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RESEARCH ARTICLE

Dietary exposure assessment of cadmium, arsenic, and lead in market rice from Sri Lanka

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Abstract

Rice is frequently reported to be contaminated with heavy metals (HMs); thus, the human health risks from its consumption have received increasing attention. A total of 165 commercial rice samples from Sri Lanka were collected to determine their cadmium (Cd), arsenic (As), and lead (Pb) concentrations. The exposure risk for Sri Lankans from the estimated daily intakes (EDIs) of these toxicants was assessed. Simultaneously, non-carcinogenic and carcinogenic risks were evaluated using hazard quotients (HQs) and the hazard index (HI). The results revealed that the average levels of Cd, As, and Pb in commercial rice were 0.080 ± 0.130, 0.077 ± 0.040, and 0.031 ± 0.050 mg/kg, respectively, with ranges of 0.003–0.727, 0.019–0.217, and 0.001–0.345 mg/kg (expressed on a dry weight basis), respectively. The average EDIs of Cd, inorganic As (iAs), and Pb were 0.772, 0.490, and 0.306 μg/kg body weight (bw)/day, respectively; these were below provisional tolerable weekly intake (PTWI) values recommended by the Joint FAO/WHO Expert Committee on Food Additives (JECFA), but iAs was above the recommended reference doses (RfDs) recommended by the United States Environmental Protection Agency (USEPA). However, approximately 25% and 75% of the Cd and iAs HQs for the Sri Lankan population, respectively, were greater than 1, suggesting a potential health risk, whereas the HQs for Pb was less than 1. Considering the additive effect, HI values of the P90, P95, P97.5, and P99 percentiles would reach 4.773, 6.458, 8.392, and 11.614, implying that intake of the combined metals might result in potential health risks.

Keywords Heavy metals · Market rice · Sri Lanka · Estimated daily intakes (EDI) · Monte Carlo simulation

Introduction

As a staple food, rice is frequently reported to be contaminated with cadmium (Cd), arsenic (As), and lead (Pb), which are all known carcinogens (Fu et al. 2008; Chandrajith et al. 2011; Li et al. 2011; Norton et al. 2014; Levine et al. 2016). Intensive agricultural and industrial activity is considered to be the major reason for elevated levels of these toxicants in rice. Officially listed as a Category I carcinogen by the International Agency for Research on Cancer (IARC), Cd is also well known to residents as a carcinogen (EFSA 2012), and it can also cause anemia and serious harm to kidneys and bones. The serious toxicity of Cd attracted scientific focus starting in the mid-1900s (Godt et al. 2006). The metalloid As is usually referred to as a heavy metal and is toxic to the majority of organ systems, particularly the kidneys (Cohen et al. 2006). Chronic As exposure can affect the vascular system and cause hypertension and cardiovascular disease (Jomova et al. 2011), and As is also defined as a Category I human

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carcinogen. Pb is also listed as a carcinogenic toxicant, accounting for approximately 0.6% of the global burden of disease (Fewtrell et al. 2003; Saleh et al. 2019). Long-term Pb exposure may cause headaches, convulsions, tremors, and paralysis, while acute exposure is always associated with gastrointestinal disorders, kidney damage, neurological effects, convulsions, and even death (Fewtrell et al. 2003).

Rice exhibits a high ability to accumulate Cd, As, and Pb and has become the primary pathway for EDI of these toxicants (Jayasumana et al. 2015b; Welna et al. 2015; Jinadasa and Fowler 2019). The high concentration of Cd in rice is a widespread global problem, especially in South Asian countries, contributing to the majority of dietary Cd ingestion and human exposure (Khan et al. 2014; Chanpiwat et al. 2019; Rehman et al. 2019). Rice is an important dietary source of inorganic As for humans, as the rice plant can absorb 10 times more As than other agricultural crops (Williams et al. 2007). Additionally, rice can also be enriched in Pb, giving rise to concern about its exposure to health risk via rice consumption (Williams et al. 2009; Liu et al. 2013).

The Joint FAO/WHO Expert Committee on Food Additives (JECFA) has published a series of provisional tolerable weekly intakes (PTWIs) for Cd and Pb. These include thresholds for Cd of 7 mg/kg bw/week (FAO/WHO 2003) and Pb of 25 mg/kg bw/week (FAO/WHO 1993). Furthermore, the US Environmental Protection Agency (EPA) has recommended reference doses (RfDs) for Cd and inorganic As (iAs) of 0.001 mg/kg bw/day (0.007 mg/kg bw/week) and 0.0003 mg/kg bw/day (0.0021 mg/kg bw/week), respectively (USEPA 1989, 1991); however, no reference dose has been recommended for Pb. Recently, the US Food and Drug Administration (FDA) provided a provisional total tolerable intake (PTTI) level of Pb for adults of 75 μg/day (525 μg/week; (FAO/WHO 2013).

Sri Lanka is a tropical agricultural island country that lies in the Indian Ocean. The primary form of agriculture in Sri Lanka is rice production, which currently achieves an annual output of approximately 2,358,400 t, with an average annual consumption of 110 kg per capita (Walisinghe and Gunaratne 2012). Sri Lanka produced 3.08 million tonnes of rice and approximately 3.93 million tonnes were consumed in 2018 (FAOSTAT 2019). As an island with a total coastline of approximately 1700 km, the consumption of fish comes from capture fisheries in Sri Lanka. Rice and fish consumption are considered to be two major dietary sources of heavy metals (HMs) for Sri Lankans (Jinadasa et al. 2014; Perera et al. 2016; Xu et al. 2020).

The present study had two main objectives. The first objective was to determine the concentrations of Cd, As, and Pb in commercial rice. The second objective was to assess the health risks relating to Cd, As, and Pb exposure via commercial rice consumption.

Materials and methods

Survey region

Sri Lanka (6–10° N; 79–82° E) is an island with a land area of 65,610 km^2, located in the northern Indian Ocean towards the southeast of India. The temperature in Sri Lanka varies with altitude, with average annual average temperatures of 27 and 15 °C in the lowlands (altitude of 100–150 m) and highlands (altitude of approximately 2400 m), respectively (Marambe et al. 2015).

Agricultural activities are the major anthropogenic sources of Cd, As, and Pb in Sri Lanka. Chemical fertilizers and pesticides usually contain high concentrations of Cd and As, with levels of 23.5–71.7 and 31.0 mg/kg recorded in phosphate fertilizers, respectively (Bandara et al. 2010; Jayasumana et al. 2015a; Ayala et al. 2018). Triple superphosphate (TSP) fertilizer is of special interest due to its high Cd content, which varies widely (Chandrajith et al. 2010). For Pb, vehicle emissions are considered another source before 2002 in addition to agricultural activity (Nandasena et al. 2010). Strong monsoon winds that brings Pb from the polluted part of the country are the most likely source of soil Pb (Ranasinghe et al. 2006).

Sampling

According to the rice cultivation in each provincial region, a total of 165 samples of polished rice (including white and red types) were collected from markets in 2018. These covered all nine provinces of Sri Lanka: North Central = NC (n = 28), Western = W (n = 28), Central = C (n = 17), Northern = N (n = 15), Eastern = E (n = 20), Southern = So (n = 13), North Western = NW (n = 12), Uva = U (n = 15), and Sabaragamuwa = Sa (n = 17); see Table S1. Each sample comprised approximately 0.5 kg of rice, which was purchased and sealed in a separate polyethylene zip-locked bag, transported to the laboratory, and refrigerated at −20 °C prior to further processing.

Sample preparation and analysis

First, rice samples were thoroughly homogenized. Approximately 30 g of each sample was flushed three times with ultra-pure water and freeze-dried (FDU-2110, EYELA, Japan). The dried samples were ground into a powder (IKA-A11 basic, IKA, Germany), passed through a #80 mesh sieve, and finally preserved in zip-locked polyethylene bags and refrigerated at −20 °C for analysis (Xu et al. 2017).

Rice samples were then digested for Cd, As, and Pb analysis. In brief, approximately 0.3 g of each sample was weighed into a Teflon tube and distilled ultra-pure HNO_3 (3.5 mL) was added. The tube was hermetically sealed in a reaction kettle and heated at 150 °C for 16 h in an oven. Once the digestion liquid had cooled down to indoor temperature, H_2O_2 (2 mL) was added and the sample heated until all liquid had evaporated. Then, ultra-pure water (4.5 mL) and HNO_3 (0.5 mL) were added to the Teflon tube for another 16 h digestion with 150 °C (Lu et al. 2019). Finally, the liquid was transferred to a polyethylene centrifuge tube until analysis. Prior to measurements, approximately 1 mL of the digest was diluted to 5 mL with ultra-pure water to keep the acid concentration below 2%. The sample was then analyzed by inductively coupled plasma mass spectrometry (ICP-MS; NexION™ 300X, PerkinElmer, USA).

The concentrations of heavy metals were calculated on a dry weight (DW) basis in all samples.

QA/qc

Analytical data for Cd, As, and Pb were controlled with reagent blanks, method blanks, duplicates, standard reference material (GBW 10020, citrus leaf), and internal standard (Rh). The measured values of Cd, As, and Pb in the GBW 10020 were 0.210 ± 0.010, 1.12 ± 0.022, and 10.15 ± 0.17 mg/kg, respectively, which were in accordance with the certified values of 0.170 ± 0.020, 1.10 ± 0.200, and 9.70 ± 0.900 mg/kg, respectively. The samples were divided into two batches, each batch of samples with 3 citrus leaf (GBW 10020) for quality control. Rhodium was used as an internal standard, and the recovery ranged from 83.5 to 114%. The standard values, self-test value, and recovery rate of citrus leaf for Cd, As, and Pb were 0.17 ± 0.02, 1.1 ± 0.2, 9.7 ± 0.9 mg/kg; 0.20 ± 0.01, 1.12 ± 0.02, 10.14 ± 0.17 mg/kg; and 110.37% ± 5.40%, 102.03% ± 1.97%, 104.95% ± 4.39% (Table S2), respectively. The detection limits of ICP-MS for Cd, As, and Pb were 0.005, 0.015, and 0.0003 μg/L, respectively. Calculating as matrix of this study, the detection limits for Cd, As, and Pb were 1.25, 3.75, and 0.0075 μg/kg, respectively.

In all cases, ultra-pure acids were used, other chemical reagents were of analytical grade, and ultra-pure water (18.5 MΩ/cm) was generated using a Millipore machine (Milli-Q REFERENCE, Millipore, USA). All glassware and Teflon tubes were soaked in nitric acid for 24 h and washed with ultra-pure water to avoid cross-contamination.

Health risk assessment

Estimated daily intake (EDI)

The EDIs of Cd, As, and Pb were calculated according to Eq. (1) (Qian et al. 2010):

$$\mathrm{EDI} = \frac{\mathrm{CF} \times \mathrm{IR} \times \mathrm{EF} \times \mathrm{ED}}{\mathrm{BW} \times \mathrm{AT}} \tag{1}$$

Here, CF is the HM level in rice (mg/kg); IR is the rice absorption rate (g/person/day), with mean daily rice depletion data obtained from the consumption questionnaires distributed in Sri Lanka (Table S4); EF is the exposure frequency (365 days/year); ED is the exposure duration (70 years); BW is the mean adult body weight of 60 kg (Diyabalanage et al. 2016, 2017); and AT is the average pathway-specific period of exposure for non-carcinogenic effects (365 days/year × the average exposure time, supposing 70 years in the present study).

Hazard quotient (HQ)

Non-cancer risk assessments were used to evaluate the potential health risk of contaminants by the hazard quotient (HQ). The method for evaluating HQ has been explained in detail by the US EPA (USEPA 2019). The HQ of each heavy metal was determined according to Eq. (2) (Qian et al. 2010) and (3) (USEPA 2001; Praveena and Omar 2017; Du et al. 2019)::

$$\mathrm{HQ}_{\mathrm{Cd,Pb}} = \mathrm{EDI} \times \frac{7}{\mathrm{PTWI}} \tag{2}$$

$$\mathrm{HQ}_{iAs} = \frac{\mathrm{EDI}}{\mathrm{RfD}} \tag{3}$$

Here, EDI is the estimated daily intake and PTWI is the provisional tolerable weekly intake.

The PTWI values applied for Cd, As, and Pb were those recommended by the JECFA.

The RfD for iAs was recommended by the US EPA. An $\mathrm{HQ} < 1$ is considered to be safe from non-carcinogenic effects, whereas $\mathrm{HQ} > 1$ indicates potential non-carcinogenic effects (Zheng et al. 2007).

Carcinogenic risk assessments are usually expressed as a risk value R, indicating an individual's increased likelihood of developing cancer over the course of a lifetime of exposure to a carcinogen. For chemical carcinogen As, the oral carcinogenic slope factor (SF) is 1.5 $[\mathrm{mg/(kg\ day)}]^{-1}$ (USEPA 1991). The SF of Cd was not assessed under the Integrated Risk Information System (IRIS) Program of the US EPA; hence, the R of As was determined according to Eq. (4) (USEPA 2001, Praveena and Omar 2017, Du et al. 2019):

$$R = \mathrm{EDI} \times \mathrm{SF} \tag{4}$$

Here, SF is the oral carcinogenic slope factor $[\mathrm{mg/(kg\ day)}]^{-1}$; and R is the carcinogenic risk ratio.

Hazard index (HI)

The hazard index (HI) was used to evaluate the overall potential risk of non-carcinogenic effects as a result of more than one element (USEPA 1986). The HI values were calculated using Eq. (5):

$$\mathrm{HI} = \sum_{n=1}^{i} HQ_n \quad (5)$$

An HI > 1 may be indicative of potential health risks (USEPA 2019).

Data analysis

Statistical analyses were performed with SPSS 16.0 (IBM, USA) and Origin 2018 (© Origin Lab Corporation, USA). Monte Carlo simulation was used for uncertainty analyses by Crystal Ball (©Oracle, Redwood City, CA, USA). The dietary exposure to HMs was calculated using the Monte Carlo simulation approach, which operated thousands of plots by stochastic sampling of probability distributions within the simulation. In the present study, approximately 10,000 iterations of Monte Carlo simulations were performed.

Results and discussion

Concentrations of Cd, As, and Pb

Summary statistics of the concentrations of Cd, As, and Pb in commercial rice samples are shown in Table S1. Specific standards for the maximum allowable concentration of HMs in polished rice, according to the Codex Alimentarius Commission set by the FAO/WHO (2013), were used for comparison. The regional distribution of concentrations among provinces is shown in Fig. 1.

Cadmium

The concentrations of Cd in commercial rice from Sri Lanka revealed a lognormal distribution (Fig. 2a). The average concentration of Cd in rice was 0.080 ± 0.130 mg/kg (median 0.033 mg/kg; range 0.003–0.730 mg/kg). Over 96.9% of samples were well below the Codex limit for polished rice. However, five samples from the Northern ($n = 1$), Western ($n = 2$), Eastern ($n = 1$), and Central ($n = 1$) provinces contained Cd concentrations of 0.488 to 0.727 mg/kg, which exceeded the Codex maximum permissible level of 0.400 mg/kg by 1.25 to 1.75 times, respectively. In the production areas of these five samples, soils or food were reported to have elevated Cd due to the agricultural fertilizer inputs (Bandara et al. 2010).

Based on one-way ANOVA, significant differences ($P < 0.05$) of rice Cd concentration were observed in different provinces (Fig. 1). The average rice Cd levels in Northern (0.117 mg/kg), Central (0.160 mg/kg), and Western (0.113 mg/kg) provinces were markedly higher than those in other provinces, while North Central (0.027 mg/kg), Uva (0.057 mg/kg), and Sabaragamuwa (0.039 mg/kg) provinces exhibited the lowest Cd concentrations on average.

Studies on Cd levels in rice from Sri Lankan markets and other worldwide countries are shown in Table S3. The Cd levels in rice in the present study were higher than those from previous studies in Sri Lanka (0.001–0.093 and 0.0017–0.0925 mg/kg) (Bandara et al. 2008, 2010) and in Iran (0.037 ± 0.023 mg/kg) based on DW (Mirlohi et al. 2013). The average Cd concentration was 0.050 mg/kg (ranging from 0.001 to 0.740 mg/kg) in China (Qian et al. 2010), and 0.099 mg/kg based on DW in Bangladesh (Meharg et al. 2013), which was also similar to the result in the present study. However, the average rice Cd concentration in Malaysia (0.160 mg/kg, based on DW) was above that in Sri Lanka (Praveena and Omar 2017).

Arsenic

The concentrations of As in commercial rice from Sri Lanka exhibited a lognormal distribution (Fig. 2b). No apparent provincial differences in rice As concentrations were observed (Fig. 1). The average concentration of As was 0.077 ± 0.040 mg/kg (median 0.067 mg/kg; range 0.019–0.217 mg/kg), which was considerably lower than the maximum Codex standard level (0.200 mg/kg). In the present study, two samples from Northern Province ($n = 2$) contained As at 0.217 and 0.223 mg/kg, which were both 1.1 times that of the Codex maximum permissible level.

Studies of As levels in rice from Sri Lankan markets and other worldwide countries are shown in Table S3. A similar average As concentration of 0.043 mg/kg (0.0025–0.213 mg/kg; median 0.0340 mg/kg, based on DW) was reported by Diyabalanage et al. (2016). Similarly, Mirlohi et al. (2013) and Praveena and Omar (2017) reported mean As values of 0.0335 and 0.087 mg/kg (based on DW) from Iran and Malaysia, respectively(Mirlohi et al. 2013; Praveena and Omar 2017). However, elevated levels of As have also been documented in Sri Lankan rice, ranging from 0.021 to 0.540 mg/kg (based on DW) (Jayasumana et al. 2015a), with phosphate fertilizers being found to be a major source of inorganic As. Qian et al. (2010) reported that 75.4% of milled rice samples ($n = 175$) contained less As than the maximum allowable concentration (MAC; 0.200 mg/kg) with an average As level of 0.119 mg/kg (< 0.008–0.490 mg/kg, based on DW) in China.

Lead

The concentrations of Pb in commercial rice in Sri Lanka exhibited a lognormal distribution (Fig. 2c). The average concentration of Pb in rice was 0.031 ± 0.052 mg/kg (median 0.015 mg/kg; range 0.001–0.345 mg/kg). The Pb

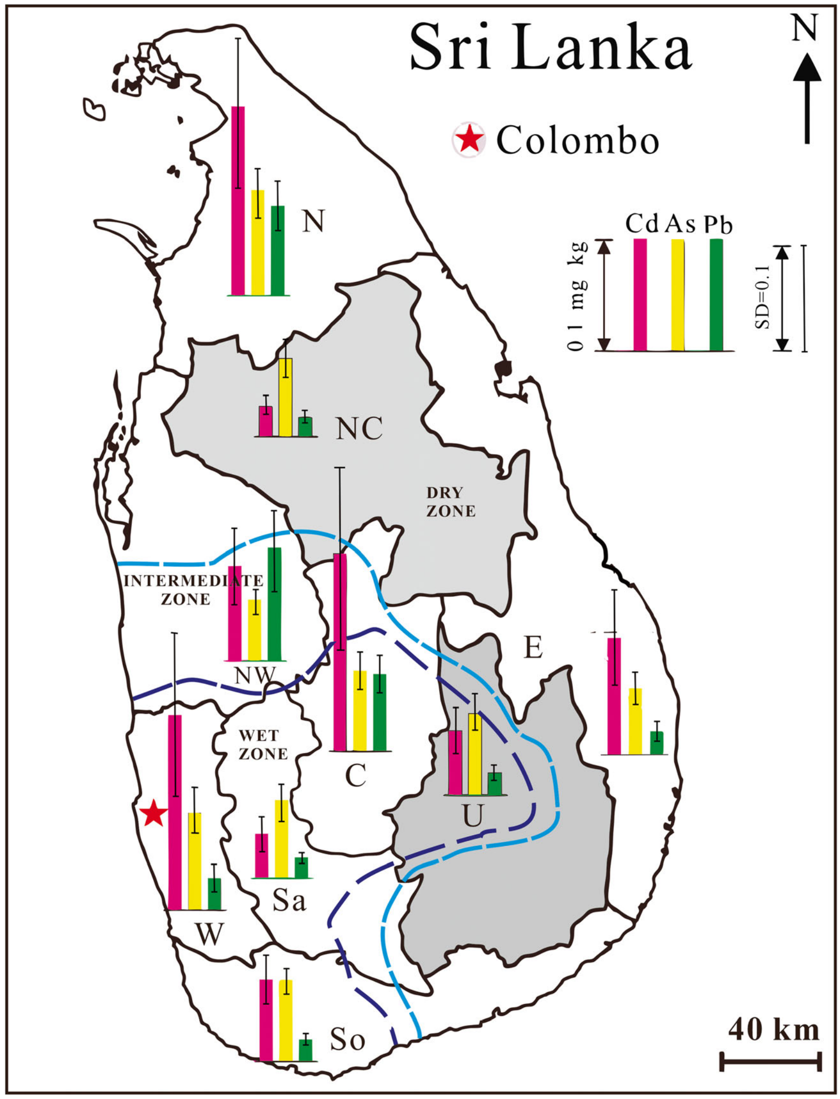

Fig. 1 Regional distribution of Cd, As, and Pb concentrations in rice in Sri Lanka

concentration in three samples collected from Central ($n = 1$), Northern ($n = 1$), and Western ($n = 1$) provinces exceeded that of the Codex threshold (0.400 mg/kg) by 1.5 to 2 times. Based on one-way ANOVA, highly significant differences ($P < 0.01$) in Pb were observed among provinces, with the average Pb concentrations in rice from Central (0.068 mg/kg), Northern (0.079 mg/kg), and North Western (0.101 mg/kg) provinces being markedly higher than those in rice from other provinces (Fig. 1). In addition to agricultural activity, vehicle emissions are considered another major source of Pb before 2002 (Nandasena et al. 2010).

Studies on Pb concentrations in rice from Sri Lankan markets and other worldwide countries are shown in Table S3. Norton et al. (2014) reported an average Pb concentration of 0.020 mg/kg (median 0.0150 mg/kg; range 0.003–0.061 mg/kg, based on DW) in commercial rice from Sri Lanka ($n = 41$), which agreed with the results of the present study. However, high concentrations of Pb in rice from Iran have been observed, with average levels of up to 0.328 ± 0.081 mg/kg (range 0.132–0.463 mg/kg, based on DW); this is almost 10 times more than the average concentration of Pb in Sri Lankan rice and exceeds the safe limit of 0.150 mg/kg set by the Iranian standard (Mirlohi et al. 2013). The average Pb concentration of 0.110 mg/kg (based on DW) in rice from Malaysia (Praveena and Omar 2017) was also higher than the average level of Pb in Sri Lanka.

Health exposure risk

Estimated daily intake (EDI)

Approximately 284 g/day of rice per capita was adopted for calculation of the EDIs of Sri Lankans (Fordyce et al. 2000). In the present study, two adult household members (IR (day)2), average rice consumption 0.589 kg/day for one person) were employed, and their average body weight (bw) was

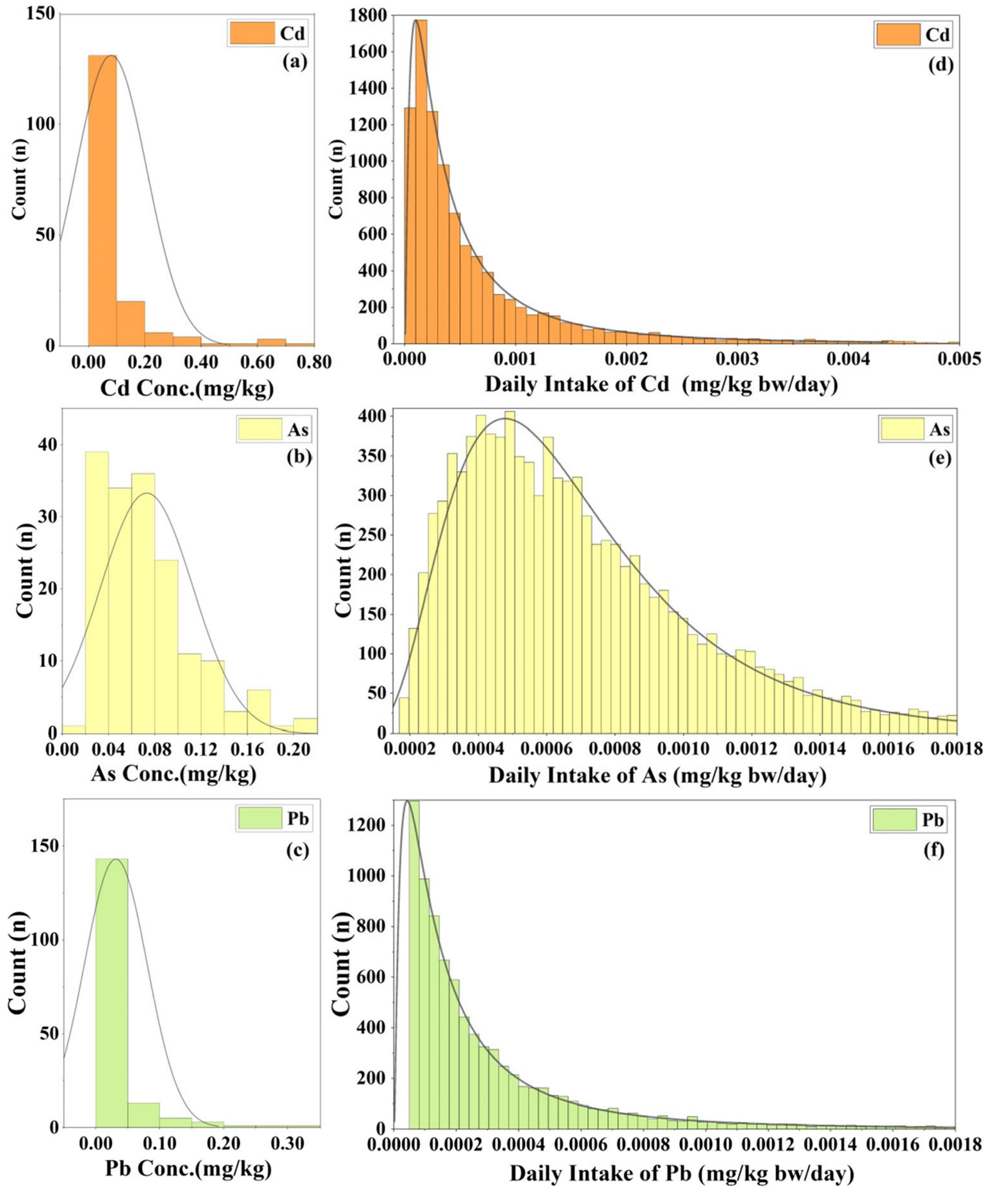

Fig. 2 Histogram of Cd, As, and Pb concentrations in rice and their risk assessment

considered as 60 kg. The EDI values for rice are listed in Table S4, and the percentiles of the estimated intake exposure to Cd, iAs, and Pb in Sri Lanka are summarized in Table 1. The *P* value of the percentile, such as P50, indicates that 50% of rice consumers are below this exposure level. The estimated exposures are given for the P50 and P75, P90, P95, P97.5, and

Table 1 Percentiles of estimated EDIs (μg/kg bw/day) in rice consumers exposed to Cd, iAs, and Pb, and corresponding HQs

Element		Average	P50	P75	P90	P95	P97.5	P99	P99.9
Cd	EDI	0.772	0.359	0.822	1.74	2.85	4.19	6.44	14.5
	HQ	0.780	0.363	0.835	1.74	2.79	4.11	6.35	16.0
iAs	EDI	0.490	0.427	0.619	0.853	1.010	1.16	1.38	1.52
	HQ	1.63	1.42	2.06	2.84	3.37	3.85	4.61	6.07
Pb	EDI	0.306	0.157	0.341	0.689	1.07	1.54	2.33	5.00
	HQ	0.086	0.044	0.096	0.193	0.298	0.432	0.654	1.39
	HI	2.496	1.827	2.991	4.773	6.458	8.392	11.614	23.46

P99 percentile levels, representing the median and highly exposed consumers for Cd, iAs, and Pb.

According to the Monte Carlo simulation results of 10,000 iterations, the EDIs of Cd, As, and Pb in commercial rice exhibited lognormal distributions, which reflected the frequency distribution characteristics (Fig. 2d, e, f). There has been considerable previous research relating to iAs. The contribution rate of iAs to total arsenic (tAs) shown on Table S5, with the iAs contribution range of south Asian countries' white rice being 63–82% (mean 69%). The average EDIs of Cd, iAs, and Pb consumption from rice were 0.772, 0.490, and 0.306 μg/kg bw/day, respectively. These were all below the PTWI values recommended by the FAO/WHO (1993, 2003), but iAs was above the RfD recommended by USEPA (USEPA 1991). The P97.5 EDIs of Cd, iAs, and Pb were 4.19, 1.16, and 1.54, μg/kg bw/day, respectively. Cd was triple the PTWI threshold, and iAs was approximately four times the RfD threshold, suggesting that Sri Lankans are confronted with exposure risk relating to Cd and As. China had similar results, with the 97.5th EDIs of Cd, Pb, and As ranging from 0.430 to 0.930, 0.770 to 1.78, and 1.69 to 3.87 μg/kg bw/day, respectively, which were well below the PTWIs suggested by JECFA (Qian et al. 2010).

In addition, no significant differences in EDIs were observed among provinces. However, the daily Cd intake from rice from Central province was slightly higher than that of other provinces (Fig. 3), suggesting that particular attention should be paid to residents in this area.

Potential health risks

The HQ is recognized as an accurate measurement for assessing the risk associated with HMs in foodstuffs. The average HQs of Cd, iAs, and Pb through rice consumption were 0.780, 1.63, and 0.086 (Table 1), respectively, decreasing in the order of iAs > Cd > Pb. The P99 HQs of Cd, iAs, and Pb were 6.35, 4.61, and 0.654, respectively, suggesting potential risk of Cd and iAs exposure from commercial rice.

The carcinogenic health risk value *R* of As via rice consumption ranged from 0.0002 to 0.0017, with an average of 0.0007; this exceeded the value recommended by the US EPA (0.0001) and indicated a carcinogenic health risk. The iAs has been mentioned in many toxicity studies on the carcinogenic effects of As. However, as the present study measured total As, of which iAs is the only one being estimated, the actual carcinogenic health risk from As in rice would be lower than that estimated in this study. Future studies are necessary to assess the carcinogenic effect of iAs via rice consumption in Sri Lanka.

The P90, P95, P97.5, and P99 values of HI estimated through rice consumption were 4.773, 6.458, 8.392, and 11.614, respectively (Table 1; Fig. 4). This indicated that high-exposure populations (less than 10%) may experience exposure health risks. The present results suggested that iAs and Cd were the major contributors to potential risk of non-carcinogenic health effects as their relative contributions to the HI were approximately 90%. Hence, HI values for the health risk of non-carcinogenic effects through rice consumption suggested that iAs and Cd are of primary importance.

Conclusions

In general, the average levels of Cd, As, and Pb in commercial rice from Sri Lanka were low; however, approximately 3.03%, 1.82%, and 1.21% of the samples contained high concentrations of Cd, As, and Pb, which exceeded the corresponding maximum Codex levels. The average concentrations

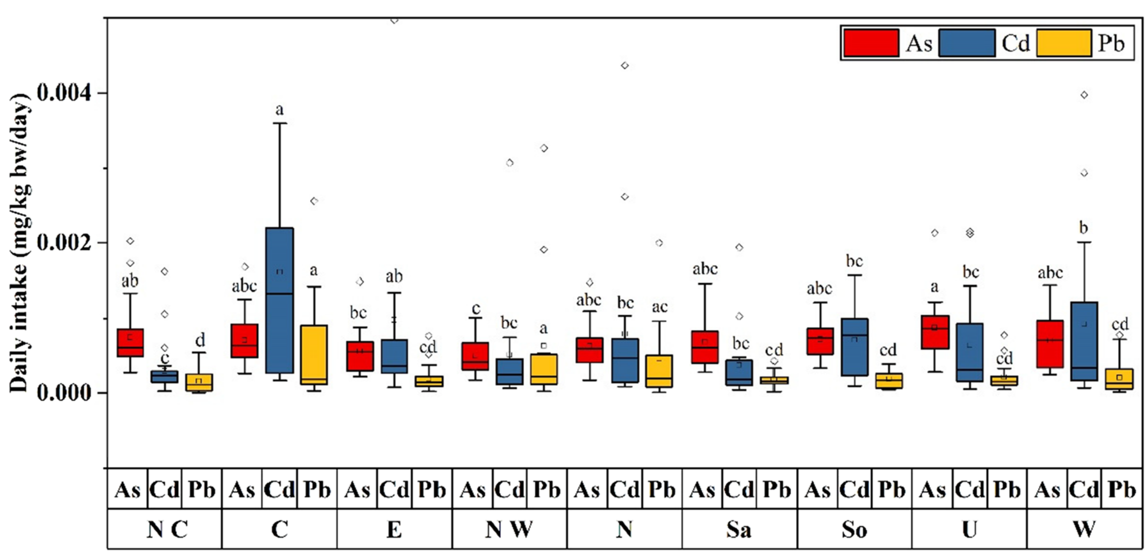

Fig. 3 Provincial Cd, As, and Pb exposure via rice intake in Sri Lanka

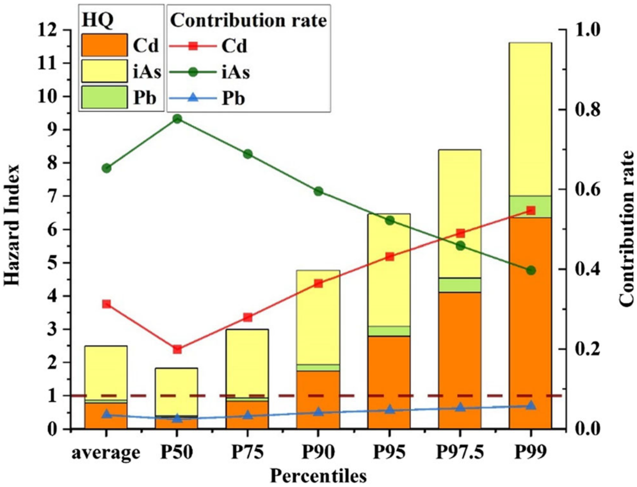

Fig. 4 Combined HIs and contribution rates of Cd, As, and Pb in rice

of Cd in rice from Northern, Central, and Western provinces were remarkably higher than those from North Central, Uva, and Sabaragamuwa provinces. The average concentrations of Pb in rice from Central, Northern, and North Western provinces were significantly higher than those from other provinces. The average EDIs of Cd (0.772 μg/kg bw/day), iAs (0.490 μg/kg bw/day), and Pb (0.306 μg/kg bw/day) from rice consumption were all below the PTWIs, but iAs was above the RfD recommended by the USEPA. Correspondingly, the estimated HQ values of individual toxins were all below 1, except iAs, while the HI values at the P90, P95, P97.5, and P99 percentiles were 4.773, 6.458, 8.392, and 11.614, respectively, implying that Sri Lankans may encounter a high risk of exceeding PTWI and RfD via rice consumption. Future studies of health risk assessments in Sri Lanka should consider multiple dietary sources.

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