As, Cd, and Pb in paddy fields, which is concerning due to the increased use of agrochemicals containing heavy metals in rice production.

Nonetheless, agronomical interventions such as implementing alternate wetting and drying water management, cultivating cultivars with low accumulation of As, Cd, and Pb, amending rice fields with sorbents, and screening irrigation water can limit the bioaccumulation of these carcinogens in paddy fields using phytoremediation techniques. Therefore, we strongly urge African governments and organizations operating in Africa to enhance the capacity of rice farmers and extension officers in adopting approaches and practices that reduce the accumulation of these carcinogenic metals in rice. This is essential to achieve the sustainable development goal of providing safe food for all.

1. Introduction

Rice (Oryza sativa L.), being a major staple food for more than half of the world's population, with over 3.5 billion people consuming rice as a primary source of calories [1–3], is cultivated on almost 159 million hectares across the globe [4]. It plays a crucial role in the diets of Asians, Latin Americans and Africans as it provides a substantial portion of dietary calorie intake (>35–60%) [5,6]. Given its importance and widespread consumption, it is essential to assess the potential risks associated with rice contamination.

In the Sub-Saharan Africa (SSA), Malawi inclusive, demand for and cultivation of rice has considerably and more rapidly increased compared to any other continent since 1995 [7]. In Malawi, rice is the second most important cereal crop after maize, and third food

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crop after maize and cassava [8]. With the growing cultivation of rice in Africa, it becomes necessary to investigate the quality and safety of the produced rice, including the presence of contaminants.

In addition to being a rich source of dietary calories, rice is also a rich source of several essential nutrients such as proteins, selenium, and Vitamin B12 [5,6]. Understanding the nutritional benefits of rice is crucial for promoting rice as a health-promoting food for humans [6,9].

Next to the cited benefits cited, rice plants have the ability to efficiently uptake and accumulate measurable quantities of carcinogens such as arsenic (As), cadmium (Cd) and lead (Pb) from soil [10–13]. Compared to other cereal crops such maize and wheat, rice has been found to accumulate these carcinogens in greater quantities [11,13]. Considering that these contaminants are class-1 non-threshold carcinogens, their bioaccumulation in rice poses a serious health risk to rice consumers [14–16]. For instance, consumption of As-rich rice has been linked to various health issues, including skin cancer, skin lesions, and pigmentation changes [17–21]. On the other hand, chronic exposure to elevated Cd levels can lead to detrimental effects on the central nervous system, kidney dysfunction, bone fractures, and lung cancers [21,22]. Pb, being a potent neurotoxin, can cause developmental delays, behavioural changes, and damage to the nervous system, especially in children. Therefore, assessing and understanding these health risks is essential for public health and safety [17,23–27].

The extent of health risks resulting from consuming contaminated rice depends on several factors, such as the concentration of contaminants, duration of exposure, age, and health status of consumers, frequency and amount of contaminated rice consumed, presence of other dietary factors, individual genetic variations, and lifestyle choices [28–30]. Understanding and considering these factors in rice cultivation is crucial for assessing and managing the risks associated with rice contamination.

Paddy fields in Africa face rapid contamination by pollutants such as heavy metals, pesticides, inorganic fertilizers and industrial waste [31–33] which threatens rice farming. The levels of As, Cd, and Pb in rice and rice paddies can vary depending on geographical location [34,35], environmental factors [34,35], agricultural practices [15,36–39], industrial waste effluents [25,40], and limited data and monitoring systems [41]. Studies in Ghana and Nigeria revealed high levels of Cd and Pb in rice samples from paddy fields due to industrial pollution [7,33,42,43]. Pesticide residues were also found in Senegal and Tanzania, affecting consumer health [40,44]. Industrial waste, due to improper management, pollutes paddy fields, as shown in a study by in Cameroon. Additionally, climate change worsens the situation, impacting rice production and increasing the risk of contamination from pollutants, as observed in Nigeria [31,33]. Therefore, comparing the reported contaminant concentrations in African rice and paddy fields with other regions provides valuable insights into the extent of the issue and the need for remediation strategies. Given the significance of rice as a staple food, the potential health risks associated with rice contamination, the lack of sufficient studies on the extent of contamination in African rice, and the need for remediation strategies, a comprehensive review on the topic is justified to provide valuable insights and contribute to addressing this global issue.

2. Legislations restricting contaminated rice on market

In view of the gravity of health risks associated with As, Cd and Pb, different regulatory agencies like World Health Organization (WHO) [34,35], Food and Agriculture Organization (FAO) [34,35], European Commission (EU) [45], United States Environmental Protection Agency (USEPA) [10,34,35] and Codex Alimentarius Commission [46] legislated Maximum permissible limits of As, Cd and Pb in various food staff including rice and rice products (Table 1). For As, the European Commission (EU) [45] legislated maximum permissible limit (MPL) of 250 μg kg^{-1} for iAs in husked rice (unpolished rice), 200 μg kg^{-1} for iAs in polished rice; and 100 μg kg^{-1} for iAs in rice destined for baby food production, which was effected in January 2016 [45]. For Cd, the European Commission also regulated MPL for total Cd at 200 μg kg^{-1} for any rice on the European Market [45]. The primary aim of The European Commission

Table 1
Maximum Allowable limit (MCL) for total arsenic, Cd and Pb (μg kg^{-1}) in rice grains as reported by various studies.

Organization	Description	iAs	tAs	Cd	Pb	Refs.
European Commission legislation	polished rice)	200	–	200	100	[26,51]
	unpolished rice	250	–	400	200	[26,51]
WHO/FAO	Polished rice	–	300	400	200	[52]
Turkish Bureau of Standards (TBS)	rice	–	100	100	100	[2,21,53]
Southern America Trading Block	rice	–	150	400	100	[53–56]
		–	250	700	200	
Chile Bureau of Standards (CBS)	Polished rice	–	200	400	100	[57,55,58]
Hungary Bureau of Standards (HBS)	Polished rice	–	200	200	100	[55,59]
	Unpolished rice	–	300	400	200	
China	Polished rice	–	150	200	100	[53,55]
	Unpolished rice	–	200	400	200	[34,35]
Ghana government	Raw rice		100	400	200	[34,35]
	Parboiled rice		150	500	300	[34,35]
Vietnamese government	Rice		200	200	200	[60]
Bangladesh government	Rice		500	200	2000	[61]
Nigeria	Rice			300	500	[62]
Egypt	Rice		200	200	300	[34,35,63]
RSA	Polished rice		200	400	200	[34,35]
	Husked polished		300	600	300	[34,35]

legislation is to protect the general population, infants, and young children from ingesting elevated carcinogens through rice and rice products consumptions by restricting importation of rice and rice products that violates the legislated MPL into EU market area [45]. Besides the European Commission, other regulatory bodies have also set regional, or country based MPL (Table 1). For instance, Codex Alimentarius Commission [26] regulated maximum contaminant limit (MCL) of 300 μg kg^{-1} for tAs in brown rice, 200 μg kg^{-1} for iAs in polished rice, 300 μg kg^{-1} for Cd polished rice and 300 μg kg^{-1} for Pb in rice for the general population (Table 1). To check compliance on the legislation, Signes-pastor et al. [47,48] out a survey and reported non-compliant batches of rice and rice products destined for to produce baby and infant foods being marketed in Europe which indicated non-compliance by rice exporters. It was speculated that this could be because there is inadequate safe rice available from major world rice exporters that meets the legislated MCLs (Table 1). It could also because most of the major rice producers lack equipment to screen contaminated rice. In view of this, toxicological studies of rice have emerged as a global health issue requiring collective scientific efforts and approaches to towards developing effective techniques to reduce As, Cd and Pb accumulation in rice in order to minimize the risks associated with consumption of rice contaminated with As, Cd and Pb [49,50].

3. Comparisons of As, Cd and Pb content in paddy soils across geographical regions

3.1. *Arsenic*

Arsenic (As) can exist in both organic and inorganic forms, with the inorganic arsenic (iAs) species being significantly more toxic than the organic As (oAs) species [10,64]. Inorganic arsenic occurs in the forms of arsenites, arsenates, and arsines, among which arsenites are more toxic than arsenates [10,64]. Among these iAs species, arsines are the most toxic [10,64]. Arsenic can be mobilized and introduced into rice paddies through various mechanisms, such as the utilization of contaminated irrigation water, the application of contaminated pesticides and chemical fertilizers, extreme weather conditions, and microbial activities within paddy soils [65,66]. Contaminated irrigation groundwater can transport As to paddy fields, leading to the subsequent contamination of rice crops with As [65,67]. Additionally, the use of phosphate fertilizers, derived from rocks that contain elevated levels of As, can significantly contribute to increased concentrations of soil As, eventually impacting the As content in rice grains [68].

Similarly, specific soil microorganisms have the ability to convert iAs into oAs forms through a process known as As-methylation [12,69]. These organic forms of arsenic are readily taken up by rice plants, leading to the potential bioaccumulation of elevated levels of arsenic in rice [12,69]. Furthermore, the presence of As in paddy soils can also be influenced by the mineral composition of the underlying rocks in the fields [70]. Rocks containing sulphides and arsenate minerals pose an increased risk of contaminating paddy soils with elevated levels of arsenic. For example, in the Nile Delta region of Egypt, rice paddies have been found to exhibit high arsenic levels, which have been attributed to the erosion of shale rocks containing elevated concentrations of arsenates and sulphides [70,71].

It is well established in scientific literature that extreme weather conditions, particularly heavy rainfall and temperature

Table 2
Arsenic and Cadmium concentrations in rice grains from various countries (μg kg^{-1}).

Country	tAs	Refs	Cd	Refs
Australia	30 (20–40)	[82]	7.5 (9–17)	[17]
Asian countries				
Bangladesh	130 (20–330)	[16]	73 (69–77)	[17]
Bangladesh	(5–2050)	[16,54]	73 (69–77)	[17]
China	140 (20–460)	[54]	34.5 (4–70)	[17]
China (mines)	150	[16,53]		
India (1)	92 (90–94)	[44]	18–20	[44]
India (2)	(180–310)	[54]	27.5 (9–78)	[83]
India (3)	50 (30–80)		19.1 (<1–36)	[17]
Japan	102 (100–104)	[44]	4 (4–5)	[44]
Thailand	140	[16]	37 (23–68)	[17]
Thailand	131 (130 134)	[44]	10 (9–11)	[44]
Taiwan	510 (100–630)	[54]		
EUROPE				
France	280 (90–560)	[16,53]		
Italy	150 (70–330)	[54]	317 (44–841)	[17]
Japan	190 (25–487)	[84]	13–33	[34,85,86]
Spain	200 (50–820)			
Spain	160–180	[16]	31	[34]
Africa				
Malawi	20–60	[77,87]	6 (<2–39)	[77]
Senegal	111 (60–161)	[44]	13 (3–58)	[44]
Ghana	90 (5–206)	[88]		
Ghana	110	[34]		
Togo	64 (20–400)			
Egypt	97	[34]	3–5	[34]
Egypt	50 (10–580)	[53]	<4	[83]
America				
USA	260 (110–400)	[16]	62 (4–123)	[17]

fluctuations, have a significant impact on the mobilization of As in rice paddies. For example, periods of high rainfall can lead to the increased downstream movement of As-contaminated water from mining and industrial sites, ultimately reaching water sources and, in turn, entering paddy fields through irrigation [72–74]. Conversely, As may accumulate in the soil and remain in a less mobile form in areas with low rainfall thereby increasing soil-As burden [72,74,75]. For temperature, high temperatures may promote the growth of microorganisms that may release As from soil particles thereby making it readily available for plant uptake thereby increasing soil-As burden as well [73–75].

Soil-As concentrations has been reported in many countries across the globe. Mandal et al. [55] and Kabata-Pendias et al. [10] reported As concentration ranging from 4800 to 6000 μg kg^{-1} across the globe. Mean arsenic concentrations of 11.6 μg kg^{-1} (range: 0.1–2500 μg kg^{-1}), 6.4 μg kg^{-1} (0.1–1300 μg kg^{-1}), 35.9 μg kg^{-1} (0.1–2900 μg kg^{-1}), 14.3 μg kg-ˡ (range: 0.1–710 μg kg-ˡ) and 36.4 μg kg^{-1} (range 0.1–540 μg kg^{-1}) for Argentina, Brazil, Chile, Mexico and Peru, respectively [10,40,55,64].

For African countries, Mlangeni et al. [68,76][68 and Joy et al. [77] reported mean soil-As concentrations of 3500 μg kg^{-1} (range: 3200–3700 μg kg^{-1}) and 2650 μg kg^{-1} (range 3200–3600 μg kg^{-1}) respectively for Malawi; whereas Mandal and Suzuki [55] and Mawussi et al. [78] reported mean soil-As concentrations of 3200 μg kg^{-1} (range: 3000–3800) and 1040 μg kg^{-1} (range: 400 μg kg^{-1} - 2900 μg kg^{-1}), respectively, for crop fields of South Africa. Nkansah et al. [79] reported mean soil-As concentration of 2000 μg kg^{-1}) (range: 250–7570 μg kg^{-1}) (Table 2) for paddy fields of Ghana whereas Nougbode et al. [80] reported mean soil-As concentration of 440 μg kg^{-1} (range: 80–1330 μg kg^{-1}) for paddy fields of Benin. These results indicated that some paddy fields are contaminated with As, with the mean soil-As concentration exceeding MCL of 1500 μg kg^{-1} legislated by the European commission [81]. Compared to soil-As concentrations reported in other regions, soil-As concentrations mostly reported in literature for paddy fields of Africa are low. However, more research is needed to reaffirm low soil-As content for African rice paddies.

3.2. Cadmium

Cadmium, Cd, is a toxic heavy metal that may contaminate paddy fields, leading to serious health problems in humans who consume rice grown in these fields. It is widespread and persistent contaminant in soil and water. The concentration of Cd usually ranges from 10 to 2000 μg kg^{-1} (median: 350 μg kg^{-1}) [22] in natural soil. Cadmium with oxidation state of +2 (Cd^{2+}) [89] is highly mobile through soil layers [22,28,89], highly Phyto-available for plant uptake in soils [22], and highly toxic to plants, humans and animals even at a very low concentration level [16,24,90]. Hence Cd^{2+} is mostly studied state of Cd. Cd may be mobilized into paddy fields through various mechanisms such as irrigation water, atmospheric deposition, and agricultural practices including use of chemical fertilisers and pesticides [91–94]. For impact of soil characteristics such as soil pH, texture, and organic matter content, Cd bioavailability tends to be higher in acidic soils than alkaline soils and tends to bioaccumulate in soils with high clay and organic matter content than those without which increase Cd availability in soi [1,95,96]. For impact of irrigation water, Cd may be introduced into rice paddies through contaminated irrigation water more especially in areas with high levels of industrial activity, where wastewater containing Cd is discharged into bodies that are used for irrigation [11]. For impact of atmospheric deposition, Cd may also be deposited onto rice paddies through atmospheric deposition. This may occur in areas with high levels of atmospheric Cd emissions from industrial activities, transportation, and other sources. For agricultural practices, some agricultural practices such as application of phosphate fertilizers may increase Cd availability in soils, as Cd may bind to phosphate and become more mobile. Additionally, the burning of rice straw within the rice fields may release Cd back into the soil thereby increasing its availability to rice plants. For soil microbial activity: some microbial processes may alter soil pH and redox conditions, which may remobilise Cd into the soil and eventually influence availability of Cd to rice plants [17,97].

The presence of Cd in soil and soil solution affects plant uptake of essential elements due to chemical similarity between most essential elements with Cd [90] as well as due to competition for the same cellular transporters [90] between Cd and essential elements. For instance, Zn^{2+} may inhibit Cd^{2+} transport within rice because both Zn^{2+} and Cd^{2+} are up-taken through the same transporters. Cd is mobilized into environment through either geogenic or anthropogenic activities or both [98]. It is reported that mining of zinc-bearing ores with Cd as a byproduct is the major anthropogenic activity releasing measurable Cd quantities into underground water, river water and agricultural soil [99].

Soil-Cd concentrations has been reported across the globe. For Sub-Saharan Africa, mean soil-Cd concentrations in soils of Malawi (mean: 30 μg kg^{-1}; range: 10–110 μg kg^{-1}), Tanzania (Morogoro region) (mean: 40 μg kg^{-1}; range: 10–140 μg kg^{-1}) and Namibia (mean 40 μg kg^{-1}; range: 20–70 μg kg^{-1}) were lower than the legislated the maximum contaminant limit (MCL) for Cd in soil in the United States of 75 μg kg^{-1} [7,77,87,100–104]. However, Cd concentrations reported in rice paddies of Limpopo River Basin in South Africa (mean: 110 μg kg^{-1}; range: 30–270 μg kg^{-1}), Zambia (mean:110 μg kg^{-1}; range: 10–300 μg kg^{-1}), Swaziland (mean: 120 μg kg^{-1}; range: 20–260 μg kg^{-1}); Katanga province in DRC (mean: 280 μg kg^{-1}; range: 10–770 μg kg^{-1}), Mozambique (mean: 90 μg kg^{-1}, range: 10–220 μg kg^{-1}) and Zimbabwe (mean = 120 μg kg^{-1}; range: 10–340 μg kg^{-1}) [7,77,87,100–104] were higher than legislated MCL of 75 μg kg^{-1} for Cd in soils of the United States. Compared to other regions, the reported mean soil-Cd concentrations in Sub-Saharan Africa are generally lower than most countries in Europe, Asia, and Indian subcontinent. For example, studies conducted in Southern America, have reported mean soil-Cd concentrations of 130 μg kg^{-1} (range: 20–380 μg kg^{-1}), 90 μg kg^{-1} (range: 30–220 μg kg^{-1}), 170 μg kg^{-1} (range: 20–480 μg kg^{-1}), 160 μg kg^{-1} (range of 30–510 μg kg^{-1}), 190 μg kg^{-1} (range of 50–670 μg kg^{-1}) and 340 μg kg^{-1} (range: 150–600 μg kg^{-1}) in rice paddies of Entre Ríos province of Argentina, São Paulo state of Brazil, northern coastal region of Peru, central valley of Chile, Cochabamba region of Bolivia and Morelos state of Mexico respectively [105–107].

3.3. Lead

Lead, Pb, is a second most hazardous substance preceding As on the substance priority list [28]. Pb is highly toxic and poisonous heavy metal to humans even at a very low concentration at which it is not phytotoxic [34,108]. It may cause neurological, developmental, and reproductive problems in humans and wildlife [109]. It is one of four metals with most damaging effects on human health [109]. Pb concentrations found in the environment result mostly results from human activities [93] such as mining and ammunition discharges. For mining, elevated concentrations of Pb have been reported in and around abandoned and active mines which is associated to discharge and dispersion of mine wastes into surrounding environment [93]. For ammunition discharges, elevated Pb concentrations have also been reported in soils of various shooting ranges worldwide [110]. Pb may enter the human body through the food chain [111,112]. Food such as fruits, rice, wheat and seafood may contain elevated amounts of lead [112,113]. Pb concentration in plants positively and highly correlates with Pb content in soil indicating that higher soil to plant-metal transfer. Just like As and Cd, Pb uptake and accumulation in rice plants is markedly influenced by cultivar genotype of rice [34]. Pb uptake and accumulation in sensitive rice plant cultivars is lower than in tolerant cultivars; hence sensitive cultivars could be encouraged for cultivation in Pb affected areas. Lead may be mobilized into rice paddies through various human activities and natural processes such as mining and smelting, industrial processes, transportation, construction and demolition of old buildings, natural weathering, irrigation water, atmospheric deposition, and application of contaminated fertilizers. For mining and smelting, the processes of heating lead containing ore at high temperatures during the smelting and extraction release lead into the air and water. For industrial processes, Various industrial processes, such as manufacturing, battery production, and paint production also release lead into the air and water. For transportation, vehicles that use leaded gasoline release lead into the air. However, leaded gasoline has been banned in many countries due to health concerns. For construction and demolition, lead may be release from lead-based paints and pipes used in older buildings during construction and demolition which may contaminate rice paddies. For natural weathering, small amounts of lead may be released into the environment from natural weathering of rocks and soil. Once lead is mobilized into the environment, it may be transported through air and water, and may accumulate in soils of rice paddies. For irrigation, irrigating rice paddies using water sources contaminated with Pb one of the main pathways through which Pb enters rice paddies. For atmospheric deposition, Pb in the atmosphere may settle on the soil particularly in areas where rice paddies are close to industrial activities or high traffic. Pb deposition may be influenced by wind direction and water runoff after heavy rainfall. For chemical fertilizers, application of fertilizers contaminated with Pb increase Pb load in rice paddies.

Soil-Pb concentrations has been reported in rice paddies across the globe. Mean soil-Pb concentration of 18 200 μg kg^{-1}, (range = 1140–224 000 μg kg^{-1}) [33,77,80,87,91,114], 33 500 μg kg^{-1} (range: 2200–318 400 μg kg^{-1}), 23 006 μg kg^{-1}, (range = 5300–73 900 μg kg^{-1}) and 14 000 μg kg^{-1} (range = 1000–52 000 μg kg^{-1}) have been reported in rice paddies of China, Thailand, Italy, and Nigeria, respectively. Comparing the reported concentrations with other studies from around the globe, it may be said that these levels are among the highest reported in the literature. For example, according to a review article by Alloway [115], the average soil-Pb concentration in urban areas of developed countries is around 200–400 μg kg^{-1}, while in developing countries, the average concentration is around 500–1000 μg kg^{-1}. Therefore, the soil-Pb concentrations reported in China, Thailand, Italy, and Nigeria are significantly higher than the average concentrations reported for both developed and developing countries. This suggests that the areas where these studies were conducted are likely to be heavily polluted with lead and pose a significant risk to human health and the environment.

4. Geographical variation of As, Cd and Pb accumulation in rice grains

4.1. Arsenic accumulation in rice

Various studies have reported elevated As concentrations in rice from certain geographical regions compared to other [53,63,116,117]. For instance, elevated As concentrations have been reported in regions with known historic use of arsenical pesticides such as the USA [63,118–120] and regions with prolonged use of contaminated tube-well groundwater such as Bangladesh, West Bengal (India), Vietnam, Thailand, Nepal and Taiwan [43,53,63,67,118,121–123]. Norton et al. [124] and Sun et al. [53,125] reported higher mean As concentration amounting to 6560 ± 2500 μg kg^{-1} in rice irrigated with contaminated groundwater with As amounting 602 ± 314 μg/L. Furthermore, elevated dimethylarsinic acid (DMA) reported in rice from the United States of America (USA) was linked to historic extensive use of methylated arsenical pesticides in the then cotton fields and dipping tanks [126] which increased As content in top agricultural soils. A study by Abdel-Rahman et al. [127], reported mean arsenic concentration of 180 μg kg^{-1} (range: 40–680 μg kg^{-1} in rice from Al-Hasa Oasis region of Saudi Arabia. The mean As concentration observed in this region is above the limit of 100 μg kg^{-1} recommended by the WHO. The elevated As concentration was linked to contaminated irrigation water whose arsenic concentration was higher (360 μg L^{-1}) 'compared to the limit of 10 μg kg^{-1} for drinking water recommended by WHO. Rahman and Hasegawa [63] also reported higher mean -As concentration of 150 μg kg^{-1} (range: 90–560 μg kg^{-1}) in rice from EU. The concentration also exceeded the legislated MCL recommended by both WHO and EC. Reports for African grown rice are limited, but available data suggests generally low grain-As concentrations in most African countries [77,87,114] with exceptions. Asuming-Brempong [121] reported low mean As concentrations of 76 μg kg^{-1} (range: 23–187 μg kg^{-1}) in rice from Ghana, while Abia et al. reported lower mean As concentration in rice of 62 μg kg^{-1} (range: 23–124 μg kg^{-1}) from the Abakaliki region of Southeast Nigeria [128]. Mlangeni et al. [87] reported a low mean grain-tAs concentration of 31 ± 12 μg kg^{-1} (range: 9–54 μg kg^{-1}, n = 35) for brown rice from Malawi. Meharg et al. [53] and Naggar et al. [129] reported low mean-tAs concentrations of 50 μg kg^{-1} (range: 10–58 μg kg^{-1}, n = 110) for rice from Egypt which was attributed to quality irrigation river water. Farooq et al. [130] reported mean As concentration of 52 μg kg^{-1} (range: 26–113 μg kg^{-1}) in rice from Mozambique. Ncube et al. (2015) reported mean As concentration in rice (52 μg kg^{-1}, range:

26–113 μg kg^{-1}) in South Africa. These concentrations are within the WHO recommended limit, indicating that the rice from these countries is relatively safe for consumption. However, it would be beneficial to compare these results with other studies conducted in these countries to get a broader understanding of the contamination levels.

Conversely, Nguyen et al. [60] reported mean As concentration in rice of 120 μg kg^{-1} (range: 40–1260 μg kg^{-1}) in north-western Vietnam. Jahiruddin et al. reported mean As concentration in rice of 153 ± 112 μg kg^{-1} and 140 ± 80 μg kg^{-1} in irrigated and rain-fed, respectively, rice from Bangladesh. These results exceed the EC and WHO limits of 100 μg kg^{-1}. Farooq et al. [130] reported mean arsenic concentration of 128 μg kg^{-1} (range: 50–330 μg kg^{-1}) in rice from southern African countries. Generally low but with some samples exceeding EU and CAC limits, posing potential health risks. Continuous monitoring of As and Pb levels in rice and other food products is necessary for food safety and public health protection.

4.2. Lead (Pb) accumulation in rice

Various studies have also reported that elevated Pb bioaccumulation in rice of certain geographical regions compared to other regions [27,53,63,116,117]. For instance, Norton et al. [27] conducted an analysis of total Pb concentration in market white rice from 13 countries across 4 continents (Table 2). The study reported the highest mean Pb concentrations of 185 μg kg^{-1} in rice from China mines impacted zones, while the lowest mean Pb concentrations of 3 μg kg^{-1} was reported for Japan [27]. This highlights a significant variation in Pb levels between the two regions, with China's mines impacted zones demonstrating considerably higher contamination. Gunduz et al. [2] also conducted a similar study, reported the highest mean Pb concentrations of 2230 μg kg^{-1} in Iranian market white rice (Table 2). This again emphasizes a substantial disparity in Pb contamination levels when compared to other regions. In contrast, Joy et al. [77] reported a relatively low mean Pb concentrations of 6.0 μg kg^{-1} for Malawi white rice (Table 2). This suggests a lower level of Pb contamination in rice from Malawi compared to other regions.

Furthermore, Shraim's study highlighted varying levels of Pb in rice grains collected from multiple countries. This included high levels of Pb in rice from the USA (4–98 μg kg^{-1}), Egypt (15–31 μg kg^{-1}), Pakistan (18–22 μg kg^{-1}), India (3–218 μg kg^{-1}), and Thailand (12–20 μg kg^{-1}) [131]. These findings demonstrate the variability in Pb contamination levels among different countries. Considering the MCL of Pb, it is essential to note that specific regulatory standards and guidelines may differ among countries. However, the maximum Pb concentration in rice recommended by WHO/FAO is 300 μg kg^{-1} [26,51]. The MCL for Pb in rice is typically set to ensure consumer safety and minimize potential health risks. Based on the results provided, it is apparent that certain geographical regions exhibit higher Pb concentrations in rice, while others demonstrate lower levels. These variations in Pb bioaccumulation in rice can be attributed to various factors, including environmental pollution, agricultural practices, soil contamination, and industrial activities [131,132]. Understanding these regional disparities is crucial for implementing effective mitigation strategies and monitoring programs to ensure food safety and minimize human exposure to elevated levels of Pb in rice.

4.3. Cadmium (Cd) accumulation in rice

Cd concentrations in unpolished rice from various countries including Bangladesh [17], Australia [17], Egypt [99], China [17,83], and Malawi [77] are reported in Table 2 which significantly vary with region. For instance, low Cd concentrations have been reported in rice from Egypt (<3.7 μg kg^{-1}), Malawi (mean = 6 μg kg^{-1}; range: <2.0–39 μg kg^{-1}), USA (mean = 6.2 μg kg^{-1}; range = 0.4–12.3 μg kg^{-1}) and Australia (mean = 7.5 μg kg^{-1}; range = 8.7–17.1 μg kg^{-1}) with Cd concentration from Egypt and Malawi being among the lowest ever reported grain-Cd concentration. On the contrary, higher grain-Cd concentrations have been reported in rice from Italy (mean = 317 μg kg^{-1}; range = 44–841 μg kg^{-1}), Bangladesh (mean = 73 μg kg^{-1}; range = 69–77 μg kg^{-1}), India (mean = 27.5 μg kg^{-1}; range = 9.4–78.0 μg kg^{-1}), and China (mean = 34.5 μg kg^{-1}; range = 3.6–69.7 μg kg^{-1}); the highest Cd concentration being in rice from Italy. However, it is important to note that the Cd concentrations in rice from most countries, as presented in Table 2 are below the Chinese National Food Guideline Limits (National Food Hygiene Standard of China) of 200 μg kg^{-1} and MCL of 400 μg kg^{-1} set by the World Health Organization (WHO) and the Food and Agriculture Organization (FAO) (Table 1). Thus, in the light of the MCL for Cd content in rice set by the FAO, WHO, and the European Commission (EC) (Table 1), it appears that the Cd concentrations reported above generally meet the regulatory limits. However, it is important to consider that Cd, being a toxic heavy metal, can have harmful effects on human health over the long term even low concentrations. Therefore, continuous monitoring and efforts to minimize Cd contamination in rice and other food sources are necessary to ensure food safety and protect public health.

5. Factors affecting As, Cd and Pb accumulation in rice grain: comparative analyses

Elevated accumulation of heavy metals in rice and rice paddies has been associated with paddy contamination through use and application of chemical fertilizers and pesticides contaminated with heavy metals [53,93,133]; floodplain downstream movement of industrial wastes effluents into rice paddies [53,93,133]; changes in rainfall patterns, temperature, and extreme weather events [134]; and limited monitoring systems of heavy metal contaminations in rice paddies. Elevated accumulation of heavy metals in rice and rice paddies has also been associated with certain geographical regions [53,135,136], irrigation with contaminated groundwater [63,105,137], high metal accumulating rice cultivars (genotype) [67,120,138–140], certain soil types [11,23,72,134,139], specific ranges of soil-pH [77,96,140–142] and available soil sorbents in soils [9]. Therefore, the following sections discuss how these factors have influenced mobilization and bioaccumulation of As, Cd and Pb accumulation in rice grown and cultivated in Africa compared to those from other global regions.

5.1. Impact of pollutants

Irrigation of rice paddies with water contaminated with As, Cd, and Pb results in hazardous pollution of topsoil, leading to increased accumulation of these elements in rice grains and compromising the quality of rice [55,143]. The quality of irrigation water plays a significant role in the loading of As in rice [64,143]. Arsenic concentrations in unpolluted irrigation water typically range from 0.1 to 0.8 mg As L^{-1}, while polluted and heavily polluted water can have concentrations exceeding 100 mg As/L and 1000 mg As/L, respectively [64,143].]. In countries like Bangladesh and Thailand, well water can contain As levels ranging from 10 to 1000 mg As/L [64,143–145].

A study conducted by Meharg et al. [65] revealed that frequent irrigation with groundwater can lead to As concentrations of 20–30 mg kg^{-1} in soils. The authors reported that groundwater adds As to the topsoil, which can be easily released into the water-soil system, resulting in elevated concentrations [65]. They concluded that the primary cause of arsenic accumulation in rice crops [65] in Bangladesh was the utilization of As-contaminated irrigation water [65]. The accumulation of As in rice grains strongly correlates with the soil As concentrations, and it relies on both the irrigation system utilizing arsenic-contaminated groundwater the continuous loading of arsenic in the topsoil [65].

Majumder and Banik [143] reported that elevated As levels in irrigation water lead to an increase in As loading in paddy fields in Bangladesh and West Bengal. However, it was observed that the source of irrigation water can vary, including groundwater from tube wells, river and/or lake water, and rainwater, with variations in acidity/alkalinity and As content [143]. The pH of rainwater affects the solubility and dissolution of heavy metals such as Cd and Pb in soils [143]. Therefore, countries receiving acidic rain may have higher available soil-As and Cd concentrations for plant uptake. Rainwater can also transport contaminants downstream to rice paddies, increasing their loading in the fields and subsequently in rice grains. The prolonged use of tube well water contaminated with As significantly contributes to the As burden in paddy fields and the bioaccumulation in rice grains [143,146]. Ahmed [147] demonstrated that irrigating rice with groundwater containing As concentrations ranging from 1850 to 5020 μg kg^{-1} may lead to grain-As accumulation ranging from 6 to 513 μg/L in rice. The author reported that excessive use of As-rich groundwater can result in massive deposition of As in paddy soils, subsequently transferring into rice grains. On the other hand, some countries or regions have low As content in their irrigation water, falling below the WHO guideline value of 20 μg As L^{-1}, suggesting that they can produce rice with acceptable As levels [147]. However, the documentation of As content in groundwater or borehole water in most African regions, including Malawi, is insufficient, despite numerous constructions of deep wells and boreholes.

In the case of cadmium, Chavez et al. [98] reported that irrigating rice with Cd-contaminated water led to substantial Cd accumulation in the top-soils (0–15 cm) in south Ecuador under cacao production. Yang et al. [148] found that irrigating rice paddies with untreated mine wastewater increased Cd concentrations in paddy soils and subsequently elevated Cd accumulation in rice grains. These observations indicate that continuous irrigation with contaminated water may increase Cd loading in the soil, potentially reducing crop yields and increasing the bioaccumulation of carcinogens in rice grains, affecting both the nutritional status and the economy of farming communities. However, Cd concentrations in irrigation water in most African regions meet the quality standards of 20 μg/L, although data is limited [149]. The low soil-Cd content in most African countries can be attributed to the limited pollution from metal or coal mining and other relevant anthropogenic activities.

It is important to note that rice paddies can also be contaminated with heavy metals from pesticides and chemical fertilizers [53,119,143,150]. Regions with a known history of using arsenic-containing pesticides and cadmium-containing fertilizers in crop production have reported elevated concentrations of As, Cd, and Pb in rice grains [53,119,143,150]. For example, extensive use of methylated arsenical pesticides in cotton fields in the USA has resulted in massive contamination of paddy soils and direct uptake of As metabolites from the soil, leading to elevated grain-As concentrations [53]. Similarly, the use of cadmium-containing fertilizers in rice crop production increases cadmium concentrations in farm rice [150]. Studies conducted in Senegal and Tanzania found high levels of pesticide residues in rice samples from paddy fields, posing health risks to consumers [44,151].

The reported facts regarding heavy metal contamination in rice production in Africa present a mixed picture. On one hand, the data suggests that compared to other regions, African countries generally have lower concentrations of arsenic (As), cadmium (Cd), and lead (Pb) in irrigation water, meeting the quality standards set by the WHO which implies that African rice crops may have lower levels of these contaminants, which could be seen as a positive aspect in terms of food safety and human health. Additionally, the limited pollution from metal or coal mining and other anthropogenic activities is perceived to contribute to the low soil-Cd content in most African countries which supported the notion of lower heavy metal contamination in African rice production. However, while rice production in most African countries may have lower concentrations of As, Cd, and Pb compared to other regions, further research and monitoring are needed to fully understand the extent of heavy metal accumulation in African rice crops considering that there is limited documentation of As content in groundwater or borehole water in most African regions, despite the construction of numerous deep wells and boreholes.

5.2. Impact of agronomical measures

Rice cultivars exhibit varying sensitivity to the accumulation of Metal(loids), with some cultivars highly sensitive to As, Cd, and Pb, while others are highly tolerant [39,139,152–154]. Tolerant cultivars tend to accumulate higher concentrations of As, Cd, and Pb in their grains compared to sensitive cultivars [13,147]. Specifically, tolerant cultivars accumulate higher iAs in their grains compared to sensitive cultivars [13,147]. Norton et al. [155,156] reported significant genotypic differences in grain-As accumulation among different rice cultivars in field trials conducted in Bangladesh, attributing the observed differences to cultivar-specific uptake and translocation capabilities. Similarly, Kuramata et al. [85] investigated the effect of cultivar on Cd uptake in rice cultivated in soils with varying levels of As. The authors found minimal or low genotypic differences in grain-total As (tAs) or inorganic As (iAs) accumulation in rice grown in low As soils, but significant or high genotypic differences in grain-tAs or dimethylarsinic acid (DMA) accumulation in rice grown in high As soils [85]. Additionally, Wu et al. [94] and Islam et al. [119] demonstrated that certain cultivars tend to accumulate a higher percentage of grain-iAs compared to others with Indica rice cultivars bioaccumulated higher grain-tAs (range: 21–296 μg kg^{-1}) compared to Japonica cultivars (range: 5–274 μg kg^{-1} [54,67]. However, none of these studies have specifically targeted African-grown rice cultivars or African rice (Oryza glaberrima), highlighting the need for further research in Africa to identify low metal (loid)-accumulating rice cultivars for safer rice production.

Regarding Cd, several studies have shown that the uptake and accumulation of Cd in rice are highly dependent on rice genotypes, with certain cultivars exhibiting a high affinity for Cd, while others show contrasting results [11,15,23,39]. For example, Song et al. [15] reported that cultivars with a high affinity for soil-Cd uptake had elevated levels of Cd and transferred a high proportion of it into rice grains, increasing the risk of contamination in the food chain. Ye et al. [139] compared the bioaccumulation of three types of rice cultivars and found that indica cultivars accumulate greater grain-Cd than hybrid and japonica cultivars. Conversely, Ma et al. [39] reported higher soil-to-grain-Cd transfer factors (TF) in hybrid cultivars than in Indica and Japonica. These observations suggest that cultivar selection plays a significant role in mitigating elevated grain-Cd accumulation in rice, underscoring the importance of cultivar screening programs to identify low-Cd accumulating rice cultivars [157,158]. However, most of such screening programs/projects have focused on major rice-producing countries [157,158], neglecting many low-income African countries that could benefit economically from producing safer rice for the global population.

Practicing non-flooded (aerobic) soil conditions, such as alternate wetting and drying (AWD) irrigation regimes, low water irrigation management, and intermittent sprinkler irrigation, have been reported to significantly reduce the bioavailability, solubility, and mobility of arsenic in soil pore water [64,159–161]. However, these practices can increase the mobility, availability, and uptake of certain Metal(loids) [54,105,162]. Aerated rice fields are often acidic with low soil pH, which can result in high Cd concentrations in non-flooded soil conditions, particularly during prolonged droughts and in soils with low pH [54,105,162]. Traditional continuous flooding (CF) practices, characterized by anaerobic soil conditions, are commonly used in most African paddy fields [44,163,164]. However, episodes of drought may lead to default alternate wetting and drying (AWD) conditions (alternating between oxic and anoxic soil conditions) or simply low water irrigation management (continuous oxic soil conditions [155,156]. Consequently, natural disasters such as floods and droughts may have variable impacts on the accumulation of As, Cd, and Pb in rice. Positive impacts of droughts on accumulation of As, Cd, and Pb in rice have not been extensively evaluated in African paddy fields. Such studies and projects focusing on paddy fields would help understand how such natural disasters affect the bioavailability, solubility, and mobility of metal (loids which in turn have effect on human health.

5.3. Impact of soil conditions on bioavailability, mobility, and uptake of Metal(loids) in rice systems

Numerous studies have demonstrated that the bioavailability of arsenic (As), cadmium (Cd), and lead (Pb) varies depending on soil types. Certain soil types have been found to enhance the retention of As, Cd, and Pb, while others poorly retain them [3,72,165–167]. For example, Ishikawa et al. [165] reported that rice cultivated in andosols soil type bioaccumulates limited amounts of As in grains compared to rice cultivated in gley lowland soils due to the limited plant-available soil-As in andosols. It has also been observed that certain forms of As exhibit higher bioavailability and are readily taken up by rice plants under specific rhizosphere conditions, while under different rhizosphere conditions, the opposite is true. Under anoxic soil conditions, for instance, AsIII species are more soluble, mobile, predominant, and bioavailable in the soil solution compared to AsV species, leading to easier uptake by plants [1,105,162,168]. Conversely, under anoxic soil conditions, AsV strongly sorbs to iron and manganese oxides and hydroxides, making it less soluble, less mobile, and less bioavailable, thus resulting in limited uptake by rice plants [105,162,168].

Regarding Cd, Sebastian et al. [3] reported that rice plants grown in vertisol tend to accumulate less Cd due to the presence of calcium carbonate, which reduces plant-available Cd and consequently limits Cd uptake. However, even in soils with low Cd content, rice plants still uptake significant amounts of Cd due to its high bioconcentration factor [15].

In the context of paddy rice fields in Africa, including Malawi, there is limited information available regarding the classification and conditions of these fields. This lack of information is due to the limited number of programs or projects specifically targeting African-grown rice. Implementing programs or projects aimed at identifying rice paddies requiring mitigation measures to limit or reduce metalloid bioaccumulation, like those in Bangladesh, Thailand, Argentina, Brazil, and other countries, would be beneficial [116,169,170].

Soil pH plays a crucial role in the bioavailability of metals in soil and porewater [72,171–175]. Low soil pH decreases the sorption of most metal (loid)s, thereby increasing their bioavailability and mobility in porewater [174,175]. Soil pH influences the pH-dependent surface charge on sorbents, reducing their affinity for As, Cd, and Pb sorption sites [176]. Consequently, this process increases the dissolution of metal (loid)s in acidic soils (low soil pH) compared to alkaline (high soil pH) soils [10,177], indicating that the composition of soil texture and its acidity/alkalinity determines the extent to which nutrients are made available to plants in soil and porewater [77,96,178,179]. Therefore, the mobility and bioavailability of As, Cd, and Pb increase with decreasing soil pH [1,96,168], significantly impacting the uptake and bioaccumulation of these metal (loid)s in rice. For instance, a study conducted by Rafiq et al. [1] and Wan et al. [92] reported a negative correlation between Cd contents in rice grains and soil pH, confirming that the bioavailability of Cd in soil is highly dependent on soil pH. The authors also reported that lower soil pH enhanced the transformation of Cd from immobile forms to more easily bio accessible and bioavailable forms [1,96,168].

In the case of Pb, it is generally insoluble and not readily available for plant uptake within the normal range [34,180]. Higher soil pH markedly decreases Pb solubility in porewater, particularly due to Pb precipitation as hydroxide, phosphate, or carbonate at high soil pH. Additionally, high soil pH (alkaline soil conditions) promotes the formation of stable Pb-organic complexes. However, soil pH interacts with other factors such as soil sorbents, soil types, soil organic matter, and soil metal content, which may vary from region to region. Considering that most paddy fields in Malawi have a soil pH range from 6.4 to 8.4 [72], and consist of luvisols, fluvisols, and gleysols soil types [72], and relatively higher iron content [181–183], further understanding is required regarding the implications of the interaction of these factors.

The content of certain soil components, such as iron (Fe), zinc (Zn), manganese (Mn), and aluminium (Al) oxides and hydroxides, significantly impacts the bioavailability, mobility, and uptake of metals and metalloids in rice systems [10,175]. These sorbents reduce the uptake of metals and metalloids, such as As, Cd, and Pb, by immobilizing them on their surfaces, rendering them unavailable for plant uptake [10,38,119,175]. The mobility, bioavailability, and solubility of soil-As, Cd, and Pb in porewater depend on the type and concentration of sorbents present in the soil and water solution. For instance, higher levels of soil Fe oxides and hydroxides may result in greater iron plaque formation on rice roots, trapping and retaining As [10,38,119,175]. The mobility of As in soil is inversely proportional to the concentration of sorbing components in soil porewater, such as Fe and Al oxides and hydroxides [10,175]. Arsenic in soil or soil/water solution is immobilized through adsorption onto Fe oxides and hydroxides by replacing surface hydroxyl groups with As ions, along with the formation of amorphous Fe(III) arsenates [10,38,119,175]. Soil Zn also influences Cd mobility and Phyto availability in soil, significantly reducing Cd uptake because Cd and Zn have similar chemistry such that they compete for the same sites during uptake [10,149,175].

The utilization of soil amendments as sorbents or adsorbents have been reported to effectively diminish the accumulation of As, Cd, and Pb in rice grains. The extent of reduction depends on their presence and concentration in the soil rhizosphere [184]. Most proposed soil amendments function by sorbing metal (loid)s onto their surfaces, thus reducing their availability for uptake. For instance, extensive research has focused on Fe-containing materials for As immobilization particularly zero-valent iron (Fe°), which exhibits high adsorption capacities for AsV in oxic soils [36,185,186]. In submerged soils, AsIII displays greater mobility than AsV due to reductive dissociation of iron oxides [37,184,187,188].

Zero-valent iron (Fe°) serves as a sorbent with remarkable adsorption capacities, particularly for AsV [9,189], in oxic soils, where AsV is the dominant species and strongly sorbs onto Fe° minerals. Conversely, in submerged soils, the mobility of AsIII surpasses AsV because reductive dissociations of iron oxides, along with the reduction of AsV to AsIII in reducing environments, are prevalent. Qiao et al. [189] demonstrated that Fe° amendment significantly decreased As bioavailability in CF water regime, but its impact under AWD water management remains unknown.

Selenium is suggested as a soil amendment that can reduce Cd accumulation in the rice, as additions of Se to the soil have been found to significantly decrease translocation of Cd from roots to shoots in rice seedlings under hydroponic conditions. Furthermore, Wan et al. [92] reported that Se additions to the soil markedly reduced Cd in rice seedlings by reducing Cd translocation from roots to shoots under hydroponic conditions. Selenium has the potential to decrease Cd bioavailability in soils thereby restricting the uptake and accumulation of Cd in plants [92].

Multiple studies have demonstrated variations in the bioavailability of As, Cd and Pb depending on soil types. Certain soil types have been found to enhance retention of these carcinogens, while others exhibit poor retention [3,72,165–167]. For instances, Ishikawa et al. [165] observed that rice cultivated in andosols soil type accumulates limited amounts of As in grains compared to those cultivated in gley lowland soils due to the low availability of As in andosols soils.

Furthermore, the bioavailability of specific As species can differ under different rhizosphere conditions.

Under anoxic soil conditions, AsIII species are more soluble, mobile, predominant and bioavailable in the soil solution compared to AsV species, thereby making them easily absorbed by plants [1,105,162,168]. Conversely, AsV exhibits stronger sorption on iron and manganese oxides and hydroxides under anoxic soil conditions resulting in lower solubility, mobility, and bioavailability [105,162,168]. Consequently, AsV species are less readily absorbed by rice plants, especially in severe cases [105,162,168]. Regarding Cd, Sebastian et al. [3] reported that rice plants grown in vertisol tend to accumulate lower levels of Cd due to the presence calcium carbonate, which reduces plant available Cd and subsequently hampers Cd uptake in plants.

However, even in soils with low level of Cd content, rice plants still absorb a larger amount of Cd due to its easy transfer from soil to plants, facilitated by a high bioconcentration factor [15]. Unfortunately, for rice paddies of Africa, including Malawi, there is limited information about paddy fields classification and conditions. This is mainly due to absence of programs or projects specifically targeting African grown rice or African rice. Implementing such programs could be beneficially in identifying rice paddies that require mitigatory measures to limit or reduce bioaccumulation of metalloids in rice, similar to efforts in Bangladesh, Thailand, Argentina, Brazil and other regions [116,169,170].

This is mainly due to the absence of programs or projects specifically targeting African-grown rice or African rice. Implementing such programs could be beneficial in identifying rice paddies that require mitigatory measures to limit or reduce the bioaccumulation of metalloids in rice, similar to efforts in Bangladesh, Thailand, Argentina, Brazil, and other regions [116,169,170]. The soil-pH plays a significant role in the bioavailability of metals in both soil and porewater. A low soil-pH tends to decrease sorption of most Metal (loids), thereby increasing their bioavailability and mobility in porewater [174,175]. Additionally, soil-pH influences the pH-dependent surface charge on sorbents, leading to decreased affinity for As, Cd and Pb onto the sorption sites [176]. Consequently, this process enhances Metal(loids) dissolution in acidic soils (low soil pH) compared to alkaline soils (high soil (pH) [10,177]. These findings indicate that the makeup of soil texture and its acidity/alkalinity determine the extent to which nutrients are made available to plants in soil and porewater [77,96,178,179]. Thus, the mobility and bioavailability of As, Cd and Pb increases as soil-pH decreases [1,96,168], which significantly affects the uptake and accumulation of these Metal(loids) in rice. For instance, studies conducted by Rafiq et al. [1] and Wan et al. [92] revealed a negative correlation between Cd contents in rice grains and soil pH, confirming the strong influence of soil pH on the bioavailability of Cd in the soil. The authors also noted that lower soil pH promotes the transformation of Cd from immobile forms to more easily accessible and bioavailable forms [1,96,168].

Regarding Pb, it is highly insoluble and generally not readily available for plant uptake within the normal soil pH range. Higher soil pH levels significantly decrease Pb solubility in porewater, primarily due to Pb precipitation as hydroxide, phosphate, or carbonate.

Additionally, alkaline soil conditions resulting from high soil pH promote the formation of stable Pb-organic complexes. However, the interaction between soil pH and other factors, such as soil sorbents, soil types, soil organic matter, and soil metal content, can vary from region to region. Therefore, considering that most paddy fields in Malawi have a soil pH range of 6.4–8.4 [72], consisting of luvisols, fluvisols, and gleysols soil types [72], as well as relatively higher iron content [181–183], a comprehensive understanding of the implications of these factors' interaction is necessary. The presence of sorbents such as Fe, Zn, Mn, and Al oxides and hydroxides in soils has a notable impact on the bioavailability, mobility, and uptake of metals and metalloids in rice systems [10,175]. These sorbents, including Fe, Zn, Mn, and Al oxides and hydroxides, immobilize metals and metalloids like As, Cd, and Pb by binding them to their surfaces, rendering them unavailable for plant uptake [38,119]. However, the mobility, bioavailability, and solubility of soil-As, Cd, and Pb in porewater depend on the type and quantity of sorbents present in the soil-water solution.

For instance, the presence of soil Fe oxides and hydroxides can lead to the formation of iron plaques on rice roots, effectively trapping and retaining As [10]. Kabata-Pendias & Pendias [10] reported an inverse relationship between the mobility of As in soil and the concentration of sorbing components, such as Fe and Al oxides and hydroxides, in the soil porewater. Adsorption onto Fe oxides and hydroxides, where arsenic ions replace surface hydroxyl groups, along with the formation of amorphous Fe(III) arsenates, immobilizes arsenic in the soil or soil-water solution.

Similarly, the mobility and availability of Cd in the soil are also influenced by soil Zn, which reduces the uptake of Cd. The application of soil amendments as sorbents or adsorbents can effectively decrease the accumulation of As, Cd, and Pb in rice grains, depending on their concentration in the soil rhizosphere. Most proposed soil amendments function by sorbing metals and metalloids onto their surface, thereby reducing their availability for uptake. Fe-based materials containing iron, serving as sorbents, have been subject to extensive research for their ability to immobilize As, with zero-valent iron exhibiting high adsorption capacities for AsV in oxygenated soils. However, in waterlogged (submerged) soils, the mobility of AsIII is greater than that of AsV due to the reductive dissolution of iron oxides. In a study conducted by Qiao et al. [189] it was observed that the amendment of Fe° significantly decreased the bioavailability of As in continuous flooding (CF) water regime. However, the effects of Fe° amendment under in alternate wetting and drying (AWD) water management regime remain unknown.

6. Strategies for mitigating As, Cd and Pd accumulation inn rice

Many studies have reported promising strategies that can be used to reduce the accumulation of heavy metals such as arsenic (As), cadmium (Cd), and lead (Pb) in rice, including water management regimes, soil management practices, breeding of low Metal(loids) cultivars, certain agronomic practices, and phytoremediation (bioremediation) techniques [124,138,190]. Let us examine these techniques one by one and their level of adoption across the world.

One effective strategy to limit arsenic accumulation in rice is to control the quality of irrigation water by reducing the As, Cd, and Pb concentration in the porewater. This can be achieved by using clean water sources or introducing alternate wetting and drying (AWD) irrigation techniques. Alternate wetting and drying (AWD), is a water-saving technique that involves periodically draining the fields to allow the soil to dry out before reflooding to promote aerobic conditions in the rhizosphere [124,138,188,191]. AWD is a water-saving technique that has shown to be effective in reducing accumulation of heavy metals such as As, Cd, and Pb in rice grains [151,188,190,192,193]. Many studies have demonstrated that AWD significantly reduces As accumulation in rice by up to 25% [194–196]. For instance, In Mlangeni et al. reported lower iAs and higher oAs concentrations in rice cultivated under AWD compared to that cultivated under CF in greenhouse experiments in Aberdeen, Scotland [138]. The study reported that a 27% lower arsenic concentration in rice under AWD (1686 mg kg^{-1}) compared to that under CF (2309 mg kg^{-1}) [138]. Additionally, Li et al. reported that AWD decreased grain total As concentration by 41–68% decrease in. AWD has also been reported to reduce soil Cd concentration in rice paddy fields and Cd accumulation in rice by at least 35% compared to continuous flooding (CF). For Pb, AWD has also been reported to effectively reduce Pb accumulation in rice by at least 19% compared to CF without significantly affecting yields and the quality of rice grain. However, the effectiveness of AWD depends on other factors such as rice varieties, soil type, and climate conditions, and no study has reported the effectiveness of AWD on African rice varieties, African soil paddies, and local climate conditions.

Controlled irrigation with less contaminated surface water sources such as freshwater has also been reported to significantly limit and mitigate elevated As accumulation in rice, particularly in areas where farmers rely on As-contaminated groundwater for irrigation. Thus, switching from tube-well groundwater irrigation to surface water (fresh canal, river, and lake water) irrigation reduces As accumulation in rice by at least 25%. However, this technique is not reported in African rice paddies.

Several soil-management practices can be used to limit the accumulation of As, Cd, and Pb in rice, including soil testing and nutrient management, phytoremediation, soil amendments, and crop rotation. Regular soil testing and nutrient management can help ensure that soil pH, organic matter content, and nutrient levels are optimal for plant growth. The optimal soil-pH range for rice cultivation (5.5–7.0) enhances relatively low solubility and minimal uptake of heavy metals by rice plants. Furthermore, adding organic matter to soil can bind to heavy metals such as As, Cd, and Pb and form complexes that are less mobile and available to rice plants, which can reduce their uptake. Organic matter can also increase the soil's water-holding capacity due to improved soil structure, which can reduce the mobility of As, Cd, and Pb in the soil.

Policy interventions such as regulations on arsenic levels in irrigation water and rice products can also help to mitigate As, Cd and Pb accumulation in rice. For instance, the European Commission has set MCL for inorganic As in rice and rice products. WHO/FAO regulated As content in rice at 300 μg kg^{-1} [26,51]. Similarly, there is need for African countries to regulate Metal(loids) concentration in rice in order to avail safe rice to the global market.

Gene editing (GE) techniques have also emerged as a potentially viable strategy in reducing the bioaccumulation toxic metalloids in rice [197–199]. The gene editing techniques have proven to be efficient and easy to use in reducing bioaccumulation of toxic metal (poids) in rice grains [197,198]. The techniques involves identifying specific genes that are responsible for the uptake, transport, and accumulation of As, Cd, and Pb in rice and then using gene editing techniques such as CRISPR-Cas9 (clustered regularly interspaced short palindromic repeats – associated), in which the identified genes are modified to reduce the plant's ability to take up or accumulate As, Cd, and Pb which involves disrupting transporter genes or modifying regulatory genes to alter the plants' responses to these elements. For example, Tang et al. [200] found that in Cd-contaminated paddy field trials, the Cd concentration in osnramp5 grains consistently remained below 50 μg kg^{-1} which significantly differed with Huazhan, the wild-type indica rice, where Cd concentrations in grains ranged from 330 μg kg^{-1} to 2900 μg kg^{-1}. These findings demonstrated that gene editing can effective limit accumulation of toxic metals in rice thereby improving quality of rice.

7. Conclusions and perspectives

A comprehensive study is needed to identify which rice cultivars are As- and Cd-sensitive or tolerant due to cultivar-dependent accumulation among the many varieties grown in Africa. No studies have investigated Malawian rice cultivar accumulation capabilities to date. Identifying cultivars can help reduce health risks associated with consuming contaminated rice and bring economic benefits to farmers.

A comprehensive field study or survey is necessary to validate the response of rice to As, Cd, and Pb accumulation in Malawi, as these parameters vary from one rice scheme to another. High soil pH reported at Lifuwu research institute farms and most parts of Nkhota-kota marshes and swamps may stimulate adsorption of As and immobilization of Cd, while low soil pH reported at Bwanje irrigation scheme, Nkondezi (Limphasa), and Nazolo and Nkhate rice schemes may stimulate release and mobilization of As, Cd, and Pb accumulation into rice paddies. Hence, both types of soils are important to investigate. The different types and quantities of sorbents in soil, such as Fe, Mn, and Al, have different adsorption capabilities for different elements, which influence the mobility and bioavailability of As, Cd, and Pb differently. Therefore, high soil iron content reported in rice paddies of Nkhota-kota and lower shire valley would be of interest to investigate how As, Cd, and Pb accumulation occurs. The impact of prolonged floods and persistent drought on As, Cd, and Pb uptake by rice plants could be significant and provide insights into the toxicological impact of climate change on rice production.

Investigating the As, Cd, and Pb content in irrigation water would also be important since irrigating rice with contaminated irrigation water has a significant impact on the accumulation of these elements in rice paddies. Malawian rice farmers use different sources of irrigation water, whose As, Cd, and Pb content is not well studied. Most irrigation schemes abstract or divert water from rivers into rice farms, while others depend on rainwater that periodically floods their rice farms. Contamination of river water may vary depending on proximity to mines and city discharges, and rice farms situated downstream of coal mines may receive contaminated irrigation water due to the downstream movement of As, Cd, and Pb. Limited data and monitoring systems make it challenging to develop effective strategies for remediation and prevention, and there is a need for improved monitoring systems to detect and respond to contamination incidents promptly.

Many African countries lack the financial resources and technical expertise needed to address paddy field contamination effectively. Therefore, there is a need for increased investment in research, training, and infrastructure to mitigate the effects of contamination and improve food safety.

Author contribution statement

All authors listed have significantly contributed to the development and the writing of this article.

Data availability statement

Data will be made available on request.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper

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