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Economic benefit of ecological remediation of mercury pollution in southwest China 2007–2022

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ABSTRACT

Methylmercury (MeHg) exposure via rice consumption poses health risk to residents in mercury contaminated areas, such as the Wanshan Hg mining area (WSMA) in southwest China. Making use of the published data for WSMA, this study developed a database of rice MeHg concentrations for different villages in this region for the years of 2007, 2012, 2017, and 2019. The temporal changes of human MeHg exposure, health effects, and economic benefits under different ecological remediation measures were then assessed. Results from this study revealed a decrease of 3.88 μg/kg in rice MeHg concentration and a corresponding reduction of 0.039 μg/kg/d in probable daily intake of MeHg in 2019 compared to 2007 on regional average in the WSMA. Ecological remediation measures in this region resulted in the accumulated economic benefits of $38.7 million during 2007–2022, of which 84 % was from pollution source treatment and 16 % from planting structure adjustment. However, a flooding event in 2016 led to an economic loss of $2.43 million (0.38 % of regional total Gross Domestic Product). Planting structure adjustment generates the greatest economic benefits in the short term, whereas pollution source treatment maximizes economic benefits in the long term and prevents the perturbations from flooding event. These findings demonstrate the importance of ecological remediation measures in Hg polluted areas and provide the foundation for risk assessment of human MeHg exposure via rice consumption.

1. Introduction

MeHg health risks. Mercury (Hg) is recognized as one of the most toxic heavy metals (WHO, 2010) and a global pollutant that can be transported in the global atmosphere (Driscoll et al., 2013). Methylmercury (MeHg) is one of the most toxic forms of Hg that can cause serious human health issues upon exposure. For example, MeHg can cross the blood-placental barrier and fetal brain tissue upon exposure to pregnant individuals, leading to a decrease in intelligence quotient (IQ) and developmental delays in fetus (Clarkson et al., 2003; Debes et al., 2006; Ha et al., 2017), and MeHg exposure may result in cardiovascular impairment in adults (Roman et al., 2011). The global economic loss from human MeHg exposure was estimated to be as high as $117 billion annually, of which the United States and China suffered losses of $21 billion and $15 billion, respectively (Zhang et al., 2021).

MeHg exposure source. Fish consumption has traditionally been considered as the primary dietary source of human MeHg exposure (Mergler et al., 2007; Sunderland et al., 2018). However, recent studies have identified rice consumption as another significant dietary source (Liu et al., 2019), especially in Hg mining areas. For example, rice consumption contributed more than 96 % of total MeHg intake in several Hg mining areas (Feng et al., 2008; Zhang et al., 2010), because rice is a staple food for local residents while fish consumption is limited in these areas (Du et al., 2016). Rice MeHg concentration as high as 174 μg/kg has been reported in an Hg mining area (Qiu et al., 2008) and rice MeHg was originated from the methylation of Hg in paddy soil, with soil Hg being from dry and wet deposition and irrigation water impacted by Hg mining activities (Meng et al., 2011). Considering that rice is a staple food for half of the global population (FAO, 2018), the risks of human MeHg exposure through rice consumption deserve more attention,

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especially in Hg polluted regions.

Human MeHg exposure in WSMM. To gain some insights on the potential health effects from human MeHg exposure through rice consumption, the Wanshan Hg mining area (WSMA) in southwest China, which was the largest Hg mining area in this country (Qiu et al., 2005), was selected as a case study. Large scale Hg mining activities in WSMA have resulted in severe Hg pollution to local environment and serious MeHg contamination in local rice (Feng et al., 2008; Horvat et al., 2003), which may potentially threaten health of local residents. Human hair MeHg concentration can be used as an accurate biomarker of human MeHg exposure (Li et al., 2011). Existing studies in WSMA have demonstrated elevated hair MeHg concentrations for the local residents than the background population. For example, the average hair MeHg concentration for adults from three villages close to the Dashuixi mine waste pile reached 1.85 ± 1.16 μg/g (n = 94) (Feng et al., 2008), and that from seven villages along Xiaxi and Aozhai River reached 2.07 ± 1.79 μg/g (n = 168) (Li et al., 2015), while that from the background area was only 0.62 ± 0.23 μg/g (n = 24) (Feng et al., 2008). The average hair MeHg concentration for children from eight primary schools in four towns was 1.1 μg/g (n = 217) (Du et al., 2016), and the average hair Hg from two primary schools in two towns was 1.85 μg/g (n = 314) (Feng et al., 2020). Apparently, hair Hg concentrations in certain adults and children in WSMA exceeded the reference value of 1 μg/g, which is calculated from the reference dose for MeHg of 0.1 μg/kg/day set by the United States Environmental Protection Agency (USEPA) (USEPA, 1997). A comprehensive risk assessment of human MeHg exposure is needed in this area to evaluate the effects of existed ecological remediation measures and develop future remediation plans.

Ecological remediation. Ecological remediation measures involve pollution source treatment and planting structure adjustment (Spiegel and Veiga, 2005). Pollution source treatment aims to control Hg released into local environment, including managing mine waste piles, dredging rivers, eliminating artisanal and small-scale Hg mining, and standardizing production in Hg-related enterprises (Wang et al., 2012; Wang et al., 2020). Planting structure adjustment, a new model for solving the problem of Hg pollution in WSMA, mainly targets heavily Hg-contaminated farmland by banning crop (such as rice) cultivation to prevent MeHg accumulation. Existing paddy fields were converted into dry-land or vegetable greenhouse to reduce the ratio of local rice consumption (Xia et al., 2020). Remediation and management of mine waste piles in WSMA were completed during 2007–2011. The relevant management for Hg-related enterprises, rivers, and Hg contaminated soils were mainly implemented during 2016–2019. Zero-cost planting structure adjustment has been implemented on a large scale in WSMA since 2017. The health effects and economic benefits consequences from the extensive ecological remediation measures mentioned above are yet to be assessed in WSMA.

Health effects and economic benefits. Existing assessment studies on the health effects and economic benefits resulted from Hg pollution control worldwide (Bellanger et al., 2013; Pacyna et al., 2010; Sundseth et al., 2010; Zhang et al., 2017) are based on the probabilistic model developed by Rice et al. (2010), a study that characterized the plausible distribution of health effects and economic benefits in the U.S. population following the assumed reduction of MeHg exposure. The parameters in this model were based on fish consuming populations (Axelrad et al., 2007; Salonen et al., 1995; Shipp et al., 2000; Stern, 2005). However, at the same MeHg exposure dose, hair MeHg concentration in rice consuming population was 2.29 times of that in the fish consuming population (Li et al., 2015). Increase of 1 μg/g hair Hg resulted in 1 point of IQ loss in children with MeHg exposure via rice consumption, which was much higher than that via fish consumption (0.3 point) (Feng et al., 2020). Rice lacks specific micronutrients found in fish (e.g., n-3 long-chain polyunsaturated fatty acids (n-3LCPUFA), selenium (Se), and essential amino acids), which are identified as beneficial for health (Li et al., 2010). These nutrients may impact the absorption, distribution, and elimination of MeHg in human body and the toxicity of MeHg in different populations due to dietary modulation. Therefore, the risk assessment model of human MeHg exposure developed from fish consumption is not suitable in population through rice consumption.

Aim of this study. To quantify the effects of ecological remediation measures enforced in WSMA, this study established a cost-benefit analysis model based on the dose–response relationship between human MeHg exposure and health effects for rice consuming populations. The model comprises the following procedures: 1) Compilation of a database for rice MeHg concentrations in various villages at different time points; 2) Calculation of MeHg intake for different genders and age groups in various village; and 3) Analysis of the health effects and economic benefits associated with different remediation measures. This is the first study assessing health effects and economic benefits for rice consuming populations under Hg pollution control measures, which can provide scientific guidance for future policy making.

2. Methods

2.1. Study area

The WSMA is located in the eastern part of Guizhou province, southwest China. It consists of five towns (WS, GLP, HD, AZ, and XX) (Full names of the region abbreviations are shown in the Table S1.), covering a total area of 338 km^2 (Fig. 1). The population in the WSMA in 2022 was 62700, of which 75 % are rural population. It is a developing region with a Per Capita Gross Domestic Product (GDP) of 54,204 RMB (USD $8054) in 2022, which was about two-thirds of the national average in China (TBS, 2022). Traditionally, the rural residents in local villages were primarily engaged in rice cultivation and were self-sufficient in this food, while the town residents in the WS purchased rice from the market. In recent years, the awareness of the risks associated with MeHg in rice has actively promoted planting structure adjustment in the WSMA, which has reduced rice consumption by 10–70 % in some villages.

2.2. Ecological remediation measures

The WSMA, known as the “mercury capital” of China (Qiu et al., 2005), produced large amounts of mine wastes during mining and smelting activities, which subsequently deposited at the upstream of major rivers. The downstream banks of those rivers are the main rice growing areas, providing transport pathways for Hg migration from the mine waste piles to the paddy fields. Ecological remediation measures have been enforced by local government in the WSMA to protect the environment and human health (Table S2). Comprehensive management and remediation for six mine waste piles during 2007–2011 were mainly through covering the soil or cement and constructing slag dams. However, a large-scale flooding in 2016 collapsed some mine waste piles, including Lengfengdong and Dashuixi, flushing a substantial amount of mine waste into the rivers and farmlands. The flooding mainly impacted the villages situated in the lower terrain of AZ, XX, and HD town. Since the release of Action Plan for Soil Pollution Prevention and Control in 2016, local government have intensified efforts to enforce ecological remediation measures of Hg pollution. The mine waste piles damaged by the flooding and the surrounding historical mine waste have been re-treated, including the Sikeng, Lengfengdong, Meizixi, and Dapingkeng. The AZ and XX rivers were remediated through dredging of substrate sludge and construction of ecological barrier walls, and the substrate sludges were disposed with phytoremediation, passivation technology and low-temperature pyrolysis technology. Planting structure adjustment was implemented in highly Hg polluted areas by converting paddy fields into dry-land or vegetable greenhouses. The total area of greenhouses was larger than 100 acres in each village of DSL, XW, LongJ, DY, ZHS, GH, and XL. The planting structure adjustment has reduced the ratio of local rice intake, resulting in a decrease in human MeHg exposure.

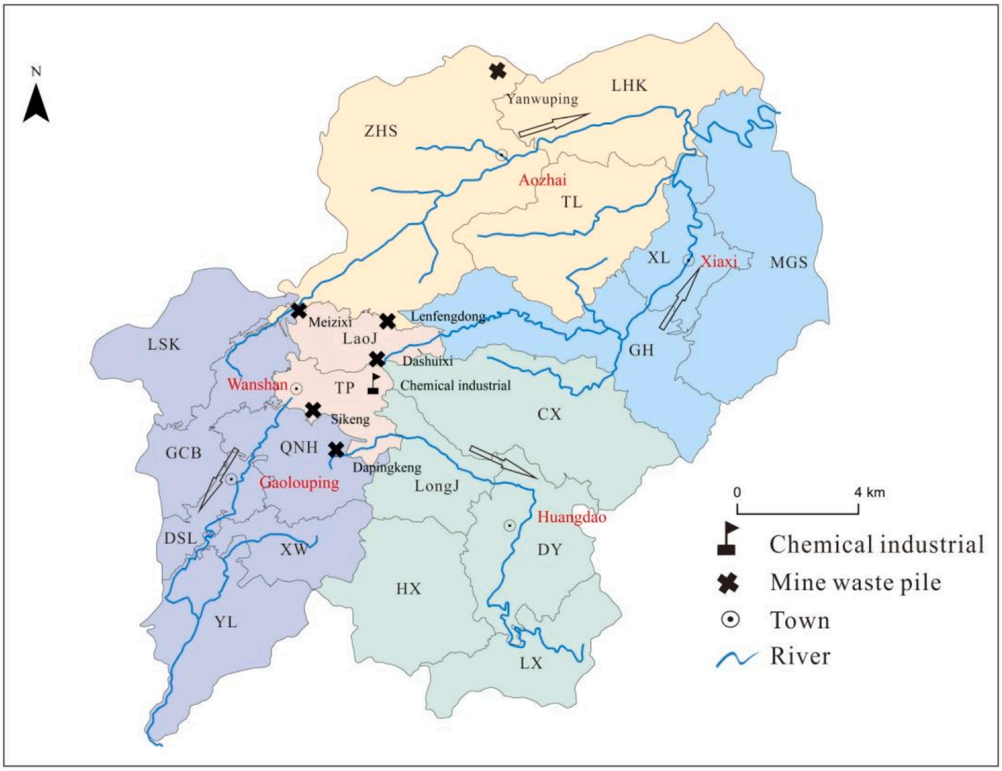

Fig. 1. Spatial location of study area.

2.3. Spatial and temporal distributions in rice MeHg

This study collected Hg concentration data in different environmental media in WSMA from 2000 to 2020 through databases such as China National Knowledge Infrastructure and Web of Science. Data with specific sampling times, sampling locations, and rice MeHg concentrations at each location were selected for analysis. Four time points (2007, 2012, 2017, and 2019) were identified based on existing studies and timelines of ecological remediation measures. The sampling points in those literatures were identified by the software of Omap, and were matched with the corresponding rice MeHg concentration by the software of ArcGIS. Rice MeHg concentrations in individual villages in the WSMA were then estimated using the inverse distance weighting method (Choi and Chong, 2022).

2.4. Ratio of local rice intake

This study considers human MeHg intake from both local rice and commercial rice bought from local market, with the latter originates from Anhui, Hubei, and other provinces. The MeHg concentration in the market rice is significantly lower than that of local rice in WSMA (Zhao et al., 2019). This approach significantly enhances the accuracy of the assessment of MeHg exposure level for local residents. The information related to the ratio of local rice intake was collected through interviews with local village committees, who understand basic information of local villages. This information includes details on the household population, number of households, resident population, area of rice cultivation, number of cultivating rice households, and area of greenhouses. The precision of the data was evaluated by comparing household population data obtained from the local village committees with those recorded by the local government and police station. It is assumed that rice consumed by households without rice cultivation is bought from the market. The ratio of local rice consumed in a village is determined from the percentage of households growing rice to the total number of households. In villages with greenhouses larger than 100 acres, the ratio of local rice consumed were significantly reduced due to planting structure adjustment (Table S3).

2.5. Probable daily intake (PDI) of MeHg

The PDIs of MeHg in WSMA were calculated based on the rice MeHg concentrations in different years and rice intake rates among residents in different groups (e.g., according to age and gender). Classification of rice sources were divided into local and market rice. The PDIs of MeHg were first calculated for the years of 2007, 2012, 2017, and 2019 using equation (1) (Du et al., 2016) and then linearly interpolated between these four time points to get values for the other years. After 2019, the PDIs of MeHg are assumed to remain constant (Zhang et al., 2023). Using the PDIs of MeHg in 2007 as a baseline, the changes of the PDIs of MeHg from 2008 to 2022 were calculated using equation (2).

$$\mathrm{PDI}_{ek} = (\mathrm{C}_{ka}\mathrm{f} + \mathrm{C}_{b}(1-\mathrm{f}) \times \mathrm{I}_{e}/\mathrm{BW}_{e} \tag{1}$$

$$\Delta\mathrm{PDI}_{i\ 2007} = \mathrm{PDI}_{i} - \mathrm{PDI}_{2007} \tag{2}$$

Where PDI is given in micrograms per kilogram of body weight (BW) per day (μg/kg/d). BW (kg) was mainly adopted from the National Physical Fitness Monitoring and I (daily intake of rice, kg/d) was obtained from the Guizhou Statistical Yearbook in 2007–2022 (Table S4). C_a and C_b (μg/kg) are the MeHg concentrations in local rice in individual villages and market rice, respectively. f is the ratio of local rice to total rice intake. ΔPDI denotes the change in human MeHg exposure level. The subscripts in the equation represent year (i), group (e), and region (k).

2.6. Health endpoint assessment

This study includes two health endpoints in health effects estimates: IQ decrement of fetus from prenatal exposure and fatal heart attack (FHA) in adults (Hu et al., 2021; Roman et al., 2011; Rothenberg et al., 2021). A linear dose–response relationship between MeHg intake and fetal IQ decrements recommended by the National Research Council (NRC) is adopted in this study (NRC, 2000). The following formula is used to calculate the changes of fetal IQ associated with the changes of PDI of MeHg in maternity:

$$\Delta IQ = \gamma \times \rho \times \Delta PDI \tag{3}$$

Where ΔIQ is the changes in IQ (points). The coefficient ρ (22.9 μg/g per μg/kg/day) is the converting factor from MeHg intake to hair MeHg concentration and γ (1 IQ points per μg/g) is the converting factor from hair MeHg concentration to IQ loss based on previous studies (Feng et al., 2020; Li et al., 2015).

A log-linear dose–response relationship between MeHg intake and FHA is used in the following equation (Guallar et al., 2002):

$$\Delta Cf = Cf \times \omega \times (1 - \exp(-\varphi \times \rho \times \Delta PDI) \tag{4}$$

Where ΔCf is the change in the deaths from FHA associated with MeHg exposure (deaths/ 100,000 persons), and Cf is the age-adjusted incidence of FHA among people aged ≥ 35 years, which is from the Tongren Center for Disease Control. φ (0.066 per μg/g) is the coefficient reflecting the relationship between hair MeHg levels and FHA risks (Salonen et al., 1995). The subjective coefficient ω (0.33) represents the probability of the causality of the associations, reflecting the substantial uncertainties due to limited epidemiological studies (Rice et al., 2010).

The value of IQ increase is estimated by avoiding the loss of productivity and thus a lower earning potential (Bellanger et al., 2013; Pichery et al., 2011). The cost of the IQ effect associated with MeHg exposure is expressed in Equation (5), which assumes a linear relationship between percentage change in lifetime earnings and change in IQ (Griffiths et al., 2007).

$$Vn = P_b \times \Delta IQ \times V_{IQ} \tag{5}$$

Where V_n is the economic value of reduced IQ, and P_b is the number of births, which is from the local government and police station (Table S5). V_{IQ} is the net loss per unit change in IQ point ($18,832 for 2008) (Bellanger et al., 2013), which is normalized by the GDP ratio between China and the U.S., adjusted by purchasing power parity (Sundseth et al., 2010). The V_{IQ} values in 2007–2022 are calculated from the growth rate of GDP in China.

The economic value from FHA deaths associated with MeHg exposure is calculated by a value of statistical life approach (Giang and Selin, 2016). Vc is calculated as:

$$Vc = P_{35} \times \Delta Cf \times V_{Cf} \tag{6}$$

Where Vc is the economic value of reducing the deaths of FHA caused by MeHg exposure, P_{35} is the number of individuals aged over 35 years (Table S5), and V_{Cf} is the value of reducing the FHA death ($630 million/death) (Giang and Selin, 2016).

2.7. Uncertainty analysis

Previous studies have employed bounding assumptions to assess uncertainties of influencing factors (Giang and Selin, 2016), or the variability of parameters in dose–response functions to specific uncertainties (Bellanger et al., 2013; Rice et al., 2010). The analysis is not based on a thorough assessment of uncertainties of the included parameters. Instead, key parameters have been identified including rice MeHg concentration, MeHg exposure level, dose–response parameters, and the economic parameters for health endpoints, which come from published uncertainty analysis and a default uncertainty value. Distributions of these key parameters are summarized in Table S6. The overall uncertainty is estimated through Monte Carlo simulation with 10,000 iterations using the software of Crystal Ball, which calculates the results using values of input parameters randomly selected from each corresponding distribution. The 2.5 % and 97.5 % percentiles of the calculated risk are considered as the overall uncertainty range.

3. Results and discussion

3.1. Spatial and temporal variations of rice MeHg concentration

Rice MeHg concentrations in different villages at different periods are presented in Fig. 2. The average MeHg concentration in local rice in 2007 was 11.8 ± 4.66 μg/kg (n = 95) (Table S3), which was significantly higher than those in the uncontaminated areas, such as European market rice (averaged at 1.91 μg/kg, ranging from 0.11-6.45 μg/kg) and Chinese market rice (averaged at 2.47 μg/kg, ranging from 0.13 to18.2 μg/kg), but lower than that in an Hg mining area in Xunyang, China (averaged at 17 μg/kg, ranging from 4.0 to 78 μg/kg) (Brombach et al., 2017; Li et al., 2012; Qiu et al., 2011). In 2007, 68.4 % (13/19) villages had average rice MeHg concentrations exceeding the threshold value (10.2 μg/kg) (Xu et al., 2020). The rice MeHg concentrations gradually decreased with increasing distance from the mine waste piles. For example, the average rice MeHg concentration in LongJ village (17.6 ± 5.92 μg/kg), which is close from DPK mine waste pile, was 76 % higher than that in downstream LX village (9.99 ± 1.48 μg/kg). Because the mine waste piles were deposited at the river upstream, Hg in mine waste can be easily leached by rain water, resulting in farmland Hg pollution by irrigation (Qiu et al., 2005). In the meantime, Hg in the mine wastes can also be emitted into the atmosphere and subsequently deposited onto the neighboring farmland soil through dry and wet deposition (Yin et al., 2016).

The average MeHg concentration in local rice in 2012 was 10.3 ± 4.30 μg/kg (n = 134), which was 12.7 % lower than that in 2007. Such a decrease in MeHg concentration can be attributed to the remediation of mine waste piles and planting structure adjustment of agricultural crops. The mine waste piles were repaired and closed in 2011, which significantly reduced Hg migration into the surrounding environment. In the same year, the “Survey on the Status of Hg Pollution of Soil and Agricultural Products in Wanshan” was conducted, which guided the adjustment of the cultivation structure of farmlands in severe Hg contaminated areas, such as banning the cultivation of Hg-rich crops. Consequently, the MeHg concentration in local rice decreased in many villages, e.g., by 6.36 μg/kg (36.2 %) in LongJ village compared to that in 2007. However, an upward trend (average increase of 28.3 %) was observed in GCB, DSL, and YL village in GLP town, probably due to the continuous atmospheric Hg emissions from a local chemical industry that is located upwind (northeast) of GLP town. Note that newly deposited Hg is more easily to be transformed into MeHg and accumulated in rice plants than other Hg with a long residence time in soil (Qiu et al., 2008).

The average MeHg concentration in local rice in 2017 was 13.2 ± 5.99 μg/kg (n = 122), which was 27.8 % higher compared to that in 2012. Such an increase in MeHg concentration may be attributed to the flooding event happened in 2016, which overridden the effects of ecological remediation measures. 84.2 % (16/19) of all the villages showed certain extents of rice MeHg concentration increase compared to 2012. The impact of the flooding event on rice MeHg concentration gradually decreased with increasing distance from the mine waste piles. For example, much larger increase in rice MeHg concentration was observed in LongJ village (9.08 μg/kg) than LX village (3.63 μg/kg) from 2012 to 2017. The mine waste piles collapsed by the flooding, which polluted river channels and farmlands along the riverbanks through floodwater. When heavy metals enter the soil, they are trapped by the ferromanganese oxide colloid film in the middle and lower soil layers, further worsen contamination (Ponting et al., 2021). It is noted that several villages, including TP, TL, and LHK, still showed decreased rice MeHg concentrations, likely due to their high elevations that minimized the impact of the flooding event. Again, the rising trends in rice MeHg concentration in the villages in GLP town should be caused by the Hg emissions from the chemical industry, as mentioned above, rather than the impact of flooding event.

The average MeHg concentration in local rice in 2019 was 7.90 ±

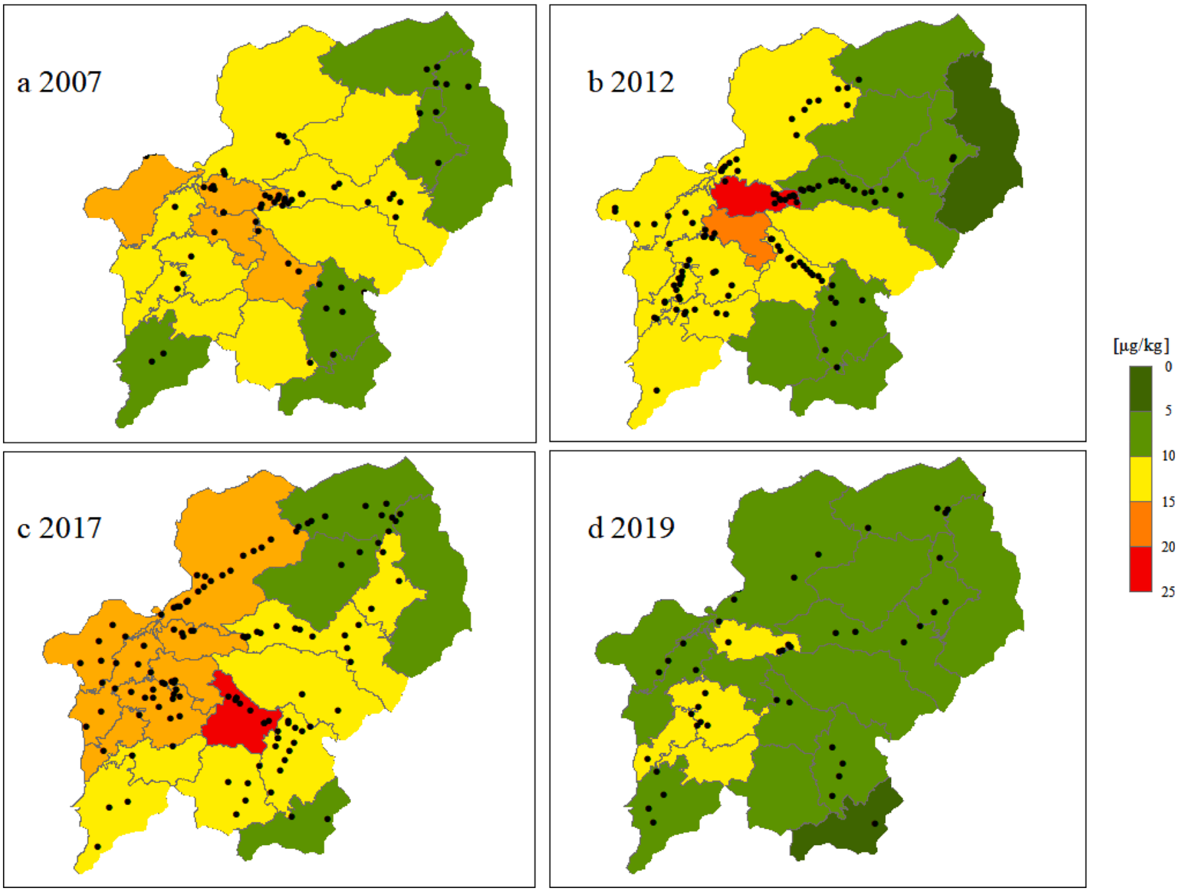

Fig. 2. Rice MeHg concentration at different villages and different periods in WSMA.

2.68 μg/kg (n = 74), which was 39.9 % lower than that in 2017. The decrease in MeHg concentration from 2017 to 2019 could be attributed to extensive ecological remediation measures enforced in this period. Large-scale treatments were taken for mine waste piles, rivers, chemical industries, and Hg contaminated soils and planting structure adjustment was exercised for farmlands with severe Hg contamination since the release of Action Plan for Soil Pollution Prevention and Control in 2016. The rice MeHg concentration in LongJ and ZHS village decreased by 13.5 μg/kg (66.2 %) and 7.67 μg/kg (45.4 %), respectively, from 2017 to 2019 due to combined effects from pollution source treatment and planting structure adjustment. In contrast, only a slight fluctuation (an increase of 1.8 %) in rice MeHg concentration was observed in LHK village, which is far away from the mine waste piles and is not seriously contaminated with MeHg. Note that the rice MeHg concentration in GCB village of GLP town decreased by 10.2 μg/kg (62.0 %), mainly due to the strict control of Hg emissions from a local chemical industry.

In summary, rice MeHg will keep balance and stable without ecological remediation measures. For instance, in the LHK Village, where did not implement any ecological remediation measures due to the long distance from the mine waste pile, the rice MeHg concentrations nearly kept stable from 2007 to 2019 (7.62 ± 2.87 μg/kg for 2007, 6.84 ± 0.86 μg/kg for 2012, 6.61 ± 0.62 μg/kg for 2017, and 6.73 ± 1.43 μg/kg for 2019). Consequently, with the implementation of the ecological remediation measures since 2007, rice MeHg concentrations in most villages showed a decreasing trend and fell into the safe limit in recent years, demonstrating the effectiveness of the existing ecological remediation measures in the study area.

3.2. PDI changes

The PDIs of MeHg via rice consumption in different age and gender groups in the WSMA are presented in Fig. 3**.** The PDIs of MeHg in this region were all lower than the reference dose (RfD) of 0.1 μg/kg/d recommended by the USEPA (USEPA, 1997). The PDIs of MeHg were slightly higher for males than females, but the differences were not significant. The PDIs of MeHg in fetus and children aged 0–9 years were 86.4 % higher than those aged > 60 years on average. Hence, fetus and children are sensitive population groups and more attention should be paid on their MeHg exposure and related health effects (Rothenberg et al., 2016).

The PDIs of MeHg in WSMA females aged 18–34 averaged at 0.052 μg/kg/d in 2007, which was higher than that of the American females (0.013 μg/kg/d) and the Norwegian population (0.043 μg/kg/d) even with high fish consumption (Abass et al., 2018; Carrington and Bolger, 2002), however, it was significantly lower than that of Japanese females (0.21 μg/kg/d) despite comparable hair MeHg concentrations between WSMA females (1.85 ± 1.16 μg/g) and Japanese females (1.73 μg/g, with a range of 0.49–5.82 μg/g) (Feng et al., 2008; Iwasaki et al., 2003). This indicated that health risks of human MeHg exposure are significantly higher from rice consumption than fish consumption under the same exposure dose condition. Therefore, a stricter standard should be established for human MeHg exposure via rice consumption. In the case of MeHg exposure from mixed rice and fish consumption, a combined risk assessment system should be used to avoid underestimation of health risks.

Looking at the PDIs of MeHg in WSMA in different years, a decrease of 59.3 % from 2007 to 2019 was observed. The PDIs of MeHg increased by only 5.92 % from 2012 to 2017, despite that the rice MeHg concentrations increased by 27.8 % due to the flooding event in this period, indicating other factors, besides rice MeHg concentration, also contributed to the variations of PDIs.

The villages in the study area were divided into different groups

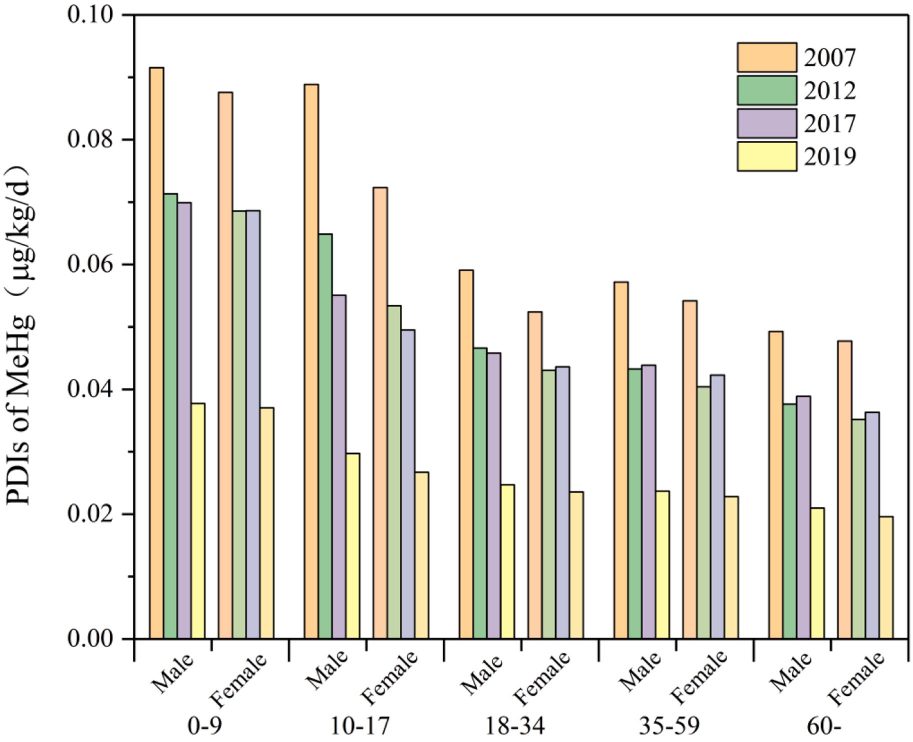

Fig. 3. The PDIs of MeHg in residents with different ages and sex in WSMA.

according to key influencing factors (Fig. 4). A significant relationship was found between PDI of MeHg and the ratio of local rice intake. For example, the PDIs of MeHg in TP village was only 85.5 % of that in YL village in 2007, despite the 171 % higher average rice MeHg concentration in the former than latter village. This could be attributed that TP was affiliated with the WS town, where has limited rice cultivation land, resulting only 20 % of the local rice consumed. Planting structure adjustment has decreased the ratio of local rice intake in 36.8 % (7/19) of all the villages, which in turn significantly reduced the PDIs of MeHg. For example, the PDIs of MeHg in DSL village decreased by 64.9 % from 2012 to 2017, despite a 64.4 % increase in rice MeHg concentration during this period. This was because the ratio of local rice intake by DSL residents decreased by 20 % due to conversion of 200 acres of rice lands to vegetable greenhouses.

The amount of daily rice intake and body weight are also important factors affecting the PDIs of MeHg. Consequentially, the PDIs of MeHg in LSK village decreased by 0.001 μg/kg/d in 2017 compared to 2012, while the rice MeHg concentration increased by 2.57 μg/kg. The daily

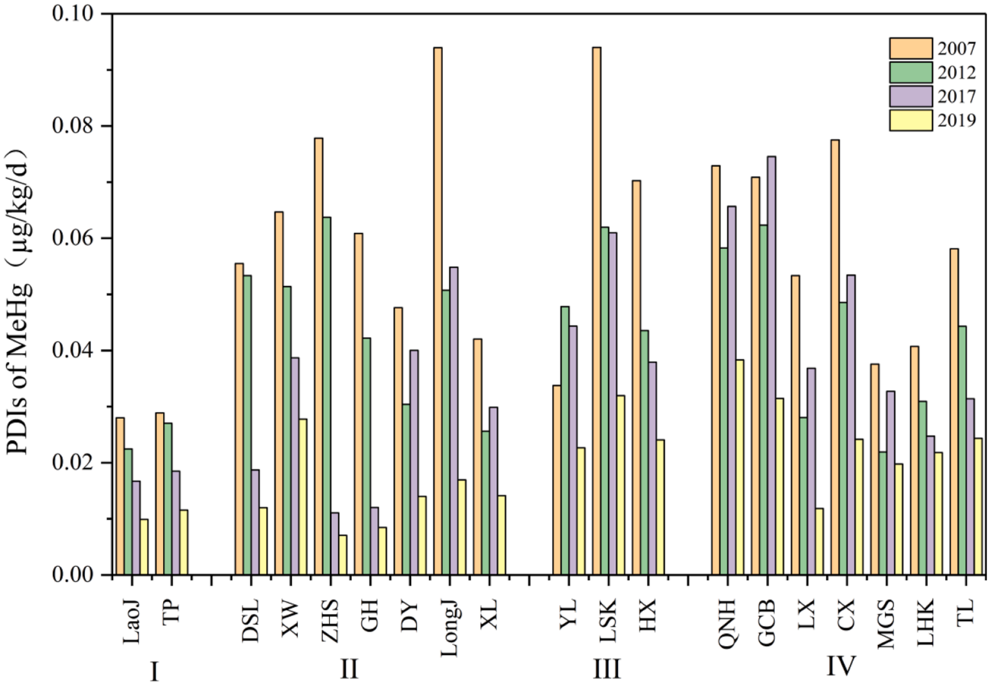

Fig. 4. PDIs of MeHg in residents from different villages in WSMA.

rice intake of local residents decreased from 314 g/d in 2007 to 208 g/d in 2019, while the average body weight increased from 58.7 kg to 64.3 kg (GBS, 2007; 2019).

In summary, the PDIs of MeHg for the residents in WSMA decreased significantly during the past two decades benefited from ecological remediation measures including pollution source treatment and planting structure adjustment, despite with perturbations from the flooding event in 2016. Reducing the ratio of local rice intake through planting structure adjustment was an important factor in controlling human MeHg exposure.

3.3. Health effects and economic benefits

The economic cost from human MeHg exposure via rice consumption was estimated to be \$3.50 million in WSMA in 2007, as a result of 1.11 points of per-fetus IQ decrement and 62 deaths per 100,000 from FHA (Figure S1). The average fetus IQ loss in the WSMA was significantly higher than the national average level in China (0.14 point) (Chen et al., 2019), despite that the PDI of MeHg of pregnant females in WSMA via rice consumption (0.052 μg/kg/d) was comparable with that via mixed fish and rice consumption (0.057 μg/kg/d) in 2011 in China (Liu et al., 2018). Such a result can be explained by the dose–response parameters in the risk assessment used in this study, which considered the fact that rice lacks a lot of micronutrients, which exist in fish (e.g., n-3 long-chain polyunsaturated fatty acid (n-3LCPUFA), selenium (Se), and essential amino acids) and are beneficial to brain development (Li et al., 2010). Hence, using MeHg exposure risk assessment model originally developed for fish consumption to assess rice consumption would underestimate IQ loss in fetus (Feng et al., 2020; Li et al., 2015).

The health effects and economic benefits (or costs) from changed human MeHg exposure were evaluated for three scenarios, namely, pollution source treatment, planting structure adjustment, and flooding event (Fig. 5). Pollution source treatment provided the greatest health effects and economic benefits during 2007–2022 by averting 339 deaths from FHA per 100,000 people and 0.3 points of per-fetus IQ decrement, with a total benefit of \$32.4 million in the whole WSMA region. Significantly spatial differences in health effects were observed among different villages. For example, a negative health effect was even observed in YL village, which was associated with increased rice MeHg concentration from 2007 to 2012. Planting structure adjustment brought significant health effects and economic benefits during 2017–2022 in some villages, which averted 48 deaths from FHA per 100,000 people and 0.5 points of per-fetus IQ decrement, equivalent to a total benefit of \$6.25 million. The health effects from planting structure adjustment were larger for those villages with higher rice MeHg concentrations. For instance, the average rice MeHg concentration in LongJ village was 38 % higher than that in DY village in 2017, but the economic benefit was 22 % higher at the same ratio of local rice intake. The economical cost of the flooding event was estimated to be \$2.43 million (25 deaths from FHA per 100,000 people and 0.28 points of per-fetus IQ decrement), accounting for 0.38 % of the Wanshan total GDP in 2016. This implies that the flooding event not only directly caused significant damage to the lives and properties of local residents, but also indirectly affected human health by increasing MeHg exposure levels.

Combining together the three scenarios mentioned above, the overall economic value was dominated by the reduced IQ loss (66 %), with only 34 % from the reduced FHA deaths (Table S7). This finding is different from that reported in a study conducted in the U.S. (Rice et al., 2010), which suggested the economic benefits of controlling human MeHg exposure were dominated by avoiding FHA (80 %). The higher contribution to the economic benefits from preventing IQ loss estimated in the present study was because the coefficient of IQ loss was based on the population of rice consuming (Feng et al., 2020), which is about three times of that of fish consumption (Axelrad et al., 2007). However, the coefficient of hair MeHg levels to FHA is adopted from results generated from fish consumption due to the lack of the actual coefficient from rice consumption. Therefore, future studies on the dose–response relationship between hair MeHg levels and FHA should be strengthened for the rice consumption population.

In general, the economic benefits of ecological remediation measures

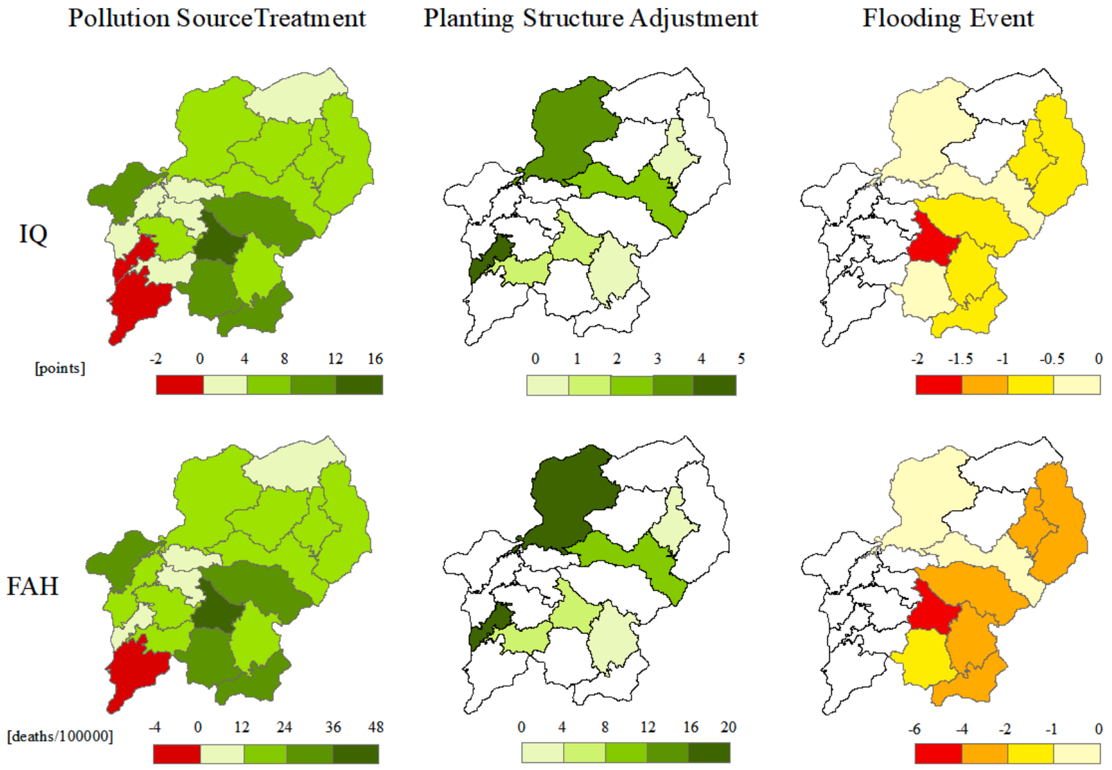

Fig. 5. Health effects of rice MeHg exposure under three scenarios in WSMA.

outweighed their economic costs. The pollution source treatments produced economic benefit of $32.4 million during 2007–2022 and will continue generating economic benefits in the future, despite the high costs of the treatments (a total of $35.9 million). Planting structure adjustment generated economic benefits of $6.25 million during 2017–2022 at no cost. However, the flooding event resulted in an economic loss of $2.43 million, which could have been prevented if enhanced management of pollution source treatments.

3.4. Uncertainty analysis

Uncertainty analysis results are presented in Fig. 6. The variability of the estimated cumulative economic benefits from ecological remediation measures is from $33.1 to 44.1 million (95 % confidence interval). The largest source of uncertainty came from the risk coefficient of hair MeHg levels to FHA (Virtanen et al., 2007), which is from $28.1 to 56.0 million, suggesting the need for more convincing evidence for the association between human MeHg exposure and FHA (Karagas et al., 2012). The coefficient of the PDI of MeHg to hair MeHg levels is also an important contributor to the uncertainty of the estimated economic benefits, because it has a wide range of the uncertainty (50 %) based on a comparison between model results and observed data of hair MeHg concentrations (Figure S2). The ratio of local rice intake is only a minor contributor to the uncertainty in the estimated economic benefits since this ratio only decreased in a few villages.

3.5. Strengths and limitations

The main strengths of the study include quantifying the costs and benefits of ecological remediation measures, which can inform decision-makers to understand the economic benefits of various approaches. This facilitates better decision-making for future remediation efforts and the promotion of relevant measures. Additionally, the study provides a foundation for assessing the risk of MeHg exposure via rice consumption based on specific rice consumption parameters.

The study also had several limitations. First, the ecological remediation is an ongoing process, and the economic benefits have long-term impacts. Therefore, this study only provides a preliminary assessment of the economic benefits associated with the ecological remediation measures. Second, the reliability in modeling human MeHg exposure and health risks is limited by current knowledge and available data, such as the dose–response relationships between human MeHg exposure and health effects. The dose–response parameters for IQ to MeHg exposure in populations via rice consumption differ significantly from fish consumption. However, the dose–response relationship for MeHg exposure and FHA in rice-consuming populations still relies on epidemiological study results based on fish-consuming populations. Reproductive outcomes and immune system effects are also important health endpoints caused by severe MeHg exposure (Mergler et al., 2007), which are not considered in this study due to the limited epidemiological data (Eagles-Smith et al., 2018). Thus, future studies should focus on comprehensive health risk assessments that consider all major health effects and strengthen research on the dose–response relationship between FHA and MeHg exposure via rice consumption.

4. Conclusions

This study established a database of rice MeHg concentrations in various villages at different time points, evaluated the changes of human MeHg exposure levels, and estimated economic benefits associated health effects under different ecological remediation measures in the WSMA. The results indicated that, after the implementation of ecological remediation measures, rice MeHg concentrations in most villages declined to levels below the safe limit and the health risks of human MeHg exposure have been efficiently controlled. The cumulative economic benefit exceeded the total investment of ecological remediation measures during 2007–2022, and additional economic benefits are expected in future years from the existing ecological remediation measures. Planting structure adjustment reduced health risk rapidly and maximized economic benefits in a short term at no cost. Pollution source treatment, although with high costs, has long-term impacts on addressing the root causes of Hg pollution related issues and can also prevent disturbances from future flooding events. The results obtained in this study provide scientific guidance for establishing ecological remediation in Hg polluted areas. This is the first study comprehensively assessing health effects and economic benefits for the rice consumption population in Hg polluted region, which provides the foundation for risk assessment of human MeHg exposure via rice consumption.

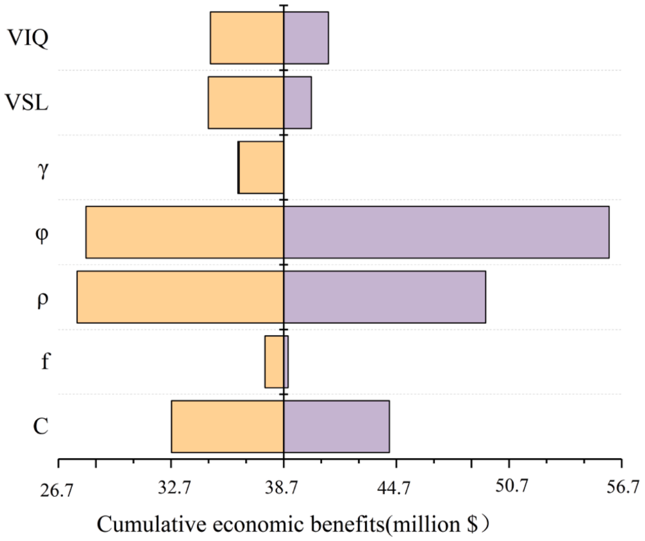

Fig. 6. Uncertainty of cumulative economic benefits caused by different parameters.

CRediT authorship contribution statement

Huifang Jiang: Writing – review & editing, Writing – original draft, Project administration, Methodology, Investigation, Data curation. **Junyao Yan:** Writing – review & editing, Methodology, Investigation. **Ruolan Li:** Writing – review & editing, Methodology, Investigation. **Shaochen Yang:** Writing – review & editing, Methodology, Investigation. **Guopei Huang:** Writing – review & editing, Supervision, Resources, Methodology. **Wenjuan Wang:** Writing – review & editing, Supervision. **Yanxu Zhang:** Writing – review & editing, Supervision. **Ping Li:** Writing – review & editing, Supervision, Methodology, Investigation, Data curation, Conceptualization. **Xinbin Feng:** Writing – review & editing, Supervision, Data curation, Conceptualization.

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.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envint.2024.108792.

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