DOCTORAL DISSERTATION

Health Impact and Economic Analysis of Soil Remediation for a Heavy Metal Polluted Mining Area in China, Under Normal and Accidental Scenarios

2023.1

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ABSTRACT

Previous research about heavy metal pollution in China has revealed high health risk for the residents. However, traditional risk assessment indicators overlook disease severity, which lead to a less comprehensive estimation. To reduce the risk, various alternative technologies have been developed for remediation, which differs in efficiency at site scale. Information on economic feasibility is significantly required for more efficient investment. However, such information is limited in China. Additionally, studies often overlook health aspect, which is one of the most important targets of risk management. Considering the accidental release of heavy metal, most studies focus on mechanism and consequence of the accident, information on remediation process is even more limited.

The major purpose of this research is to provide field-scale information on economic feasibility of soil remediation. considering this purpose, the thesis is divided into 3 chapters with the following objectives respectively: (Chapter 2) Develop an estimation method for health impact assessment with proper indicator; (Chapter 3) Estimate the cost-effectiveness for different soil remediation alternatives and identification of the potentially best alternative; (Chapter 4) Estimate the health impact of soil heavy metal after accidental release and assess the economic feasibility of soil remediation process.

In Chapter 2, Mercury (Hg) concentration in exposure media in China is collected with a literature survey. The exposure dose is based on ingestion, inhalation and dermal exposure. Disability-Adjusted Life Years (DALY) is quantified based on exposure dose and dose-response relationship. As a result, the total average DALY caused by lifetime Hg intake is 6.63×10^{-6} years for areas without specific Hg sources and 9.96×10^{-4} years for areas with specific Hg sources in China. The DALY level in China is much higher than 6.91×10^{-7} years in Japan. Ingestion exposure of methylmercury contributes over 90% of DALY in both China and Japan, indicating the priority in pollution control. This assessment proved DALY as a proper indicator for health impact quantification. This methodology is applied for the following other two sections.

In Chapter 3, we choose Jinding mining area as target site and estimated the DALY level before and after remediation. Four soil remediation alternatives are compared based on economic feasibility and health benefit and the potentially best alternative is identified. The reduction rate of DALY is over 96% for all the remediation alternatives except soil washing, indicating the remediation is effective. The net benefit is estimated as 0.70, 1.00, 1.11, and 1.14 (billion \$) for soil replacement, soil washing, stabilization and phytoremediation, respectively. Considering removal efficiency and economic feasibility, phytoremediation is the potentially best alternative.

In Chapter 4, a specific agricultural land in Jinding mining area is selected. Health impact and soil remediation is discussed under a hypothetical tailing dam failure scenario. The health impact under accidental scenario is estimated and compared with normal scenario. A cost-benefit analysis discusses the economic feasibility of the remediation. The analysis shows that the DALY caused by the dam failure will be 41.8 times higher than normal scenario. The accidental release of heavy metal will cause a high influence. Remediation reduced 96.83% of the total DALY caused by the dam failure, indicating an effective remediation process. Cost-benefit analysis showed a negative benefit rate of -81.41%. The remediation is not economically beneficial under accidental scenario. In order to avoid the unbeneficial remediation process, the dam failure should be prevented before happening.

As for the conclusion and implication of this research, for Jinding mining area, under normal scenario, the remediation is effective, and phytoremediation is identified as the potentially best alternative. Under accidental scenario, remediation is effective but not economically beneficial, indicating higher importance of accident prevention. For the research field of soil remediation, this research provides site-scale information on economic feasibility of soil remediation. DALY is a proper indicator for health impact and is suitable for cost-benefit analysis. The methodology for health impact estimation developed in this research can be applied in future researches.

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CHAPTER 1: INTRODUCTION

1.1 Research background

1.1.1 Soil heavy metal pollution state in mining areas in China

1.1.1.1 Heavy metal risk

Metals with density over 5 g/cm^3 are defined as heavy metals (Oves et al., 2012). Heavy metal in the environment can enter human body via various exposure routes such as oral ingestion, dermal contact, and inhalation (Kolo et al., 2018; Shi et al., 2011). Most heavy metals are toxic to human body. Even though some heavy metals are necessary nutrients at a low concentration, some extremely toxic heavy metals are harmful even at a low concentration, such as Mercury (Hg), Lead (Pb) and Cadmium (Cd). Arsenic (As) is a metalloid element, but due to its property and toxicity, As is often discussed together with other heavy metal elements (Chen et al., 1999). If human body is exposed to heavy metals, they may cause severe health problems. For example, Hg toxicity threats nervous system and may also cause kidney damage (Kim et al., 2016); Pb is harmful to the nervous system, especially for children, Pb may also cause damage in kidney and liver (Silver et al., 2016); Cd can cause renal disease and various carcinogenic health problems including breast and lung cancer (Buha et al., 2017; Ju et al., 2017; Venza et al., 2015). Exposure to As can result in disorder in skin and cardiovascular system, which may also be carcinogenic (Chen et al., 1995; Mazumder et al., 2000; Huang et al., 2015). Heavy metal in the soil will also reduce the quality of soil and be harmful to the environment (Alloway, 2012; Briffa et al., 2020; Wei et al., 2016). Therefore, controlling and reducing heavy metal in the environment is of significant importance to both human and the ecosystem. Presently, monitoring and controlling heavy metal contamination is a prioritized environment issue worldwide (Lacarce et al., 2012; Schneider et al., 2016).

1.1.1.2 Heavy metal pollution state in China and anthropogenic sources

Heavy metal contamination has been a severe problem in China due to the rapid development of industry and economy (Duan et al., 2016; Zhang et al., 2016; Zhong et al., 2018). According to the Bulletin of the National Survey of Soil Pollution, the heavy metals Cd, Hg, As, Cu, Pb and Zn exceed the standards of 7.0%, 1.6%, 2.7%, 2.1%, 1.5% and 0.9%, respectively(Ministry of Ecology and Environment of the People’s Republic of China, 2014). Wu et al. (2022) conducted a provincial level of heavy metal contamination analysis, the result indicated that soil heavy metal contamination in China is widely distributed in various provinces. Pollution in southern China is more severe than northern China. Cd contamination is the most serious. Several pollution hotspots (provinces) are identified, the hotspots of Pb and Cd pollution are mainly distributed in mid and south China. Yang et al. (2018) examined the heavy metal concentration in 402 industrial regions and 1041 agricultural regions and found that heavy metals Pb, Cd, Hg, Cr and As exceeds the Grade II environmental quality standard for soils in China (GB15618-1995). The Cd contamination is the most severe, Cd concentration in all the examined sites exceeds the Grade II standard. The mean Cd concentration is 79.2 times higher than the Grade II standard. Hg, As and Pb also have high exceedance rate, especially in industrial regions. Risk assessment revealed that Cd contamination is the most important for ecological risk, Pb and As contamination is highly responsible for non-carcinogenic health risk and carcinogenic health risk, respectively.

1.1.1.3 Heavy metal pollution state in mining and smelting areas

Heavy metal in soil can be generated by various sources, such as mining activities, metal smelting and processing and improper fertilizer application and irrigation (Zhang & Yang, 2020). As is reported by Wu et al. (2022), mining process and factory processing is the most important reason for heavy metal pollution in the most heavily polluted areas. Especially the development of non-ferrous metal mining and smelting produced serious heavy metal pollution. Nonferrous metals are crucial material for industrialization and is one of the most important parts for economy system in China (Song et al., 2019; Xue et al., 2022). China is one of the biggest nonferrous metal producers in the world (Li et al., 2014; Shen et al., 2017). In 2018 alone, China produced approximately 58 million tons of various nonferrous metals, including 5.11 million tons of Pb and 9.03 million tons of Zn (National Bureau of Statistics, 2019). The massive production of nonferrous metal caused a great amount of heavy metal pollutants to be released into the environment. Ma et al. (2013) reported about 88t of Cd released by the national nonferrous metal and mining process. Cai et al. (2021) investigated the heavy metal contamination in agricultural soil of 88 sites around mining and smelting area. The result showed a high health risk for residents from the contaminated soil, especially for children. Several other studies supported this result (Chen et al., 2015; Man et al., 2010). Adnan et al. (2022) reported heavy Hg, As, Pb and Cd pollution in soil of most smelting sites in China.

Pb/Zn mining and smelting is one of the most important parts of the nonferrous metal industry. Researchers have reported heavy metal pollutions such as As, Cd and Pb to be found in Pb/Zn mining and smelting areas in Hunan, Guizhou, Guangdong, and Yunnan (Williams et al., 2009; Xie et al., 2014; Zhao et al., 2015). (Shen et al., 2017) reported high Cd and Zn concentration in the vicinity of a Pb/Zn smelter in Shaanxi province. (Šajn et al., 2013) studied a Pb/Zn smelter and found that the pollution area reached approximately 150 m^2 with concentration of various heavy metals exceeding their intervention values. Kan et al. (2021) analyzed the mining tailings of 34 Pb/Zn mining sites and assessed the soil contamination. The result revealed a high heavy metal concentration, the concentration of As, Cd, Hg and Pb all exceeded the Grade II environmental quality standard. The health risk assessment also showed high non-carcinogenic risk caused by As and Pb and high carcinogenic risk caused by As and Cu. Luo et al. (2023) analyzed over 2300 samples of Pb/Zn smelters across China, the mean concentration of Pb, Zn, As and Cd extremely exceeded the background value of Chinese soil. From these studies we can conclude that heavy metal pollution in Pb/Zn mining and smelting areas in China is severe. Therefore, it is significant to understand the pollution state and potential in soils around Pb/Zn mining areas, as well as providing insights of potential and feasibility of remediation strategies for those contaminated soils (Li et al., 2020; Xu et al., 2021).

1.1.2 Soil heavy metal pollution remediation alternatives

To reduce the influence of heavy metal pollution on the environment, soil remediation is required for the polluted sites. Presently, various technologies have been developed for soil remediation. These technologies utilize different mechanisms to remove heavy metal from soil or stabilize heavy metal into a less bioavailable state, thus preventing the pollutants from entering the environment. Compared with other types of pollutants, HMs could not be destroyed biologically but could be only transformed from one oxidation state or organic complex to another, becoming either more soluble, less toxic, less bioavailable, or volatilized to be treated (Chen & Li, 2018; Garbisu & Alkorta, 2001). Thus, the technologies often used for HM treatments are soil replacement (excavation) (Lambert & Leven, 2000), stabilization/solidification, soil washing, and phytoremediation (Wuana & Okieimen, 2011).

Soil remediation technologies are usually categorized into 3 types. Physical remediation, chemical remediation, and bioremediation.

Physical remediation mainly refers to technologies that separate heavy metal from soil by physical process (Wu et al., 2022). Soil replacement (excavation) is a typical physical remediation technology that directly remove the heavy metal concentrated soil from the site to monitored land fill or treatment facilities. In some cases, fresh uncontaminated soil or materials are refilled to the site to keep the function of the land. This method is the major remediation alternative before 1984 (Gong et al., 2018). This methodology is effective and can completely remove the heavy metal from the target site. However, the excavated soil may require further treatment and long-term monitoring to be safe to environment. Also the cost of soil excavation is relatively high, the average cost of soil excavation reported by FRTR is \$270 to \$460 per ton of soil (FRTR, 2012).

Chemical mainly utilizes the chemical property of soil and heavy metals. Usually chemical agent is applied to the soil to realize the reduction or stabilization of heavy metal in soil. Soil washing is a technology that removes heavy metal from soil by leaching with various chemical reagents or extractants (Chen et al., 2016; Park & Son, 2017). Soil is normally dug out and treated with the leaching solution ex-situ. The heavy metals are transferred into liquid phase via chemical dissolution, ion exchange, chelation or desorption and separated from the soil during the leaching process (Ferraro et al., 2016). After the treatment, the treated soil can be refilled to the original site. Soil washing can permanently remove the heavy metals from the soil and is relatively less time consuming (Park & Son, 2017). But the leaching solution is concentrated with heavy metal after the remediation, the solution may require further treatment before released to the environment. Stabilization/solidification is another chemical remediation technology which aims at reducing the bio-chemical availability of heavy metals in soil instead of removing them. Stabilization/solidification refers to a series of technologies, where solidification means encapsuling pollutants in a monolithic solid of high structural integrity (Khan et al., 2004) and stabilization means amending the contaminated soil with chemical reagents to reduce the availability of heavy metals (Chen et al., 2009). Stabilization/solidification is a typical in-situ treatment method, with relatively low cost and simple operation. However, the heavy metal is not completely removed from the soil, indicating the possibility of being activated again if the soil property changes.

Bioremediation reduces the heavy metal in soil by utilizing the ability of organisms or microorganisms. Phytoremediation makes use of plants to extract or stabilize the heavy metal in the contaminated soil. The efficiency of phytoremediation is influenced by various factors, such as climate, soil property and heavy metal species (Khalid et al., 2017). Presently many plant species are found to be able to extract certain types of heavy metal from soil, at the same time being able to tolerate the high heavy metal level. Even though most phytoremediation studies are done in laboratory and only few field studies have been carried out, because of the low cost and environmental-friendliness, phytoremediation is a promising technology in the future. The main disadvantage of phytoremediation is the time consumption. Unlike other remediation alternatives which can normally be applied within 1 year, phytoremediation usually takes several years.

1.1.3 Economic feasibility of soil remediation

Soil remediation is a costly and complicated process. Considering site-specific situation, it is not always economically feasible (Beinat & Nijkamp, 1998; Ciccu et al., 2003). The remediation alternatives differ in cost, applicability, and effectiveness. Chen & Li (2018) compared the general cost-effectiveness of several types of soil HM remediation technologies and discovered that various factors of the contaminated site, such as types of HMs and contamination level, will influence the effectiveness of soil remediation. Therefore, to pursue better efficiency of the investment, field-scale information about the cost and benefit of soil remediation is significantly required by decision-makers.

Even though economic feasibility is an important issue, studies regarding the economic aspect of soil remediation with heavy metal(loid)s are scarce (Khalid et al., 2017). In China, most studies on soil remediation focused on removal efficiency and remained at the laboratory stage, and only limited field scale reports are available (Gong et al., 2018). Many field-scale studies focused on single remediation alternatives such as phytoremediation experiments (Wan et al., 2016; Wei et al., 2021; Zhang et al., 2021). Therefore, discussing the financial feasibility of different remediation alternatives on a field scale is required.

Cost-benefit analysis (CBA) is a financial assessment tool that is heavily applied for decision-making. However, limited research performed CBA on a field scale for soil remediation (Huysegoms et al., 2019b). Most research on the financial aspect of soil remediation technologies focused on other types of benefits except health benefits, such as the value of land and properties (Lavee et al., 2012; Ospanbayeva & Wang, 2020), the value of cash crops and human income (Wan et al., 2016; Zhang et al., 2021). However, many of these studies often overlook health impact, one of the most important protection targets of risk management (Swartjes, 2011). There are a few studies that included the assessment of health benefits. For example, Sohail et al. (2020) conducted a phytoremediation experiment at an agriculture site and assessed the health risk reduced by the remediation, but the health benefit was not discussed from a financial point of view; Huysegoms et al. (2018, 2019a) conducted field studies of soil remediation at industrial sites, but this type of assessment has not been applied to agriculture soil. Considering the high transferability and the unique biological toxicity of heavy HMs in the soil-crop system, the health risk of HM contaminated agriculture soil deserves stricter management (Cai et al., 2021). Therefore, information on monetized health benefits obtained by remediation of HM contaminated agriculture soil is required for this field.

1.1.4 Cost-benefit analysis under normal scenarios and accidental scenarios

Cost-benefit analysis have been conducted for soil remediation under normal scenarios, but there are few studies about soil remediation for accidents. The most important technical accident in mining areas is the failure of tailing dams. A tailing dam is a typical structure built for storing tailing and waste in mining areas, preventing the HM containing tailing from causing environmental problems. Due to the large amount of tailing and wastewater stored, the consequences of tailing dam failure can be severe. For example, the break of the Aznalcóllar tailing dam on Apr. 25th, 1998, in Spain, flooded approximately 4600 hectares of land (Hudson-Edwards et al., 2003). Tailing dam failure requires immediate remediation actions. Otherwise, the tailings will possibly influence the land continuously. In China, in 1985, a tailing dam failure happened in a Pb/Zn mine in Chenzhou (Liu et al., 2015). The clean-up action was only applied to limited areas. Research showed that the HM in the soil in the untreated areas extremely exceeded the Chinese agricultural soil standards even 17 years after the failure. Therefore, the remediation and recovery process after tailing dam failure is important for tailing dam management. Understanding the remediation process for tailing dam failure is significant for future management and policymaking, including economic understanding.

Researchers have discussed the mechanism and consequences of past tailing dam failure events. Such as seismic behavior of the dam structure (Ishihara et al., 2015) and long-term impact on the marine environment (Sá et al., 2021). However, for future management of tailing dams, predictive studies which provide a quantitative risk assessment of potential tailing dam failure are required. Predictive risk assessment of tailing dam failure focused on two aspects, the likelihood of a tailing dam failure happening, and the risk caused by the tailings released into the environment. Most existing research focused on the previous aspect. For example, Nišiç et al. (2018; 2021) estimated the risk of tailing dam failure in two Serbia metal mines with a hierarchy risk matrix considering the likelihood and consequence severity of the tailing dam failure. Liu et al. (2019) developed a Bayesian network-based simulation model to assess the water pollution caused by tailing dam failure in China. Compared to these researches, some other risk assessments extended further to consequences after the failure, such as health risk and economic loss. (Barcelos et al. (2020) developed a hypothetical dam failure event in a gold mine in Brazil and conducted a quantitative risk assessment accordingly. As a result, As, Cd and Zn showed potential human health risks. But the author did not include economic considerations. In China, Liu et al. (2020) assessed the risk of the tailing dam collapse in a waste slag site. The author estimated the vulnerability of buildings and health risk assessment for the residents under a failure scenario. Financial loss is evaluated based on land price, clean-up budget, placement of residents, etc. Health impact was not included.

1.1.5 Disability adjusted life years (DALY)

Risk assessment was a popular research field and indicators have been developed to quantify the health risk caused by heavy metals. The most widely used indicator is Hazard Quotient (HQ) developed by US EPA (1989a). This indicator quantifies the health risk by comparing the exposure dose and the reference dose (RfD). The higher the HQ is, the higher the exposure dose compared to RfD. If HQ exceeds 1, the risk is considered intolerable. Another widely used indicator is Margin of exposure (MOE) (Benford et al., 2010). MOE is defined as the ratio between the lower side of benchmark dose (BMDL) and the estimated exposure dose. If the BMDL is not available, other parameters such as no observed effect levels (NOEL), no observed adverse effect levels (NOAEL) or lowest observed adverse effect levels (LOAEL) can be applied instead. These indicators reveal the extent of exposure and quantify the health risk level. However, these indicators focus on the possibility (risk) of the health impact, the effect of end-point disease is not included.

In this research, Disability-adjusted life years (DALY) is applied to quantify the health risk and impact caused by Hg exposure. DALY is developed by Murray (1994). DALY is a health impact indicator that quantifies the health risk by combining the dose–response relationship with the impact of diseases or disabilities on the human body. **Fig 1-1** shows the concept diagram of DALY. Compared with traditional indicators applied in CBA for soil remediation (Huysegoms et al., 2018; Stewart & Purucker, 2011), DALY is expected to create a stronger connection between exposure dose and endpoint effect. There is a similar framework: Quality-Adjusted Life Years (QALY), which also assess the health impact on life, but the origins of disability and quality of life weights are different. QALY tends to focus on life adjustments within one year or a single period, while DALY focus on longer time span such as lifetime. In this research, remediation is expected to control heavy metal pollution for a long period, so we choose DALY for its longer assessment span.

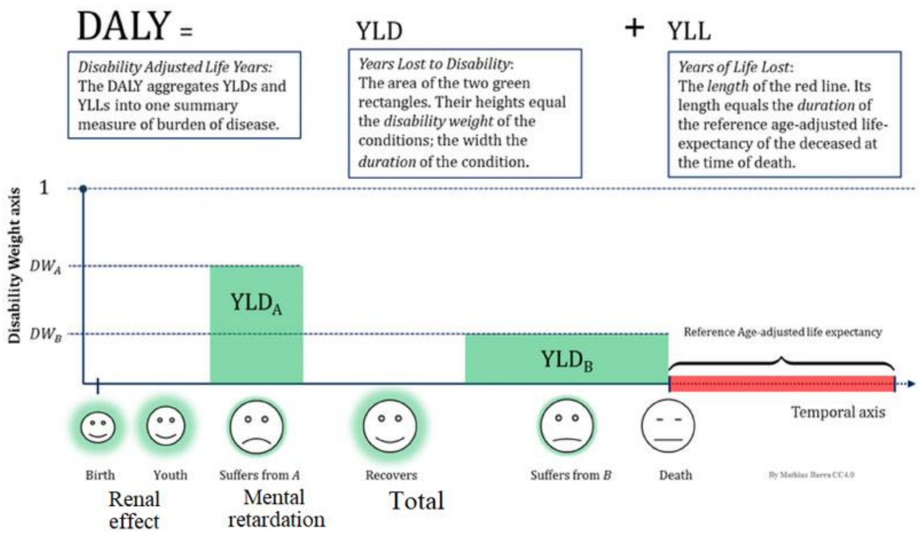

Fig 1-1. Concept diagram of Disability-Adjusted Life Years (DALY)

DALY has been frequently applied in cost-benefit analysis. The World Health Organization (WHO) reported the loss of $ 108,600 per DALY considering the financial loss during treatment, post-treatment period and due to premature death (Lee et al., 2009; WHO, 2001). With the average financial burden, DALY is able to monetize the health impact into financial loss, which can be further applied to cost-benefit analysis. Therefore, DALY is considered the proper indicator for health impact in this research.

1.2 Research objectives

1.2.1 Development of quantitative assessment methodology for DALY

Traditional health impact indicators such as Hazard Quotient (HQ) and Margin of exposure (MOE) quantify the health impact by simply comparing the exposure dose with certain thresholds. This comparison provides the possibility of adverse effect but overlooks the actual severity of the adverse effect. The possibility of disease is difficult to be applied directly in economic analysis, the health impact needs to be monetized to compare to other types of benefit and cost of soil remediation. DALY, as demonstrated above, can fill the gaps left by traditional indicators. The 1st objective of this research is to develop a quantitative assessment methodology for DALY based on the exposure dose and the end-point effect of the heavy metal. This methodology is crucial for the rest of the subtopics in this thesis and is expected to be provide the basis for future researches.

1.2.2 Cost benefit analysis for heavy metal remediation alternatives under normal scenario

Site-scale information on soil remediation feasibility is rare, especially in China. Economic analysis of soil remediation is important for decision making in practice. The health effect, as one of the important targets of risk management, has been overlooked in such analysis. The 2nd objective of this research is to conduct a site-scale cost-benefit analysis for soil remediation technologies. The benefit consideration is focused on the health aspect of remediation. Comparing health benefit with other types of benefit reveals the importance of health effect of remediation. The result will provide information on the economic feasibility of soil remediation alternatives at the target site, and also provide reference for local decision makers.

1.2.3 Soil remediation effectiveness under accidental scenario

Studies on accidental release of heavy metal focus on mechanism and consequences. Economic analysis of remediation process has rarely been conducted. Limited information is available on the economic feasibility of soil remediation under accidental scenarios. However such information is significant for accident management. The 3rd objective of this research is the assessment of economic feasibility of soil remediation process for a hypothetical tailing dam failure event. The impact of accident event and the economic feasibility of corresponding remediation process is estimated. By comparing the normal and accidental scenarios, the difference in health impact and economic feasibility can be revealed. The indication can provide reference for accident management and planning.

1.3 Research framework

In this thesis, in Chapter 2, we first developed a methodology of DALY quantification. This methodology is applied in Chapter 3 and Chapter 4 to evaluate the health impact caused by heavy metal exposure. Chapter 3 and 4 follow a similar basic analysis procedure. The health impact is estimated based on heavy metal exposure before and after remediation. The reduction of health impact is identified as the health benefit of the remediation, which is further utilized in the cost-benefit analysis for the remediation process. Fig 1-2 shows the analysis flow of the three chapters.

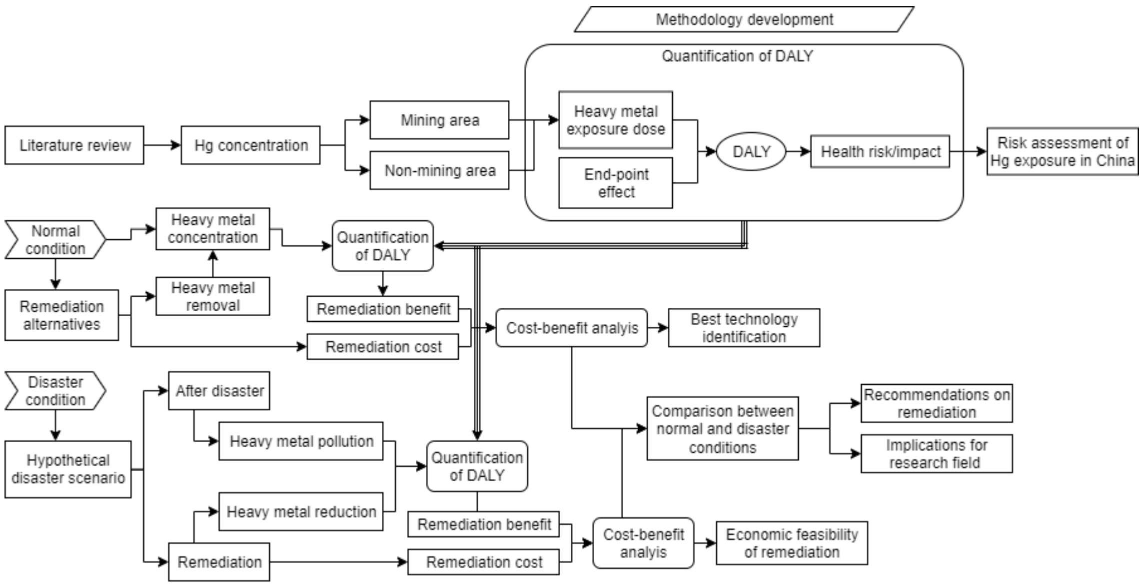

Fig 1-2. Research framework

(1) Development of the quantification methodology for DALY. This methodology is based on heavy metal intake from soil and the end-point effect of adverse effects caused by the heavy metals. This methodology is applied for assessment of average Mercury (Hg) exposure in China. The methodology is tested with this analysis and is applied in the other parts of this thesis.

(2) In this thesis, a cost-benefit analysis is conducted for soil remediation alternatives. The target area is agricultural lands in a selected Pb/Zn mining town. DALY is evaluated based on the soil heavy metal concentration before and after remediation and is transferred to monetized health benefit. Cost-benefit analysis is based on the evaluated cost and health benefit of each remediation alternative. Identification of the potentially best remediation alternative is the result of this analysis.

(3) Corresponding to the assessment under normal scenario, an analysis under accidental scenario is also conducted. A hypothetical tailing dam failure scenario is developed for the target area, and an

affected agricultural land is selected as the target of the analysis. DALY is estimated with the same methodology. The analysis results under normal and accidental scenarios are compared. Influence of the hypothetical tailing dam failure is noted. Differences between normal and accidental scenario are identified. The economic feasibility is assessed for the remediation process following the tailing dam failure event.

CHAPTER 2: QUATIFICATION METHODOLOGY OF DALY AND APPLICATION IN RISK ASSESSMENT OF MERCURY EXPOSURE IN CHINA

Quantification of DALY is important for cost-benefit analysis in this research. In this Chapter we develop a methodology to quantify DALY for heavy metal exposure. This methodology will be tested by an assessment of average health impact caused by Mercury (Hg) exposure in China. the assessment will show the capability of DALY as a health impact indicator. After that, this quantification methodology will be applied for cost-benefit analysis in Chapter 3 and Chapter 4.

2.1. Introduction

Hg is a highly toxic heavy metal which can cause reverse effect to various human organs and systems, chronic Hg exposure is especially effective on nervous system and kidney (Poulin & Gibb, 2008). In 1956, Minamata disease, caused by severe Hg poisoning, was discovered in Minamata city in Japan (Laham, 1963). This epidemic facilitated the Minamata convention, an international convention on Hg control (Kessler, 2013). With the experience of Minamata disease, Japan has been well improved in Hg regulation ever since. On the other hand, China is one of the biggest Hg producer and consumer in the world, counting 30%~40% of the total global Hg emission (Lin et al., 2017). Coal combustion and gold mining are the two main anthropogenic sources of Hg in China (Feng & Qiu, 2008). To discover the Hg pollution level in China, Japan is selected as a reference area for China. As Minamata convention came into practice in 2017, phasing out most Hg involving products and calling for a stricter Hg regulation internationally (Kessler, 2013), Hg monitoring and control on national and international scale became more important than before. A basic understanding of present Hg pollution state is necessary for solution effectiveness assessment in future pollution control (Liu et al., 2018).

Traditional researches about Hg exposure focused on regional pollution level or vicinity of anthropogenic Hg sources such as Hg mining areas and metal smelters. For example, some researchers in China focused on the Wanshan mining area, an abandoned Hg mine located in Guizhou province, Southwest China (Adlane et al., 2020). These studies revealed a high pollution level in those areas. Some other researchers studied relatively larger areas with high industrialization or urbanization level (Zhang et al., 2020; Qi et al., 2020). However, most of these researches were about restricted areas. Research on a national scale is rare. Thus, an investigation of national average pollution level in China is required.

Most of the previous researches focused on single Hg exposure source. Research of total human exposure (Ott, 1990), which start from human as a receptor of pollutants and include multiple exposure sources and routes, remained as a minority. The most frequently studied exposure sources are soil and foodstuff. Zhou et al. found a higher Hg level than the reference background level in agriculture soils in China (Zhou et al., 2018). Liu et. al identified fish as the main MeHg source in non-polluted areas in China, contributing 56% to the total MeHg intake (Liu et al., 2018). Zhang et al. studied a polluted area in inland China and found rice to be the main exposure source for MeHg, fish only contributed 1~2% to the total MeHg exposure (Zhang et al., 2010). The main exposure source of MeHg in China remained debatable. An investigation of MeHg source in polluted areas in China is also required.

In this research, a methodology of quantitative assessment of Disability-adjusted life years (DALY) is developed. This methodology is further applied to the assessment of the average health risk and impact caused by Hg exposure in China. DALY, as an indicator for health impact, has barely been used in quantitative risk assessment of heavy metal in China. Compared to traditional health risk indicators, DALY can assess the health risk by combining dose-response relationship with the impact of diseases or disabilities on the human body. Consequently, DALY is expected to present a more comprehensive health impact level (Verones et al., 2020).

The main objectives of this research are 1) investigation and comparison of the average human exposure level to Hg in China and Japan; 2) identification of the most important exposure routes; 3) assessment of DALY caused by Hg exposure level for population in both countries.

2.2. Methodology

2.2.1 Data acquisition and processing

Meta-Analysis is a statistical technique for combining the findings from independent studies and has been well used in national scale heavy metal analysis. For example, Zhou et al. (2018) reviewed 189 papers and studied 6 heavy metals in agriculture soils in China. Liu et al. obtained soil heavy metal data of 85 mines across China and investigated the pollution level (Liu et al., 2019). In this research, a meta-analysis is performed to investigate the average Hg concentration in China. the concept of meta-analysis first came up in 1976, as “the analysis of the results of statistical analyses for the purposes of drawing general conclusions” (Hedges, 1992). Meta-analysis refers to the statistical methodology of combining the results of replicated research studies for new and larger-scale findings. By conducting a literature review, meta-analysis can provide a quantitative estimate over all the included studies. We collect data from research papers published between 2010 and 2020 from the Web of Science. Key words combination of “mercury” and “risk” or “exposure” and “China” or “Japan” is applied. The papers included must meet the following criteria: (1) it must be a field study with sample collection and determination of Hg concentration; (2) the target area must be in China or Japan; (3) the target environment media must include at least one of the following: indoor air, ambient outdoor air, soil, and diet. The papers are checked by title, abstract and full text for screening, and finally 202 papers with data from 131108 samples are included. A database is then constructed with information of each paper, including first author and published year; research target area; types and amount of the samples; Hg concentration in samples (mean value and standard deviation). Some studies on Hg are about the vicinity of some certain Hg source. Those studies have a relatively smaller target area and higher pollution levels. If we simply integrate the data in all the literatures, the result will be affected by extreme values and could not show the average Hg concentration level properly. Therefore, those researches are separated from others. In this research, we divide the data of Hg concentration in China into two groups, "group A" and "group B". Group A includes papers where the major pollution source in the research area is not clearly demonstrated. Some studies demonstrate the pollution source as anthropogenic sources such as "industrial activity" or "agricultural activity". In these researches, Hg pollution is distributed in relatively larger areas, and the pollution level is expected to be relatively lower. Group B includes the papers where the pollution source is limited to a specific plant or mining area, such as Hg mining areas and fluorescent lamp factories. The study areas in group B are smaller and usually the surrounding areas of those plants or mining areas, so the Hg pollution is distributed in a limited area, the pollution level is also expected to be higher than group A. Dividing the two groups helps us to distinguish the areas with different pollution levels and avoids the extreme values to influence the understanding obtained by this analysis. Hg presence in environment media is categorized into 9 groups: outdoor total gaseous mercury (TGM), outdoor particulate boundary mercury (PBM), indoor TGM, indoor PBM, total Hg in soil, total Hg in aquatic products, total Hg in other foodstuffs (all other types of foods except aquatic products), MeHg in aquatic products and MeHg in other foodstuffs. The average Hg/MeHg concentrations of these environmental media for each group are evaluated by a weighted average by the sample quantity in each literature.

In the literatures included in this research, the number of literatures about China and Japan are significantly imbalanced. Only 3 papers are about Japan, and some data for Japan is not available. For outdoor TGM, the governmental atmospheric monitoring data of 2018 (n=2770) (Ministry of the Environment Japan, 2018) is applied in this research. For outdoor PBM, we assume that the ratio of TGM and PBM in Japan is the same with "group A" of China. Then the outdoor PBM is evaluated by this ratio and the outdoor TGM concentration. Indoor TGM is evaluated with the same method. MeHg dietary intake is not available in the literatures, in this research, we assume that all the MeHg intake of Japanese population is from fish consumption. MeHg in fish is reported to be 75%~100% (FAO/WHO, 2016) of the total Hg in fish, in this research, we apply a ratio of 90% to evaluate MeHg intake of Japanese people. The average concentrations of different environmental media and total sample quantities are shown in **Table 2-1** and **Fig 2-1**. In **Fig 2-1**, the numbers on the bars are the numbers of samples for each exposure media.

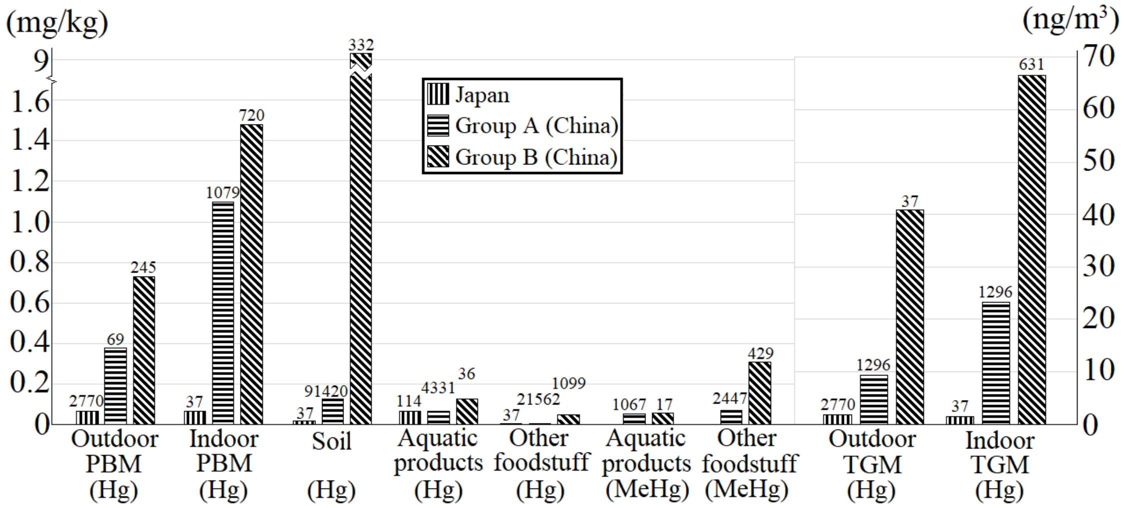

Fig 2-1. Hg and MeHg concentration in environmental media

Table 2-1. Average Hg and MeHg concentration in environmental medias.

		Japan	Group A (China)			Group B (China)		
Concentration		Average	Average	5th percentile	95th percentile	Average	5th percentile	95th percentile
Outdoor	PBM	0.07	0.38	0.12	0.92	0.73	0.02	3.2
	TGM	1.84	9.37	6	23	41.3	2.4	302
Indoor	PBM	0.07	1.66	0.15	10.59	0.36	0.32	0.58
	TGM	1.48	23.25	7	246	66.62	13	200
Soil		0.02	0.13	0.03	0.2	9.18	0.01	37.9
Diet (Aquatic products)	MeHg	25.66	58.72	16.3	115	66	17	73
	Hg	0.07	0.08	0.02	0.18	0.13	0.02	0.29
Diet (Other foodstuffs)	MeHg	-	7.53	0.2	30	31.11	0.05	65.45
	Hg	0	0.01	0.01	0.01	0.05	0	0.14

Data obtained from literature review (Unit is (mg/kg) for PBM, soil and Hg in diet; (μg/kg) for MeHg in diet; (ng/m^3) for TGM)

2.2.2 Total human exposure assessment

In this research, we set 3 main exposure routes for the exposure scenario: ingestion, inhalation, and dermal exposure. The average daily dose (ADD) is calculated by the following equations (EPA, 1989b):

$$\mathrm{ADD_{inhalation}} = \mathrm{C} \times \mathrm{InhR} \times \mathrm{EF} \times \mathrm{ED}/(\mathrm{BW} \times \mathrm{AT}) \tag{2-1}$$

$$\mathrm{ADD_{ingestion}} = \mathrm{C \times IngR \times EF \times ED/(BW \times AT)} \tag{2-2}$$

$$\mathrm{ADD_{dermal}} = \mathrm{C \times SA \times SL \times ABS \times EF \times ED/(BW \times AT)} \tag{2-3}$$

C is the Hg concentration in corresponding environmental media; InhR is the inhalation rate; EF is the exposure frequency; ED is the exposure duration; BW is body weight; AT is the time during which the exposure is averaged; IngR is the ingestion rate; SA is the human skin surface area; SL is skin adherence factor; ABS is the skin absorption factor.

Additionally, inhalation and dermal exposure dose caused by indoor and outdoor atmosphere are calculated separately and summed into a total daily dose according to the average time spent indoors and outdoors respectively.

$$\mathrm{ADD_{total}} = \mathrm{(ADD_{outdoor} \times T_{outdoor} + ADD_{indoor} \times T_{indoor})/24} \tag{2-4}$$

ADD_{total} is the total daily dose; ADD_{indoor} and $ADD_{outdoor}$ are the doses calculated respectively by indoor and outdoor atmosphere Hg concentrations. $T_{outdoor}$ is the average times spent (h) outdoors. $T_{indoors}$ was defined by (24h-$T_{outdoor}$).

The ingestion dose is the sum of soil ingestion exposure and total diet ingestion exposure.

$$\mathrm{ADD_{ingestion-soil}} = \mathrm{(C_{soil} \times IngR \times EF \times ED)/(BW \times AT)} \tag{2-5}$$

C soil is the Hg concentration in soil; IngR soil is the soil ingestion rate.

In some researches, only total dietary exposure is available, we can’t separate aquatic products from them, so those values are applied as “other foodstuffs”. Intake by aquatic products and other foodstuffs are evaluated respectively by the following equation:

$$\mathrm{ADD_{ingestion-diet}} = \mathrm{(C_{diet} \times FC \times EF \times ED)/(BW \times AT)} \tag{2-6}$$

C diet is the average Hg concentration in food; FC is the food consumption rate. Then the total ingestion exposure will be calculated as the sum of exposure by soil ingestion and total diet ingestion.

$$\mathrm{ADD_{ingestion}} = \mathrm{ADD_{ingestion-soil}} + \mathrm{ADD_{ingestion-diet}} \tag{2-7}$$

Parameters applied in exposure evaluation in this research is listed in **Table 2-2** (Duan, 2015) (National Institute of Advanced Industrial Science and Technology, 1997).

2.2.3 Disability adjusted life years

Considering the requirement for health impact assessment of this thesis, the indicator must meet two major requirements. The first requirement is to provide a comprehensive consideration of health impact, including both probability and severity of disease caused by heavy metal exposure. The second requirement is the health impact should be able to be monetized for cost-benefit analysis. Traditional indicators such as Hazard Quotient and Margin of Exposure failed to assess health impact based on disease severity. Simple possibility is also difficult to be transferred to financial loss. DALY estimates the health impact based on probability and severity of disease, which provides more comprehensive estimation of health impact. DALY is monetized by an average financial burden, which can be applied in cost-benefit analysis, therefore, DALY is a proper indicator for this research.

The methodology for quantifying DALY was developed by Murray (1994). The original quantification methodology is demonstrated in **supplemental material S1**. In this research, DALY is defined by years life lost (YLL) and years lost to disability (YLD).

$$\mathrm{DALY} = \mathrm{YLL} + \mathrm{YLD} \tag{2-8}$$

The exposure is defined as a life-long exposure and is not expected to cause a fatal effect, therefore in this research, YLL is assumed as 0. YLD is defined by:

$$\mathrm{YLD} = \text{disease probability (DP)} \times \text{disease weight(DW)} \tag{2-9}$$

2.2.3.1 Disease probability

Fig 2-2 shows the quantification process of disease probability, the numbers refer to corresponding equations in the text. The disease probability is defined by accumulative expectation of exposure dose and dose-response relationship.

$$\mathrm{DP} = \int \mathrm{f(x)} \times \phi(\mathrm{x})\,\mathrm{dx} \tag{2-10}$$

x is the exposure dose; f(x) and ϕ(x) are the probability density functions of exposure dose and dose-response relationships, which are assumed to follow the log-normal distribution in this research.

$$F(x) = \frac{1}{x\sigma\sqrt{2\pi}} \exp\left(\frac{-(\ln x - \mu)^2}{2\sigma^2}\right) \tag{2-11}$$

μ is the mean value of ln(x); σ is the variance of ln(x).

For Hg exposure, μ and σ can be calculated by the mean value (μ_x) and standard deviation (σ_x) of Hg exposure dose obtained from the literatures. These parameters can be transferred with the following equations.

$$\mu = \ln\left(\frac{\mu_x^2}{\sqrt{\mu_x^2 + \sigma_x^2}}\right) \tag{2-12}$$

$$\sigma = \ln\left(1 + \sigma_x^2/\mu_x^2\right) \tag{2-13}$$

For dose-response relationship, μ and σ can be calculated by the geometric mean value (GM) and geometric standard deviation (GSD). These parameters can be transferred with the following equations.

$$\mathrm{GM} = e^{\mu} \tag{2-14}$$

$$\mathrm{GSD} = e^{\sigma} \tag{2-15}$$

In the present research, GSD for dose-response function is defined by the difference of metabolizing rate and sensitivity in the human body, 1.4 (Masuyama, 1977) and 2.7 (Nordberg & Strangert, 1976) [μg/(kg-BW×d)], respectively. The integrated GSD is calculated by:

$$\left(\ln(\mathrm{GSD})\right)^2 = \left(\ln(\mathrm{GSD}_1)\right)^2 + \left(\ln(\mathrm{GSD}_2)\right)^2 \tag{2-16}$$

GM is defined by assuming that the NOAEL (no observable adverse effect level) could induce a 0.5% possibility of disease (Nakanishi et al., 2003).

$$0.005 \fallingdotseq \phi(GM/GSD^{2.58}) = \phi(NOAEL) \tag{2-17}$$

Therefore:

$$GM = NOAEL \times GSD^{2.58} \tag{2-18}$$

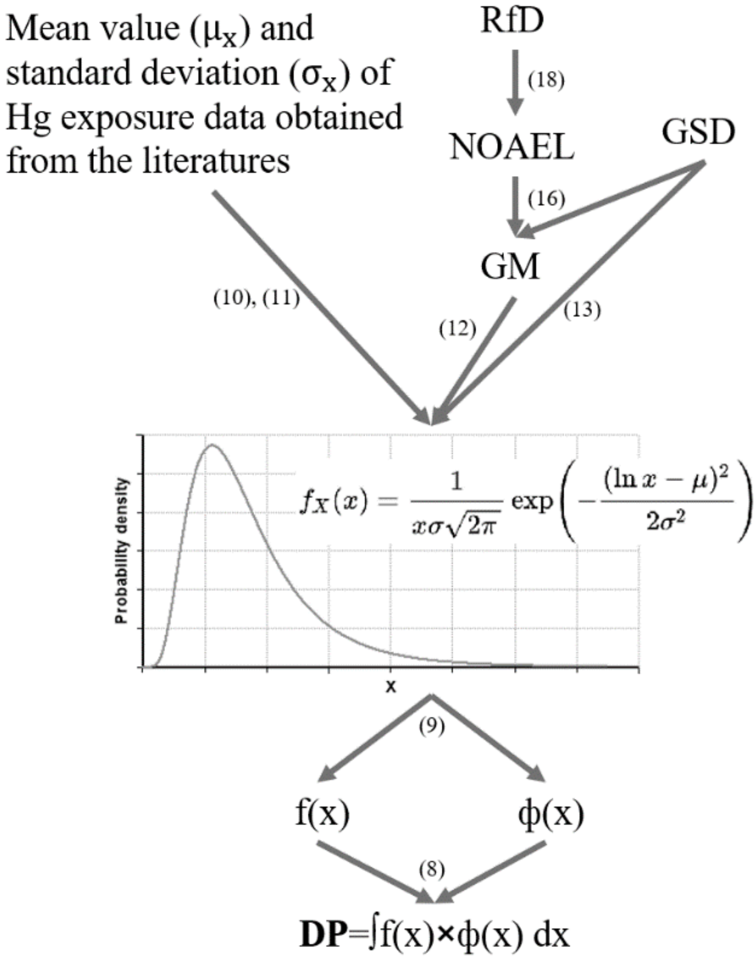

Fig 2-2. Quantification of disease probability (DP)

The NOAEL value is not available, they are estimated from RfD (reference doses). Since the RfD could be derived from NOAEL (Barnes et al., 1988):

$$RfD = NOAEL/(UF \times MF) \tag{2-19}$$

Therefore

$$\mathrm{NOAEL} = \mathrm{RfD} \times \mathrm{UF} \times \mathrm{MF} \tag{2-20}$$

UF, uncertainty factor; MF, modification factor. In this research, we assume that UF=10 and MF=1. Reference oral exposure dose applied in this research are shown in **Table 2.3** (The Risk Assessment Information System, 1998; US EPA, 2001). The information for reference dose and the corresponding health effect for inorganic or elemental mercury was not available, so reference dose for mercuric chloride was applied in this research.

2.2.3.2 Disease weight

Table 2-2. Parameters applied in exposure assessment

Parameters		China[a]	Japan[b]	unit	Parameters	China	Japan	unit
Body weight (BW)		61.90	58.35	kg	Soil ingestion ($IngR_{soil}$)	50.00	47.70	mg/d
Life expectancy		74.83	81.16	year	Exposure frequency (EF)	365.00	365.00	d/year
Skin surface area (SA)		16000.00	16000.00	cm^2	Exposure duration (ED)	74.83	81.16	year
Food consumption (FC)	Aquatic products	29.60	92.00	g/d	Dermal absorption factor (ABS)	0.001	0.001	-
	Other foodstuffs	896.50	1067.90	g/d				
Time outdoors ($T_{outdoors}$)		4.22	1.20	hour/d	Skin adherence factor (SL)	0.70	0.70	mg/($cm^2\times$ d)

Reference: a: X. Duan et al., 2015; b: Research Center for Chemical Risk Management, 2007)

Table 2-3. Reference dose (oral)

Toxicant	Reference dose mg/(kg-BW*d)	Health effect
Mercuric chloride	3.00×10^{-4} (a)	Autoimmune glomerulonephritis
MeHg	1.00×10^{-4} (b)	Mild mental retardation

Reference: a: The Risk Assessment Information System, 1998; b: US EPA, 2001

In this research, the end-point effect is selected as autoimmune glomerulonephritis for chronic oral inorganic Hg exposure and mild mental retardation for chronic oral MeHg exposure. For inorganic Hg, a disability burden of 0.105 per person per year for Chronic kidney disease IV (WHO, 2013) is applied. For MeHg, the disease burden is set as 0.361 per person per year for mild mental retardation (Lopez et al., 2006). The disease duration is set to 5 years.

2.2.4 Sensitivity analysis

The sensitivity analysis is performed with a single parameter (SA) methodology (Pu et al., 2016), this methodology works by changing the value of all the parameters within its variation range one at a time while holding other parameters constant. Concentration of Hg, inhalation rate, ingestion rate, body surface area, and body weight are selected as the target parameters for the sensitivity analysis. The result of the sensitivity analysis reveals the influence on the evaluated exposure dose caused by variation of parameters.

2.3. Result and discussion

2.3.1 Information of literatures

The number of literatures by publication years and research targets of literatures is shown in **Fig 2-3**. The number of publications received a great increase after 2017, the year when Minamata convention came into practice, especially for group A. This indicate that with the Minamata convention came into force, research on the present Hg pollution level have drawn more attention, especially about Hg pollution situation on larger scales.

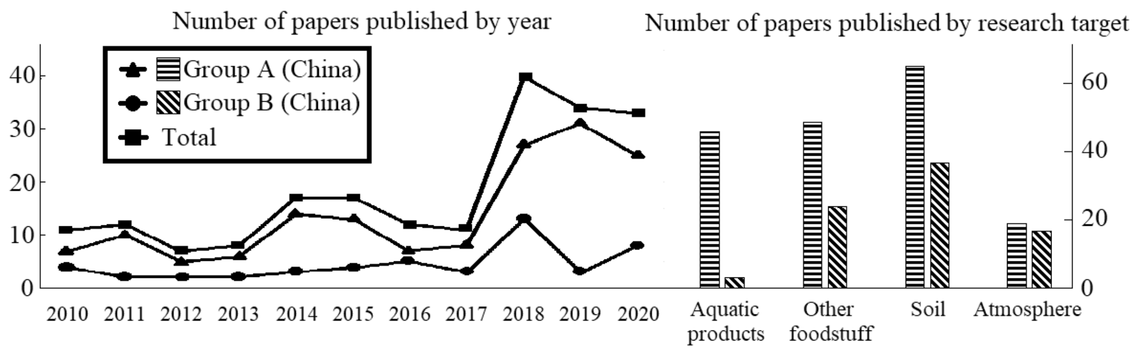

Fig 2-3. Number of literatures on Hg concentration in exposure media

For group A, dietary Hg receives the most attention, followed by soil and air. Aquatic products and other foodstuffs receive almost the same attention. For group B, soil is the most researched media, diet takes the second place. Only a few research studies the aquatic products consumed by residents. The reason for this might be that in the vicinity of Hg sources, the surrounding soil and crops grown from that soil receive the influence of those Hg sources more directly. On the other hand, aquatic products might not be necessarily produced in those areas.

Atmosphere is a less popular target for researchers in both groups, so more research about atmospheric Hg might be able to create a better understanding of the present pollution state of atmosphere in China.

2.3.2 Hg concentration and exposure assessment

The average Hg exposure dose of different exposure routes are shown in **Fig 2-4**. The average total Hg exposure dose is 7.81×10^{-4} mg/(kg-BW×d) for group B, 2.18×10^{-4} mg/(kg-BW×d) for group A in China, and 1.48×10^{-4} mg/(kg-BW×d) in Japan. The average total Hg daily exposure dose is 8.64 μg/d in Japan, close to 8.4 μg/d evaluated by Japanese government (Food safety commmision of Japan, 2004). Among the 3 main exposure routes, ingestion exposure is the most important route of Hg exposure, contributing 99.48%, 95.47% and 96.28% to the total exposure dose for Japan, group A and group B of China, respectively.

It is noticeable that the general ingestion exposure dose of group A and group B in China are 0.41 and 4.12 times higher than Japan, respectively. However, for inhalation exposure, the ratio increases to 11.9 and 37.0. This may indicate that the pollution sources in China may have a more severe impact on the atmosphere than foodstuff in Japan. Therefore, the necessity of control in atmospheric Hg in China, especially in polluted areas, is not neglectable.

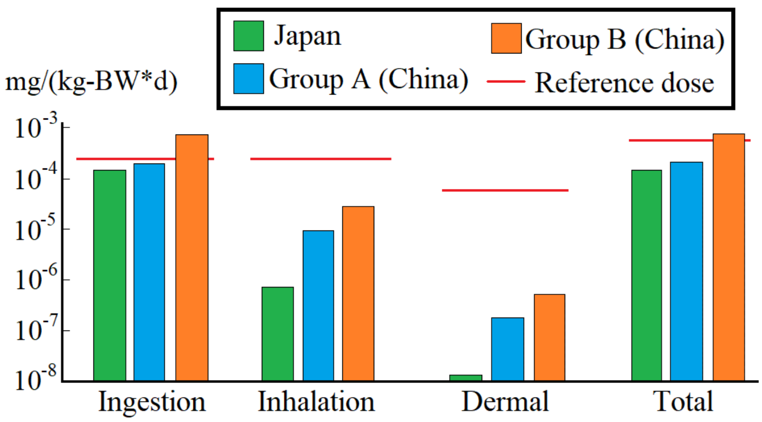

Fig 2-4. Exposure dose of Hg in China and Japan

The average MeHg exposure doses of different exposure routes are shown in **Fig 2-5**. In China, MeHg exposure dose via dietary intake is 0.137 μg/(kg-BW×d) for group A and 0.479 μg/(kg-BW×d) for group B. Much higher than the 0.057 μg/(kg-BW×d) evaluated by Liu (Liu et al., 2018) for non-polluted areas, indicating a relatively high pollution level. MeHg intake caused by aquatic products is 0.028 and 0.029 μg/(kg-BW×d) for group A and B in China. The daily MeHg intake by aquatic product consumption is 0.093 μg/(kg-BW×d) in Japan, close to the result evaluated by Takahiro (Watanabe et al., 2021). The intake of MeHg from aquatic products in Japan is higher than China. This may be caused by difference of diet structure between the two countries. The proportion of aquatic products in total diet is 8.62% in Japan and 3.3% in China.

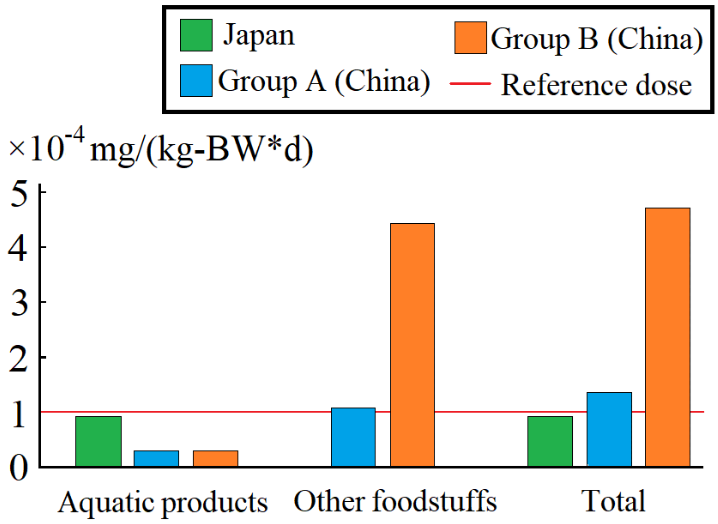

Fig 2-5. Exposure dose of MeHg in China and Japan

Aquatic product, which is regarded as a main Hg exposure source, contributed 20.51% and 16.93% in group A and B of China, respectively. Comparing to the fact that aquatic products only take up 3.3% of the total diet in China, it should be regarded as one of the important Hg exposure sources to Chinese population. However, the evidence is insufficient to support that aquatic product is the most important source, further investigation may be required.

For both Hg and MeHg, exposure is dominated by dietary intake. Controlling Hg concentration in food should be the most effective way of reducing the exposure of Chinese population to Hg, and aquatic products is one of the main sources to be considered. In Japan, fish is regarded as the main source of Hg exposure. To control the Hg concentration in fish species, water quality standards are made for public water areas. Specially, local governments can make additional regulation for certain requirements. Manufactures handling Hg in their production process are also responsible for notifying the amount of release to the government, as a part of the Pollutant Release and Transfer Register (PRTR) system (Ministry of the Environment Japan, 2015).

Among all the concentrations and parameters, the variation range of outdoor TMG is the largest (variation range: 5.81%~731.24%). However, sensitivity analysis shows that the variation of Hg concentration in “total Hg in other food stuffs” (variation range: 8.51%~300.00%) has the strongest influence on the result of exposure analysis (variation range: 19.76%~275.41%). The reason may be the ingestion intake takes a much higher part of the total Hg intake.

2.3.3 Disability adjusted life years (DALY)

The assessment result of DALY is shown in **Fig 2-6**. Total average DALY in Japan is 6.91×10^{-7} years per lifelong exposure, lower than 6.63×10^{-6} years for group A and 9.96×10^{-4} years for group B in China. This result indicates a higher health impact level in China. The tolerable DALY level defined by WHO was 1.0×10^{-6} DALY per person per year (WHO, 2010). Considering the average life expectancy, the tolerable DALY per lifelong exposure should be 8.12×10^{-5} years and 7.48×10^{-5} years for Japan and China, respectively. Taking this tolerable DALY as a reference, for group B in China, the total DALY caused by Hg exposure exceeds the tolerable threshold, indicating a relatively severe health impact to the population.

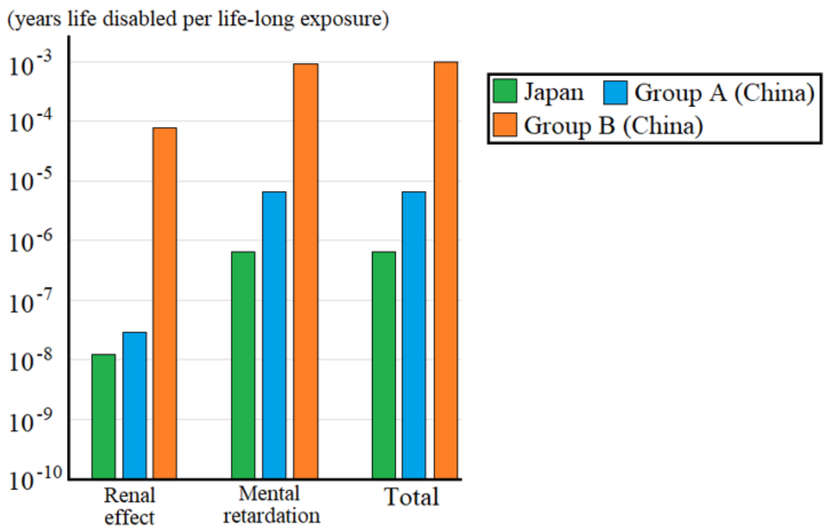

Fig 2-6. Average DALY of Chinese and Japanese Population

The most important endpoint effect is mental retardation caused by dietary MeHg exposure, responsible for 98.16% of the total DALY in Japan, 99.53% for group A and 91.48% for group B in China. This result indicates that the reduction of MeHg in foodstuff might be a priority.

Human health impact is an important part in the process of LCIA. DALY have been used in multiple LCIA researches such as for lead emission (Altuntas, 2020) or wastewater (Chen et al., 2019) and served as an indicator of health impact. The result of this research provides the information for health impact as a main factor in future LCIA.

2.4 Conclusion

For sustainable metal use for wide industrial sector, benefit and risk needs to be managed from the view of life cycle thinking. Human exposure, as an important part of the metal life cycle, is evaluated in this research. We conduct a meta-analysis about Hg concentration and exposure in China and Japan. The average exposure dose and DALY caused by Hg and MeHg exposure are evaluated. The average total Hg exposure dose is 2.18×10^{-4} mg/(kg-BW×d) for group A, 7.81×10^{-4} mg/(kg-BW×d) for group B in China, and 1.48×10^{-4} mg/(kg-BW×d) in Japan. MeHg intake amount is 0.137 μg/(kg-BW×d) for group A and 0.479 μg/(kg-BW×d) for group B by total dietary intake in China, and 0.093 μg/(kg-BW×d) by consumption of fish in Japan. The result shows that population in China are exposed to a higher amount of Hg and MeHg exposure, especially in areas with specific Hg sources such as Hg mines and metal smelters. The most important exposure route is ingestion exposure in both countries. However, the pollution sources may also have a strong influence on the atmosphere. Thus, inhalation exposure in highly polluted areas should also be paid attention to. The average DALY is 6.91×10^{-7} years per lifelong exposure for Japanese population, 6.63×10^{-6} years for group A and 9.96×10^{-4} years for group B in China. Average DALY for group B in China exceeds the tolerable DALY level set by WHO. The result of this research indicates that the necessity in controlling Hg pollution in highly polluted areas in China. This research provides the knowledge of present Hg pollution state of China on a national scale, which is seldomly discussed before. This knowledge will be significant for solution effectiveness analysis in future Hg control in China. The identification of crucial exposure routes provides policymakers with reference for Hg regulation and pollution remediation. DALY assessed in this research, as an indicator for health impact, can be utilized in future LCIA analysis.

CHAPTER 3: SOIL REMEDIATION WITH DIFFERENT REMEDIATION ALTERNATIVE TECHNOLOGIES UNDER NORMAL SCENARIO

In Chapter 2, a methodology of DALY quantification is developed. In this Chapter, the methodology is applied in cost-benefit analysis for agricultural soil remediation in a typical mining area. This methodology reveals reduction of DALY level before and after the remediation, which is treated as the benefit of remediation. Four remediation alternative technologies are assessed with the same methodology and compared for feasibility. This Chapter focuses on the normal scenario, the accidental scenario in the next Chapter.

3.1. Introduction

Due to the rapid industrialization and urbanization in China, soil heavy metal pollution has become a prioritized issue (Wu et al., 2022). Previous studies have discussed major anthropogenic sources of heavy metal released to the environment. Metal mining is identified as one of the major sources of soil heavy metal pollution (Yang et al., 2018). Metal mining is a highly pollutive industrial process due to the release of various hazardous heavy metals (HMs) (Lin et al., 2006). As one of the biggest mining countries in the world, China is facing severe environmental problems caused by mining activities. However, the environmental impacts have been heavily ignored during the mining process (Li et al., 2014).

Remediation actions are necessary for contaminated sites due to the high HM pollution level. Various technologies have been developed for soil remediation. These alternative technologies utilize different mechanisms to reduce the concentration or bioavailability of heavy metal in soil, thus reducing the influence of soil heavy metal on the environment (Liu et al., 2018). According to the mechanism utilized, the remediation alternatives can be categorized into different types. Different remediation alternatives significantly vary in heavy metal removal efficiency and economic effectiveness (Khalid et al., 2017). In practice, both environmental and economic effect are important for selecting the optimal remediation alternative (Chen & Li, 2018).

Considering the economic efficiency is a significant aspect of soil remediation, few previous researches provided discussion about this issue at site-scale (Khalid et al., 2017). Most field scale studies serve as experiment of remediation technologies and focus on single type of remediation alternative (Wan et al., 2016; Wei et al., 2021; Zhang et al., 2021). Few studies provided the insight of advantages and disadvantages of different remediation alternatives at the same site. Additionally, when previous researchers discuss the economic feasibility of soil remediation, health benefit is often overlooked (Lavee et al., 2012; Ospanbayeva & Wang, 2020; Wan et al., 2016; Zhang et al., 2021). Health impact is one of the most important protection targets of risk management (Swartjes, 2011), therefore including health assessment in cost-benefit analysis for soil remediation is necessary.

This research attempts to fill the gap by conducting a field-scale cost-benefit analysis for four remediation alternatives based on remediation cost and health benefit. The objectives of this research are 1) based on HM exposure dose and estimation of health benefit and benefit rate generated by remediation of HM-contaminated agriculture soil in the vicinity of a typical mining site and 2) a field-scale CBA of agriculture soil remediation alternatives and identification of the potentially best remediation technology. This research could provide information for field-scale soil remediation decision-making and address the significance of health benefits assessment in the CBA of soil remediation.

3.2. Material and methods

3.2.1 Research area

The target area of this research is the agricultural land in Jinding Town (**Fig 3-1**). The Jinding Pb/Zn mine is located at the center of Jinding Town. Bijiang River flows across the Jinding Town and the mining area (Cheng et al., 2018). Jinding deposit is the largest Pb/Zn deposit in China and the youngest sediment-hosted super giant Pb/Zn deposit globally (Xue et al., 2007). Jinding area has a long history of mining activity. Due to the HM release caused by mining processes, HM pollution is found in agricultural soil in this area (Jin, 2017). Up to now, the environmental action is still focused on reducing the pollution due to the mining process, and few remediation technologies have been applied to reduce the existing pollution caused by previous mining activities (Li, 2020).

3.2.2 Scenario definition

3.2.2.1 General assumptions

In this research, the target area was set as all the agricultural lands of Jinding Town because confirming the target population for a selected agricultural land was difficult. Correspondingly, the population affected by the target agricultural land was the entire population of Jinding Town. Given that the difference in HM concentration among different areas could not be determined, HM pollution was assumed to be averagely distributed in the target land, and the HM concentration of agriculture soil in the Bijiang watershed (which is a part of the target area) reported by Jin (2017) was applied as the average soil HM concentration of the target area. The remediation alternatives were assumed to be evenly applied to the target agriculture soil with the same unit cost and for the same soil depth to maintain consistency with the assumption of soil concentration. As reported by Wan et al. (2016), phytoremediation was found to reach a soil depth of 0–20 cm. The remediation of surface soil was successful in reducing the HM concentration in the crops grown from the remediated site, indicating effective remediation. Consistency was maintained between the definitions of scenarios to compare the removal ability of different technologies. Therefore, the remediated soil depth in this research was set as 0.2 m. The exposure dose of HMs was evaluated on the basis of the intake of crops harvested from the target area. For the consistency of crop concentration and exposure dose, the entire population of Jinding town was assumed to be affected by the crops harvested in the target area, and the average consumption of crops by Chinese people (Duan et al., 2015) was applied.

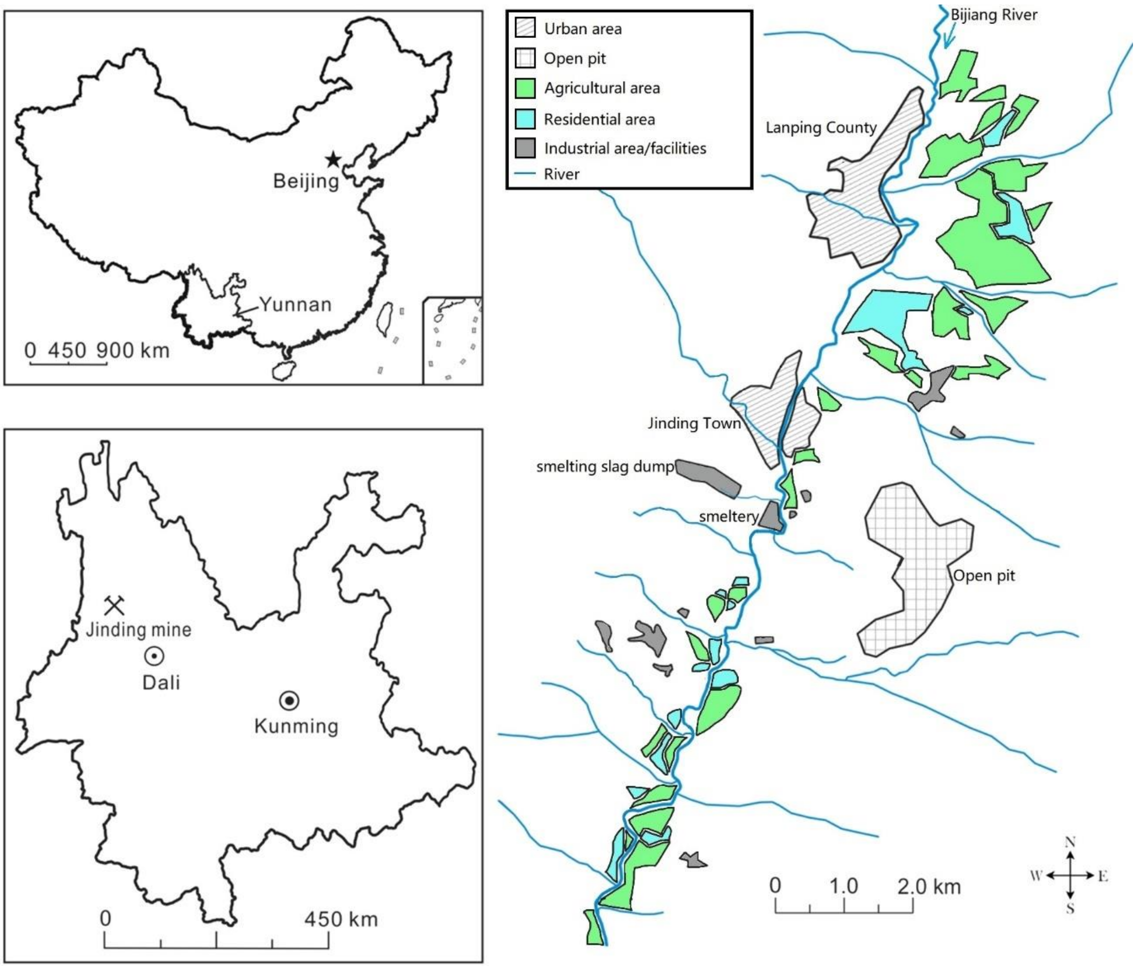

Fig 3-1. The map of study area (modified from Cheng et al., (2018))

3.2.2.2 Heavy Metal exposure

This research focused on the HMs with high concentration and high toxicity in the target area. HMs, such as As, Pb, Cd, Hg, Cu, Ni, and Co, has relatively high toxicity (Xu et al., 2008). Considering the soil HM concentration in the Jinding mining area (Cheng et al., 2018; Jin, 2017), As, Pb, and Cd were selected for HM exposure analysis. Due to the lack of environmental actions in the history of the target mining area, the HMs released by the mining process entered the environment. During long-term transportation, the HMs entered and accumulated in the agriculture soil in the target area, causing the soil to present a high HM concentration. The exposure scenario is presented in **Fig 3-2 a**. In this research, the major exposure routes are soil ingestion, soil particle inhalation, crop consumption, and groundwater consumption. The analysis procedure of this research is presented in **Fig 3-2 b**.

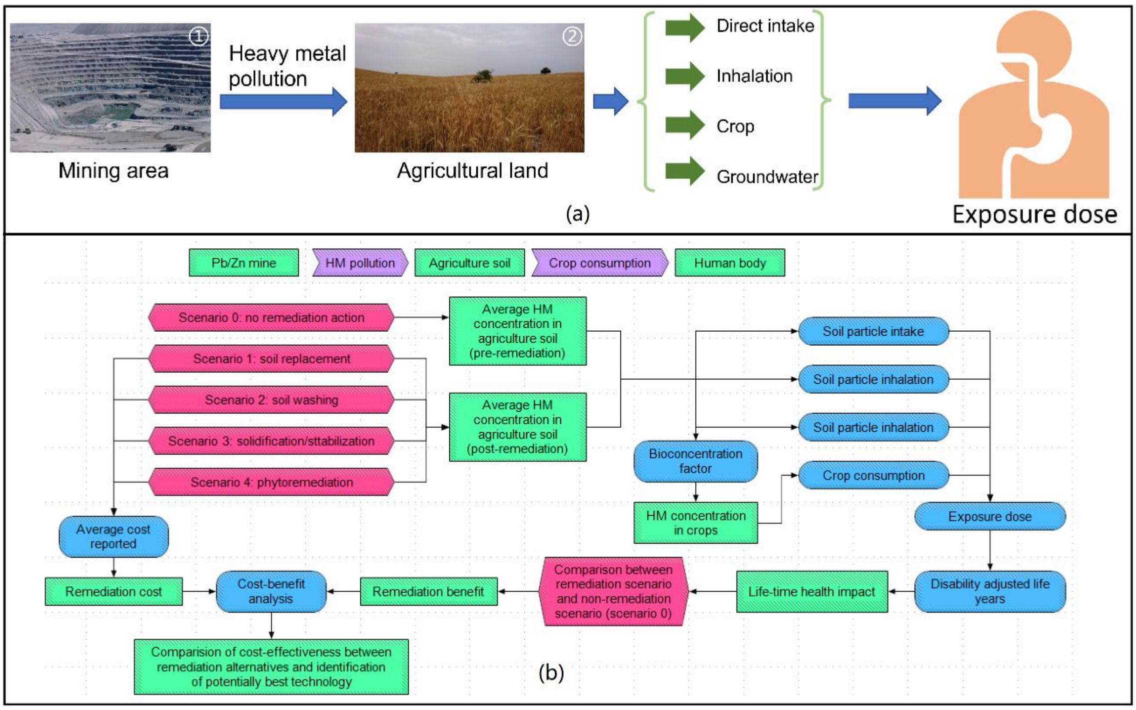

Fig 3-2. Exposure route (a) and analysis structure (b) of this research (Image sources: ① (Jahn, 1984); ② (Photogir, 2020).)

3.2.2.3 Remediation scenarios

Compared with other types of pollutants, HMs could not be destroyed biologically but could be only transformed from one oxidation state or organic complex to another, becoming either more soluble, less toxic, less bioavailable, or volatilized to be treated (Chen & Li, 2018; Garbisu &

Alkorta, 2001). Thus, the technologies often used for HM treatments are **soil replacement (excavation)** (Lambert & Leven, 2000), **stabilization/solidification**, **soil washing**, and **phytoremediation** (Wuana & Okieimen, 2011). In the present research, five scenarios were defined. In scenario 0 (baseline scenario), no remediation technology was applied. This scenario served as the baseline scenario. In scenarios 1–4, one of the four selected technologies was applied in each scenario. This research focused on the comparison between the available technologies. Therefore, the combined remediation process was not considered in the definition of scenario. As no remediation action has been taken in this area, the soil HM concentration after remediation was assumed in accordance with the mechanism of remediation technology and average RE reported by previous works. In scenarios 1 and 4, the soil HM concentration after remediation was assumed to be equal to Chinese soil background concentration (Liang et al., 2021). In scenarios 2 and 3, the RE was reported by previous studies. Information of each scenario is listed in **Table 3-2**.

Scenario 0. Baseline scenario

In this scenario, no remediation technology is applied. This scenario reveals the present health impact on the resident and serves as a control scenario. The reduction of health impact under each other scenario will be calculated based on the original health impact level. for all the scenarios, we assume that HM concentration in soil is assumed to be constant after remediation and discount rate is not considered.

Scenario 1: Soil replacement (excavation)

Soil replacement (excavation) is a typical ex-situ remediation strategy that removes the contaminated soil from its original site and transported for disposal. The disposed heavy metals will be transported to qualified disposal sites and will be monitored or further treated to prevent polluting the environment. In this scenario, after contaminated soil is excavated from the target site, fresh soil is refilled to the original site. The fresh soil is assumed to be taken from uncontaminated site and the soil metal concentration after remediation is supposed to be equal to the background value. In this research, we select the China soil background value. The average cost of soil excavation reported by FRTR is \$270 to \$460 per ton of soil (FRTR, 2012). A bottom boundary value of \$270 per ton is assumed in this research. Soil HM concentration after remediation is assumed as background value (considering fresh soil). The density of soil is $1.3 t/m^3$ (Tuyishimire et al., 2022), therefore, the unit cost of scenario 1 is $\$208/m^3$.

Scenario 2: Soil washing

Soil washing refers to ex situ techniques that employ physical and/or chemical procedures to extract metals contaminants from soils. The heavy metals will be extracted by the washing agents and solutions from the target soil. The solution and agents that are used in the washing process will be further treated for disposal or release.

The efficiency of soil washing heavily depends on metal and soil property, so the selection of remediation technology focus on these tow aspects. Qiu et al.（2010) reported a soil washing method using combined Na_2EDTA and oxalate as washing agent to remove As, Pb and Cd in the polluted soil. The property of the experimented soil used by Qiu et al. （2010) is similar to the present research (**Table 3-1**). The maximum removal rate is As 65.6%; Pb 42.9%; Cd 33.8% (Qiu et al., 2010). In this scenario, we assume chemical solution is applied to extract the heavy metal from contaminated soil. Unfortunately, no detailed information on the cost for this methodology is available. As is reported by FRTR, average cost of soil washing is \$70 (large sites) ~ \$183 (small sites)/m^3 (FRTR, 2012). Therefore, an average value of \$70/m^3 is applied in this research.

Table 3-1. Soil property of (Qiu et al., 2010) and the present research

Property	Jinding area (Chinese purple soil, He et al., 2003)	(Qiu et al., 2010)
Sand (%)	52	55.8
Silt (%)	30	28.9
Clay (%)	18	15.3
Organic compound (%)	1.7	5.1
pH	7~8	7.31
As concentration (mg/kg)	19.62	70.41
Pb concentration (mg/kg)	138.97	254.35
Cd concentration (mg/kg)	5.03	1.73

Scenario 3: Stabilization/solidification

Instead of reducing the heavy metal content in soil, Solidification/stabilization (S/S) reduces the movability and bioavailability. Physical amendment or chemical agent is added to the soil to transform heavy metal into solidified or other more stable state, reducing the extraction by plants and other organisms in the environment. Solidification/stabilization can be conducted in in-situ form and ex-situ form. In this scenario, a physical/chemical agent is applied to the soil to lock the heavy metals in a more stable state and reduce bioavailability. The stabilized heavy metals will remain in the soil but with a low bioavailability, preventing it from being absorbed by the crops and exposed to human beings. Therefore, the stabilized heavy metals are isolated from the ecosystem of the target soil.

For the stabilization/solidification of heavy metals in the agricultural soil, one major issue is that the amendment added must influence the function of the agriculture soil in the future. Among all the agents available, biochar created from agricultural waste is regarded as a proper agent because it will not hinder the further usage of agriculture soil (Wang et al., 2021). The removal rate of bioavailable Pb and Cd by biochar can be up to 92% and 71% (Houben et al., 2013). Arsenic removal by biochar is less efficient. A research showed that a Fe-oxide assisted biochar stabilization can provide 12% removal of available As in the soil (Wu et al., 2020). Unfortunately, presently in China, stabilization/solidification remained at lab experiment state, actual application cases have not yet been carried out. Therefore, little information about application cost is available. As an average cost of solidification/stabilization reported by (FRTR, 2012), the cost for small sites (under 1 ha) is $216 for easy remediation (only metals) and $248 for difficult remediation (metals and volatile organic compounds). The cost for large sites (over 1 ha) is $124 for easy remediation and $190 for difficult remediation. Considering the situation of the target site (heavy metal remediation for a 9.3 km^2 land), an average cost of $124/m^3 for solidification/stabilization is applied in this research.

Scenario 4: Phytoremediation

Phytoremediation is to grow plants in contaminated soils, relying on green plants to remove heavy metals or stabilize them into harmless status (Mahmood-ul-Hassan et al., 2020). Phytoremediation is often regarded as a cost-effective technology, however, due to its mechanism, it can cost much more time than other technologies.

In this scenario, hyperextraction plants are cultivated in the target area. The heavy metal is then extracted in the plants and removed from the target field. We assume that phytoremediation should remove the heavy metals caused by artificial sources from the soil and return the soil to a natural state. This assumption will cause the soil heavy metal concentration to be reduced to background value. Therefore, the removal rate of phytoremediation is the same with soil replacement.

The most important issue in phytoremediation application is the combination of and extraction plant species and the target metals. However, limited field-scale reports are available on phytoremediation. Wan et al. (2016) reported a field study using Sedum alfredii H. and Pteris vittata to extract As, Cd and Pb in agriculture soil, with successful decrease detected in soil concentration of all 3 elements. The research area of Wan et al. (2016)'s research is also a Pb/Zn mining area in South west China, with a severe pollution history. So we expect that the climate, soil property and heavy metal concentration between the target area of the two researches should be similar. Therefore, in this research, we apply this phytoremediation strategy. The 2-year removal rate of As, Cd and Pb in Wan et al. (2016)'s research is 55.3%, 85.8% and 30.4%, respectively. In this research, the 8-year removal rate of As, Cd and Pb is 42.92%, 98.07% and 81.27%, respectively. Compared to Wan et al.

(2016)'s research, both the removal rate in this research are higher, but the remediation duration is also relatively longer.

The cost of phytoremediation reported by Wan et al. (2016) is \$75,375.2 per hm^2, with an initial cost of \$34,684.50 per hm^2 and an optional cost of \$40,690.70 per hm^2 for 2 years manipulation (\$20345.35 per hm^2 annually). This result is applied as initial and annual cost in the present research, Unlike other technologies, phytoremediation requires a long-term application to reduce the heavy metal concentration in soil. The period of remediation is much longer and will last for several years. Thus, according to the extraction ability of the 2 plants, the remediation period must be evaluated. The 2 hyperextraction plants are assumed to be able to intercrop with wheat. Therefore, the main exposure route is still dietary intake from wheat consumption.

The remediation target is assumed to reduce the soil heavy metal concentration to China soil background value and then stay constant. Wu et al. (2007) studied samples of Sedum alfredii H. and Pteris vittata from 6 sites, reporting an average heavy metal in the shoots above the ground. The accumulated heavy metal concentration is 1373 mg/kg As, 680 mg/kg Pb in the shoot of Pteris vittata and 2290 mg/kg Pb, 708 mg/kg Cd in the shoot of Sedum alfredii H. There is no evidence of heavy metal being released or volatized into the atmosphere by these two plant species, so in this research we consider the heavy metals are concentrated in the shoots of the plants and are removed from the target soil. The yearly production of biomass of Pteris vittata can reach 36 t/ha (Chen et al., 2002). As is reported by (Yang et al., 2002), Sedum alfredii H. can be harvested 3~4 times a year with a seasonal biomass production of 1.8 t/ha. The remediation duration is estimated according to the extraction rate and biomass of Sedum alfredii H. and Pteris vittata (**Supplemental material S2**).

3.2.2.4 Mass balance

Unlike other types of pollutant, heavy metals cannot be completely removed from the environment, in order to reduce the environmental and health impact in the target area, they can either be transferred to other areas with lower impacts or transfer to states that has a lower bioavailability (Chen & Li, 2018; Garbisu & Alkorta, 2001). Considering the different mechanisms of the remediation alternatives, heavy metals have different end-point fate in different scenarios. The mass balance concept of four scenarios is shown in **Fig 3-3**. For soil replacement and phytoremediation, the polluted soil and heavy metal concentrated plant tissues are moved from the target site to some certain disposal sites where the pollutants will be under strict management and monitoring to minimize the impact on environment. For soil washing, the water/solution used to treat the polluted soil is degraded into wastewater. Such water must undergo further treatment process to minimize the pollution level and finally be released to the environment. For stabilization/solidification, the heavy metals are transferred into states with low bioavailability and remain in the soil. Since little information is available on the environmental effect of these heavy metals during the end-point process, we exclude those processes from the health impact analysis boundary of this research.

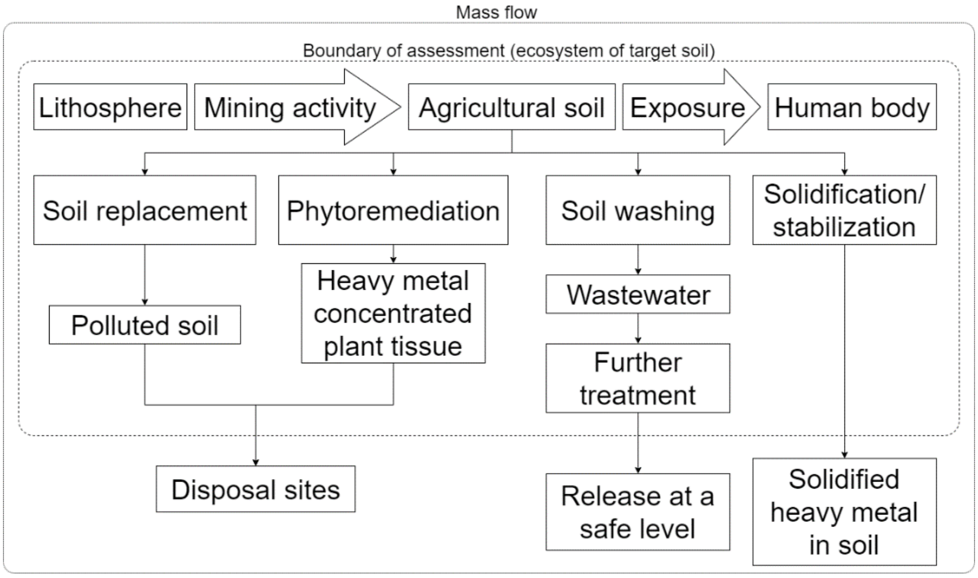

Fig 3-3. Mass balance of soil remediation under normal scenario

In the demonstration of average cost reported by FRTR (2012), these end-point processes are also mentioned. For soil replacement, the reported average cost (\$270 ~ \$460 per ton) includes excavation/removal, transportation, and **disposal at a RCRA permitted facility**. So the polluted soil will be transported to disposal facility as an end-point treatment. For soil washing, the key cost drivers of the reported average cost (\$ 70/m^3) are: economy of scale, quantity of material treated has a large impact, processor speed, and the **amount of waste being processed**. Therefore processing the waste material influences the cost of soil washing. For stabilization/solidification, the key cost drivers of the reported average cost (\$ 124/m^3) are: type of waste, moisture content in the sludge drives up costs compared to solid, contaminant concentration and type determine the amount of reagents added to the waste to attain the required treatment standards, size of the mobile s/s system, choosing the correct size mobile s/s system to adequately handle the throughput of waste volume. There is no information about the treatment of the waste material because the amendments remain in the target soil with solidified heavy metals. The cost information from FRTR included some of the end-point treatments. For phytoremediation, as reported by Wan et al. (2016), the cost of phytoremediation with Sedum alfredii H. and Pteris vittate includes initial capital and operational cost. The operational cost included the cost of labor and materials, cost of using large machines, and the other direct or indirect costs. Furtherly, The cost of using large machines included rent for machines during harvest, incineration, and **disposal of dangerous waste**. So the disposal of heavy metal concentrated plant tissues and other harmful waste materials is also included in the cost of phytoremediation. Judging from this information, we assume that in this research, the environmental effect of the end-point treatments of the waste materials generated during the remediation process is included in the average cost.

3.2.3 Assessment of Heavy metal exposure and disability-adjusted life years

3.2.3.1 Soil Heavy Metal concentration

The $c_{i,j,k}$ is defined as the soil HM concentration for scenario i ($0 \leq i \leq 4$), the jth year from remediation application ($0 \leq j \leq 75$, $j = 0$ means before remediation, $j \geq 1$ means after remediation) and k each type of HM ($k = 1$ for As; $k = 2$ for Cd, $k = 3$ for Pb). Correspondingly, the $\sigma c_{i,j,k}$ is the standard deviation of $c_{i,j,k}$. The reduction in HM concentration was regarded proportionally to avoid complicated calculation procedures. Therefore, for any $\sigma c_{i,j,k}$ ($1 \leq i \leq 4$):

$$\sigma c_{i,j,k} / \sigma c_{0,1,k} = c_{i,j,k} / c_{0,1,k} \tag{3-1}$$

For scenario 0:

$$c_{0,j,k} = c_{0,0,k} \ (1 \leq j \leq 75) \tag{3-2}$$

For scenarios 1–3, the remediation was assumed to be applied within 1 year (Chen & Li, 2018; Liu et al., 2018), and the soil HM concentration stayed constant afterward. The HM concentration in the soil after remediation under scenarios 1–3 was calculated by RE, where $RE_{i,k}$ is the removal efficiency of the ith scenario and the kth HM (**Table 3.2**).

$$c_{i,j,k} = \begin{cases} c_{0,j,k} & (j = 0) \\ c_{0,j,k} \times (1 - RE_{i,k}) & (1 \leq j \leq 75) \end{cases} \tag{3-3}$$

For scenario 4, as phytoremediation usually takes more than 1 year to remove the HM in soil (Liu et al., 2018), the length of remediation duration ($RD_{4,k}$) must be evaluated. $RD_{4,k}$ is estimated as 8 years. The estimation of remediation duration is shown in **Supplemental Material S2** and **Table S1**.

The soil HM concentration of jth year from the beginning of remediation under scenario 4 is as follows:

$$c_{4,j,k} = \begin{cases} c_{0,j,k} - j \times \Delta c_k & (0 \le j < RD_{4,k}) \\ c_{0,j,k} \times (1 - RE_{4,k}) & (RD_{4,k} \le j < 75) \end{cases} \quad (3\text{-}4)$$

where Δc_k is the annual reduction of soil concentration of the kth HM in scenario 4. For the calculation of remediation cost, the longest duration was considered.

$$RD_4 = \max(RD_{4,k}) \quad (1 \le k \le 3) \quad (3\text{-}5)$$

Table 3-2. Scenario information

Scenario	Remediation technology	UC_i (Unit Cost)	Removal efficiency ($RE_{i,k}$)		
			As	Cd	Pb
0	None (baseline scenario)	0	0%	0%	0%
1	Soil replacement	$208/m^3$ [a]	42.92% [b]	98.07% [b]	81.27% [b]
2	Soil washing	$70/m^3$ [a]	65.6% [c]	33.8% [c]	42.9% [c]
3	Solidification/stabilization	$124/m^3$ [a]	12% [d]	71% [e]	92% [e]
4	Phytoremediation	$3.47 + 2.03(annual) /m^2$ [f]	42.92% [b]	98.07% [b]	81.27% [b]

References: a: (FRTR, 2012); b: assumption of this research; c: (Qiu et al., 2010); d: (Wu et al., 2020) e: (Houben et al., 2013); f: (Wan et al., 2016)

3.2.3.2 Average daily dose (ADD)

This research considers four main exposure routes: soil ingestion; soil particle inhalation; groundwater consumption, and crop consumption (**Fig 3-2 a**). All these four exposure routes are affected by the HM concentration of surface soil. The exposure dose via direct soil ingestion and soil particle inhalation is directly affected by the HM concentration. The soil HM concentration will influence the concentration in agricultural products and groundwater, further influencing the other two exposure routes. The average daily dose is estimated as $ADD_{i,j,k,m}$, where i is for scenario i ($0 \le i \le 4$), j is for the jth year from remediation application ($0 \le j \le 75$, j = 0 means before remediation, $j \ge 1$ means after remediation), k is for each type of HM (k = 1 for As; k = 2 for Cd, k = 3 for Pb) and m is for each exposure route (m=1 for soil ingestion, m=2 for soil particle inhalation, m=3 for groundwater consumption and m=4 for crop consumption).

For soil ingestion; soil particle inhalation, and groundwater consumption, the heavy metal exposure dose is calculated by the GERAS-1 (Geo-Environmental Risk Assessment System, tier-1) model simulation (Kawabe et al., 2003). GERAS-1 is an onsite screening exposure assessment model developed based on another risk assessment model, CSOIL (Brand et al., 2007). The GERAS-1 model can assess human exposure to HMs and organic matter at the site where the pollutants are directly released. GERAS-1 estimates the pollutant concentration based on the fugacity of pollutants between soil solid particles, soil pore-water, and soil pore-air. The concentration in the exposure media (atmosphere, crops, groundwater) is estimated by the transfer process from soil to those exposure media. Finally, the exposure dose is estimated with the concentration in the exposure media according to the human exposure route, activity pattern, and living environment. Detailed information (equations) on the models can be found in the **supplemental material S4** and (Brand et al., 2007; Kawabe et al., 2003). The model input requires soil HM concentration, soil parameters, and exposure parameters. The parameters are fixed values for the scenario, and with the input of soil HM concentration ($c_{i,j,k}$), $ADD_{i,j,k,m}$ ($1 \leq m \leq 3$) is estimated by GERAS-1. The value of soil parameters and exposure parameters applied to GERAS-1 is shown in **Table 3-3**.

Table 3-3. Parameters applied for GERAS-1 simulation (because no specific temporal definition is applied for the scenario, default normal temperature is applied for temperature).

Soil parameters	Values	Exposure parameters	Unit	
			Adults	Children
Air fraction (%)[a]	0.21	Time outside (weekday) (h/d)[d]	0	2
Water fraction (%)[a,b]	0.2	Time inside (weekday) (h/d)[d]	24	22
Solid fraction (%)[a,b]	0.59	Time outside (weekend) (h/d)[d]	4	5
pH (-)[b]	6.2	Time inside (weekend) (h/d)[d]	20	19
Temperature (K)	300	Years living (y)[d]	64	6
Organic carbon (%)[c]	0.017	Body weight (kg)[e]	62	20
Clay content (%)[b]	0.18	Soil direct ingestion (g/d)[e]	50	85
Density (g/cm^3) [a]	1.33	Groundwater consumption (L/d)[d]	2	1
		Inhalation rate (m^3/d)[e]	16.1	8.8

Reference: a: (Tuyishimire et al., 2022); b: (Tang et al., 2014); c: (He et al., 2009); d: default settings in GERAS-1 (Brand et al., 2007; Kawabe et al., 2003); e: (Duan et al., 2015; Zhao et al., 2016).

The main crop in the target area is wheat (Ministry of Civil Affairs of PRC, 2016, **supplemental material S3**). In GERAS-1, the intake of crops is estimated separately as leaf crops and root crops. However, either of these two categories is suitable for wheat. Therefore, the exposure dose by wheat intake was calculated as follows (US EPA, 2002):

$$\mathrm{ADD_{i,j,k,4}} = \mathrm{c_{i,j,k}} \times 1000 \times \mathrm{BCF_k} \times \mathrm{IngR} \times \mathrm{EF} \times \mathrm{ED}/(\mathrm{BW} \times \mathrm{AT}) \tag{3-6}$$

Where $ADD_{i,j,k,4}$ is the average daily dose of crop intake (μg/kg-BW/day), $c_{i,j,k}$ is the HM concentration in soil (mg/kg), IngR is the wheat ingestion rate (g/day), EF is the exposure frequency (day/year), ED is the exposure duration (year), BW is body weight (kg), and AT is the average exposure time (day). The BCF_k is the ratio of the kth HM concentration in plants and soil (Liu et al., 2007). An average bio-concentration factor of wheat under different soil HM concentrations reported by (Wang et al., 2017) was applied in this research (**Table S2**).

As a result of the exposure assessment, all exposure dose of the four exposure routes is summed as a total average daily dose ($ADD_{i,j,k}$).

$$\mathrm{ADD_{i,j,k}} = \sum_{\mathrm{m=1}}^{4} \mathrm{ADD_{i,j,k,m}} \tag{3-7}$$

The standard deviation of the average daily dose ($\sigma ADD_{i,j,k}$) is required for DALY calculation. In this research, we assume that the daily dose and HM concentration are changed proportionally. Therefore, we have ($\sigma c_{i,j,k}$ is the standard deviation of $c_{i,j,k}$):

$$\mathrm{\sigma ADD_{i,j,k}}/\mathrm{\sigma c_{i,j,k}} = \mathrm{ADD_{i,j,k}}/\mathrm{c}_{i,\mathrm{j,k}} \tag{3-8}$$

3.2.3.3 Quantification of disability-adjusted life years

The methodology for quantifying DALY was developed by Murray (1994). DALY is a health impact indicator that quantifies the health risk by combining the dose–response relationship with the impact of diseases or disabilities on the human body. Compared with traditional indicators applied in CBA for soil remediation (Huysegoms et al., 2018; Stewart & Purucker, 2011), DALY is expected to create a stronger connection between exposure dose and endpoint effect. Thus presenting a more comprehensive health impact level for CBA. In this research, no information is available on proper average age of onset, therefore age-weighting is not considered. Time discount rate is discussed in sensitivity analysis. In this research, DALY is defined by years life lost (YLL) and years lost to disability (YLD).

$$\mathrm{DALY} = \mathrm{YLL} + \mathrm{YLD} \tag{3-9}$$

The exposure is defined as life-long exposure, and it is not expected to cause a fatal effect. Therefore, in this research, YLL was assumed as zero. The yearly DALY of the jth year under the ith scenario caused by the exposure of kth HM ($DALY_{i,j,k}$) was calculated as follows:

$$\mathrm{DALY_{i,j,k}} = \text{Disease Probability}\left(\mathrm{DP_{i,j.k}}\right) \times \text{Disease Weight}\left(\mathrm{DW_k}\right) \tag{3-10}$$

Disease probability is defined by exposure dose and dose–response relationship as:

$$DP_{i,j,k} = \int f_{i,j,k}(x)\phi_k(x)dx \tag{3-11}$$

Where x is the exposure dose (μg/kg-BW/day), $f_{i,j,k}(x)$ is the possibility distribution of exposure dose of the kth HM in the jth year under scenario i, and $\phi_k(x)$ is the possibility distribution of dose–response relationship of the kth HM. In this research, $f_{i,j,k}(x)$ and $\phi_k(x)$ were assumed to follow lognormal distribution.

$$f_{i,j,k}(x) = (1 - x\sigma_{i,j,k}\sqrt{2\pi})\exp\left(-\left(\ln x - \mu_{i,j,k}\right)^2/2\sigma_{i,j,k}^2\right) \tag{3-12}$$

$$\phi_k(x) = (1 - x\sigma_k\sqrt{2\pi})\exp\left(-(\ln x - \mu_k)^2/2\sigma_k^2\right) \tag{3-13}$$

For $f_{i,j,k}(x)$, $\sigma_{i,j,k}$ and $\mu_{i,j,k}$ could be derived from the mean value ($ADD_{i,j,k}$) and standard deviation ($\sigma ADD_{i,j,k}$) of exposure dose.

$$\mu_{i,j,k} = ADD_{i,j,k}^2/\sqrt{ADD_{i,j,k}^2 + \sigma ADD_{i,j,k}^2} \tag{3-14}$$

$$\sigma_{i,j,k} = \exp\left(\sqrt{\ln\left((\sigma ADD_{i,j,k}/ADD_{i,j,k})^2 + 1\right)}\right. \tag{3-15}$$

For $\phi_k(x)$, σ_k and μ_k are defined by:

$$\mu_k = \ln(GM_{res}) \tag{3-16}$$

$$\sigma_k = \ln(GSD_{res}) \tag{3-17}$$

Where GM_{res} is the geometrical mean of the dose–response relationship, and GSD_{res} is the geometrical standard deviation of the dose–response relationship. For all HMs, GSD_{res} is defined by the difference in metabolizing rate (GSD_1) and sensitivity (GSD_2) in the human body, that is, 1.4 and 2.7 (μg/kg-BW/day), respectively (Masuyama, 1977; Nordberg & Strangert., 1976).

$$\left(\ln(GSD)\right)^2 = \left(\ln(GSD_1)\right)^2 + \left(\ln(GSD_2)\right)^2 \tag{3-18}$$

(Nakanishi et al., 2003) reported a definition of GM_{res} of Hg by assuming that the no observable adverse effect level ($NOAEL_k$) could induce a 0.5% possibility of disease. In the present research, this assumption was applied to the target HMs as follows:

$$0.005 \fallingdotseq \phi_k(GM_{res}/GSD_{res}^{2.58}) = \phi_k(NOAEL_k) \tag{3-19}$$

$$GM_{res} = NOAEL_k \times GSD_{res}^{2.58} \quad (3\text{-}20)$$

The NOAEL applied in this research was 0.8 μg/kg-BW/day for As and 10 μg/kg-BW/day for Cd (U.S. EPA, 1987, 1988). The NOAEL for Pb ($NOAEL_3$) was calculated from the NOAEL blood Pb level (50 μg/L). For adults (Kobayashi, 2006):

$$\text{NOAEL blood Pb level} = 0.09 \times NOAEL_3 \times BW \quad (3\text{-}21)$$

The NOAEL intake level of Pb ($NOAEL_3$) was 8.96 μg/kg-BW/day.

Table 3-4. Parameters applied in estimation for this research

Parameter	Description	Value	Unit	Reference
$c_{0,j,1} \pm \sigma c_{0,j,1}$	Original soil As concentration ± standard deviation	19.62 ± 7.56	mg/kg	a
$c_{0,j,2} \pm \sigma c_{0,j,2}$	Original soil Cd concentration ± standard deviation	5.03 ± 2.90	mg/kg	a
$c_{0,j,3} \pm \sigma c_{0,j,3}$	Original soil Pb concentration ± standard deviation	138.97 ± 58.03	mg/kg	a
c_{BG1}	China background soil concentration of As	11.2	mg/kg	b
c_{BG2}	China background soil concentration of Cd	0.10	mg/kg	b
c_{BG3}	China background soil concentration of Pb	26.00	mg/kg	b
$RD_{4,1}$	Remediation duration of As	1	year	c
Δc_1	Annual soil As concentration reduction in scenario 4	19.01	mg/kg	c
$RD_{4,2}$	Remediation duration of Cd	8	year	c
Δc_2	Annual soil Cd concentration reduction in scenario 4	1.96	mg/kg	c
$RD_{4,3}$	Remediation duration of Pb	3	year	c
Δc_3	Annual soil Pb concentration reduction in scenario 4	15.76	mg/kg	c
IngR	Wheat ingestion rate	143.5	g/day	d
EF	Exposure frequency	365	day/year	e
ED	Exposure duration	1	year	e
AT	Averaged exposure time	365	day	e
BW	Body weight	61.9	kg	d
L	Life expectancy	75	year	d
A	Remediation area	9.3×10^6	m^2	f
D	Depth of remediated soil	0.2	m	a
IC_4	Initial cost of phytoremediation	3.47	$/m^2$	g
AC_4	Annual cost of phytoremediation	2.03	$/m^2$	g
P	The population of the target area	46000	people	h
FB	The average financial burden of DALY	108,600	$/year of DALY	i

References: a: (Jin, 2017b); b: (Liang et al., 2021); c:(author's estimation, supplemental material S2); d: (Duan et al., 2015); e: (EPA, 1989b); r: (Ministry of Civil Affairs of the People's Republic of China, 2016); g: (Wan et al., 2016); h: (National Bureau of Statistics, 2021); i: (WHO, 2001; Chowdhury et al., 2020)

The DW_k values of the three target HMs (Lopez et al., 2006) were 0.056 for DW_1 (hyperpigmentation caused by As exposure), 0.098 for DW_2 (end-stage renal disease caused by Cd exposure), and 0.090 for DW_3 (mild mental retardation caused by Pb exposure). Value of parameters applied in health impact assessment is listed in **Table 3-4.**

3.2.4 Cost Benefit Analysis

The three chosen endpoint effects are either fatal or dependent on each other. The specific antagonistic or synergistic interactions among As, Cd, and Pb on these endpoint effects could not be identified. Therefore, as reported by previous researchers (Han et al., 2021; Naddafi et al., 2022), the health impact of HMs was simply summed for a total impact. For scenario i ($1 \leq i \leq 4$), the reduction in personal lifetime DALY under scenario i ($\Delta DALY_i$) was calculated by the sum of the yearly DALY of each year and each type of HM under this scenario.

$$\Delta \mathrm{DALY_i} = \sum_{j=1}^{75} \sum_{k=1}^{3} (\mathrm{DALY_{0,i,j}} - \mathrm{DALY_{i,j,k}}) \tag{3-22}$$

Considering no remediation action has been conducted in the target area, little detailed information on application cost is available. Therefore, the average cost derived from previous studies was applied in this research to evaluate the cost-effectiveness of each scenario. For scenarios 1–3, the cost was calculated by volume unit cost and volume of soil remediated as follows:

$$\mathrm{C_i} = \mathrm{A} \times \mathrm{D} \times \mathrm{UC_i} \tag{3-23}$$

where C_i is the cost of scenario i (\$), A is the remediation area (m^2), D is the depth of remediated soil (m), and UC_i is the unit cost of the ith remediation technology (\$/m^3).

Especially for phytoremediation (i = 4), as the remediation was assumed to last for more than 1 year, the cost was divided into the initial cost in the first year and the annual operation cost. The cost information of phytoremediation by Pteris vittata and Sedum alfredii H. was reported by Wan et al. (2016) as follows:

$$\mathrm{C_4} = \mathrm{A} \times (\mathrm{IC_4} + \mathrm{RD_4} \times \mathrm{AC_4}) \tag{3-24}$$

Where C_4 is the cost of scenario 4 (\$), IC_4 is the initial optional cost of the phytoremediation technology (\$/m^2), and AC_4 is the annual operation cost of the phytoremediation technology (\$/m^2). This research focused on health benefits for the benefit of remediation technologies. Therefore, benefit was calculated by the reduction in DALY and the financial burden of DALY as follows:

$$B_i = \Delta DALY_i \times P \times FB \tag{3-25}$$

Where B_i is the health benefit of remediation technology ($), ΔDALY is the DALY reduced due to the remediation in the ith scenario (year of DALY/person), P is the population of the target area (people), and FB is the average financial burden of DALY ($/year of DALY). Finally, the net present value (NVP) and benefit ratio (BR) of each scenario were calculated as follows:

$$NPV_i = B_i - C_i \tag{3-26}$$

$$BR_i = NPV_i / C_i \tag{3-27}$$

Where NPV_i is the net present value of the ith scenario, and BR_i is the benefit ratio of ith scenario. Parameters applied in cost benefit analysis is listed in **Table 3-4**.

3.2.5 Sensitivity analysis

In the sensitivity analysis, considering the major assumptions applied in this research, plausible pessimistic estimates were substituted for important indicators to discuss the influence of variation in input factors on the result of NPV assessment (CRC CARE, 2018). A corresponding case defines each variation. The result included 14 cases, including an "original case," in which no variation was applied and served as a reference for comparison. NPV and BR were estimated for each case, and the variation ratio in NPV of several cases compared with that in the original case was calculated and discussed.

$NPV_{i,t}$ and $BR_{i,t}$ are defined as the net present value and benefit rate for the ith scenario in case t, respectively. In addition, the variation ratio of NPV of the ith scenario in case t ($1 \le t \le 8$) is defined as $VR_{i,t}$. $NPV_{i,t}$ and $BR_{i,t}$ were calculated using the same procedure as that in $NPV_{i,t}$, and $BR_{i,t}$, only with a certain input parameter variation demonstrated in detail below. For $VR_{i,t}$:

$$VR_{i,t} = NPV_{i,t} / NPV_{i,0} \tag{3-28}$$

Case 0 is the original case in which no variation was applied. Therefore, $NPV_{i,0} = NPV_i$, and $BR_{i,0} = BR_i$. Variations of +20% and +40% for the unit cost (cases 1 and 2) and −20% and –40% for RE (cases 3 and 4) were applied to discuss how they could influence the result. In addition, the removal rate of phytoremediation was modified by plant extraction ability (HM concentration in the plant) and caused a change in remediation duration. Another removal rate was discussed in the sensitivity analysis. Soil depth was assumed as 0.2 m in this research. It could significantly influence the amount of soil to be remediated. Therefore, +20% and +40% variations were applied (cases 5 and 6). The health benefit of an entire life expectancy (75 years) was also estimated. In some studies, the assessment period of benefit was relatively shorter. Therefore, variations of –20% and –40% time spans (cases 7 and 8) were applied to the CBA to discuss relatively short-term benefits. Considering a discussion with similar variation was identified in previous research (Huysegoms et al., 2018, 2019a), the extent of variation (20% and 40%) was defined in accordance with the author's assumption.

Some additional independent cases are added to the calculation to further discuss the possible actual situations and the uncertainty of the result. 30-year and 10-year time spans (cases 9 and 10) were discussed for short-term benefit. A range of 2%–4% discount rate is recommended by the EU commission (2004), here we discuss 2% and 4% discount rates as independent cases (cases 11 and 12).

For remediation cost, we applied the minimum value ($270) of the reported range $270 ~ $460 per ton of soil for soil replacement, in the sensitivity analysis, we apply the maximum value ($460) as independent cases for soil replacement (cases 13). For the remediation cost of other alternatives, the data source did not provide variation ranges, so no independent cases are defined.

As for removal efficiency, for soil replacement, we applied the national background value, in the sensitivity analysis, we apply the background value of Yunnan Province (18.40 mg/kg As, 0.22 mg/kg Cd and 40.60 mg/kg Pb) (Cheng et al., 2018), the province where the target area is located, as an independent case (case 14). For soil washing, we applied the maximum removal rate reported by Qiu et al.（2010), here in the sensitivity analysis, we apply the minimum removal rate (54.7% As, 28.6% Cd and 15.8% Pb) as an individual case (case 15). For solidification/stabilization with biochar, no additional information is available on the removal rate, so we do not define independent cases. For phytoremediation, the RE was assumed to be the same as soil replacement (HM level decreases to background level). In the reference research by Wan et al. (2016), the HM removal rate was 55.3% for As, 85.8% for Cd, and 30.4% for Pb. These removal rates were applied as an independent case (case 16).

All the mathematical calculations in this research were performed in Microsoft Excel for Microsoft 365 MSO (version 2207 build 16.0.15427.20182) 64-bit.

3.3. Results

3.3.1 Reduction in health impact

The average personal daily exposure dose under each scenario is shown in **Fig 3-4**. Soil washing was most effective in As exposure reduction (75.31% reduction rate), soil replacement was most effective in Cd exposure reduction (98.07% reduction rate), and stabilization/solidification was most effective in Pb exposure reduction (89.29% reduction rate). Each remediation alternative has different advantage in HM removal. This indicates different remediation technology may be suitable for sites with different major pollutants.

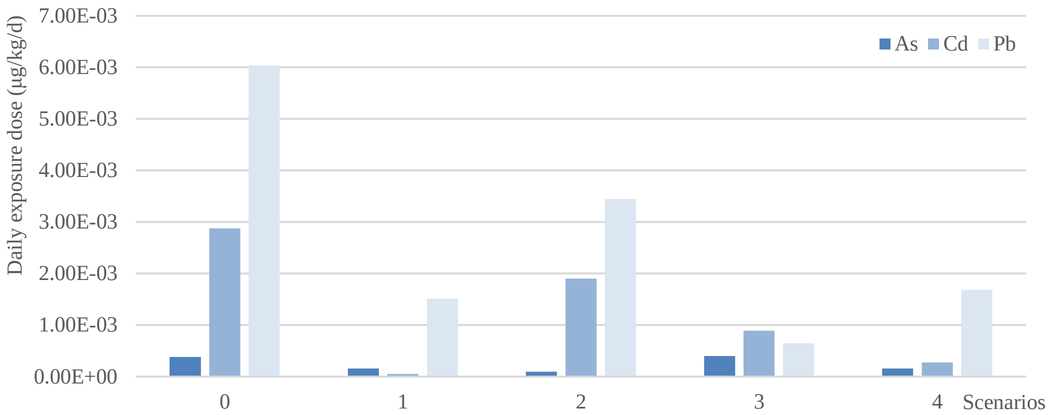

Fig 3-4. Daily exposure dose of each scenario. Scenario 0: baseline scenario. Scenario 1: soil replacement. Scenario 2: soil washing. Scenario 3: stabilization/solidification. Scenario 4: phytoremediation (daily exposure dose of scenario 4 is a lifetime averaged value

The personal lifetime DALY after remediation and the reduction rate of DALY in each scenario are shown in **Fig 3-5**. The personal lifetime DALYs were 2.71×10^{-1}, 5.29×10^{-4}, 4.45×10^{-2}, 2.26×10^{-3}, and 5.94×10^{-3} (years per lifetime exposure) for scenarios 0–4, respectively. Scenario 0 showed the original DALY level of the residents. All four technologies could effectively reduce the DALY level of the residents. Soil replacement was most effective in health impact reduction (99.81% reduction rate). Soil washing exhibited the lowest reduction rate (83.60% reduction rate). The relatively low reduction rate for soil washing was due to the relatively low Pb removal rate. The original DALY caused by Pb exposure was the highest among the three target elements. Therefore, improving the chemical agent's Pb removal efficiency could greatly improve soil washing efficiency. Even though the removal rate of phytoremediation was assumed to be the same as soil replacement, exposure and health risk were assessed for the relatively long remediation period of phytoremediation. The general health impact reduction in scenario 4 was lower than in scenario 1.

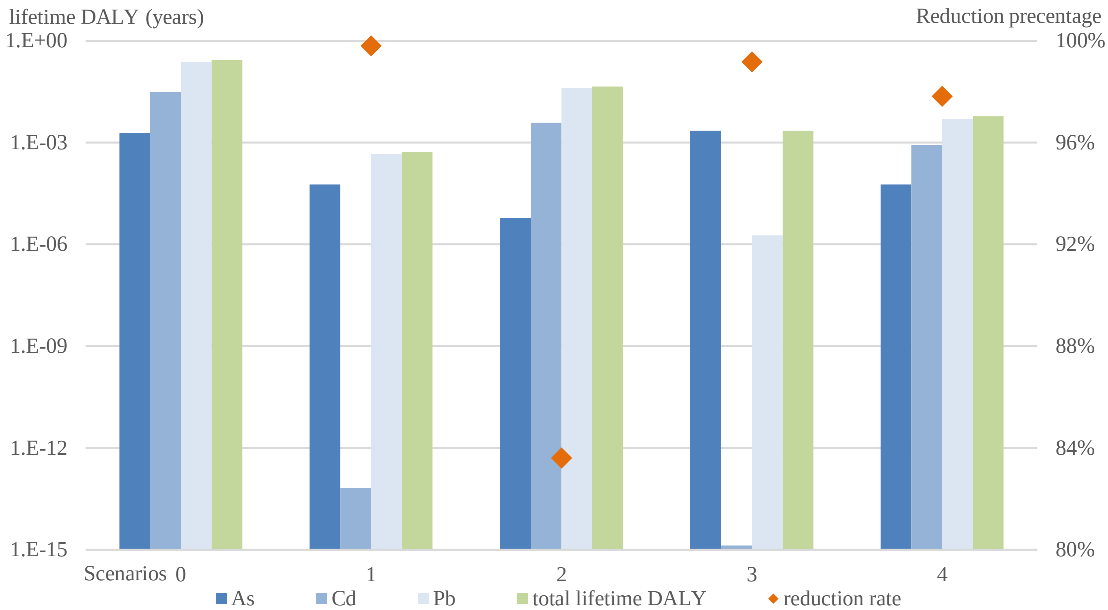

Fig 3-5. Personal lifetime DALY and reduction rate of each scenario (Scenario 0: baseline scenario; scenario 1: soil replacement; scenario 2: soil washing; scenario 3: stabilization/solidification; scenario 4: phytoremediation)

3.3.2 Cost Benefit Analysis

The cost and benefit of each scenario are shown in **Fig 3-6**. The costs of remediation were 0.65, 0.13, 0.23, and 0.18 (billion $) for scenarios 1–4, respectively The cost of soil replacement was the highest among the costs in all four scenarios. Even though phytoremediation is normally considered an economically promising remediation technology (Cunningham et al., 1995), the long remediation period (8 years) resulted in a higher cost of phytoremediation than soil washing in the present study.

Considering the remediation benefit, the benefits of soil replacement, stabilization/solidification. and phytoremediation were over $1.3 billion. The benefit of soil washing was relatively lower than that of the other three technologies due to the low RE of Pb discussed previously. Under all four scenarios, the benefit exceeded the remediation cost, indicating that all the remediation technologies in this study are economically feasible.

NVP and BR are presented in **Fig 3-6**. The NVP of phytoremediation was the highest ($1.14 billion), closely followed by stabilization/solidification ($1.11 billion). The NPVs of soil washing ($1.00 billion) and soil replacement ($0.70 billion) were relatively low. As soil washing has a very low cost, it still showed the highest BR even though its total benefit was the lowest (770.78%), indicating that soil washing is the most economically feasible technology in this study. The BR of phytoremediation was close to that of soil washing. The BR of soil replacement was the lowest (107.32%) but was still a positive value. The results confirmed the economic feasibility of the remediation technologies on the target agriculture soil.

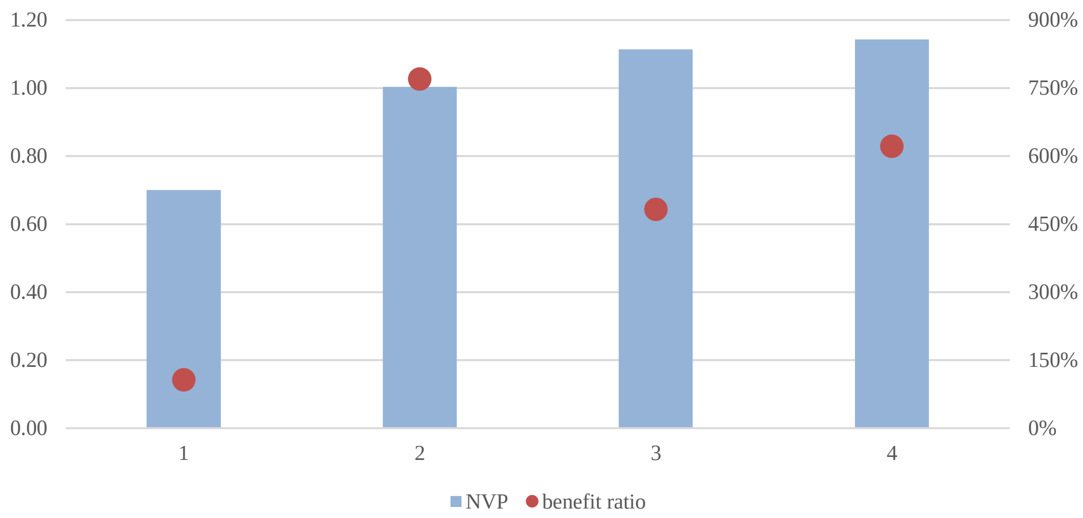

Fig 3-6. Cost, benefit, net present value (NPV), and benefit ratio (BR) of each scenario (Scenario 1: soil replacement; scenario 2: soil washing; scenario 3: stabilization/solidification; scenario 4: phytoremediation.)

3.3.3 Sensitivity analysis

Table 3-5. Result of $NPV_{i,t}$ and $BR_{i,t}$ from sensitivity analysis

Case	Soil replacement		Soil washing		Solidification/stabilization		Phytoremediation	
	$NPV_{1,t}$	$BR_{1,t}$	$NPV_{2,t}$	$BR_{2,t}$	$NPV_{3,t}$	$BR_{3,t}$	$NPV_{4,t}$	$BR_{4,t}$
0	7.01E+08	107.32%	1.00E+09	770.78%	1.11E+09	483.09%	1.14E+09	622.36%
1	5.70E+08	72.76%	9.78E+08	625.65%	1.07E+09	385.91%	1.11E+09	502.52%
2	4.39E+08	48.08%	9.51E+08	521.99%	1.02E+09	316.49%	1.07E+09	416.45%
3	6.19E+08	94.79%	8.49E+08	651.76%	1.09E+09	473.48%	1.11E+09	550.17%
4	3.07E+08	47.07%	6.58E+08	505.57%	8.70E+08	377.08%	1.02E+09	391.52%
5	5.60E+08	70.63%	9.76E+08	616.69%	1.06E+09	379.91%	1.12E+09	552.59%
6	4.39E+08	48.08%	9.51E+08	521.99%	1.02E+09	316.49%	1.08E+09	448.00%
7	4.30E+08	65.85%	7.77E+08	596.62%	8.45E+08	366.47%	8.74E+08	475.79%
8	1.59E+08	24.39%	5.50E+08	422.47%	5.76E+08	249.85%	6.03E+08	328.37%
9	-1.11E+08	-17.07%	3.23E+08	248.31%	3.07E+08	133.23%	3.32E+08	180.95%
10	-4.72E+08	-72.36%	2.10E+07	16.10%	-5.13E+07	-22.26%	-2.87E+07	-15.61%
11	4.51E+07	6.91%	4.54E+08	349.06%	4.63E+08	200.69%	4.90E+08	266.76%
12	-2.26E+08	-34.54%	2.28E+08	174.94%	1.94E+08	84.10%	2.20E+08	119.87%
13	2.41E+08	21.69%	-	-	-	-	-	-
14	6.68E+08	102.27%	-	-	-	-	-	-
15	-	-	4.66E+08	358.24%	-	-	-	-
16	-	-	-	-	-	-	1.00E+09	1125.83%

The NPV and BR of each case are presented in **Table 3-5**. Cases 1–8 showed the output of the result with variations of 20% and 40% of four selected input parameters: unit cost, RE, soil depth, and timespan. The results showed that within 40% variation of the selected parameters, all the output NPV values remained positive, indicating that the remediation technologies could remain economically feasible under 40% variation of input parameters. This finding provides evidence for the robustness of the results in this research.

For cases 9–12, independent cases were discussed for time spans and discount rates. With a 30-year time span (case 9), the NPVs of soil replacement became negative. With a 10-year time span (case 10), the NPVs of all the scenarios except soil washing also became negative. With a discount rate of 2% (case 11), the NPVs remained positive, but with a 4% discount rate (case 12), the NPV of soil replacement became negative. These four cases showed that time-relevant parameters might strongly influence the output. In case 13, a different assumption for phytoremediation removal rate was applied. Consequently, the NPV output of phytoremediation dropped by approximately 12.3%. However, a considerable increase in BR was observed, because changing the HM removal amount decreased the remediation duration demanded by phytoremediation, causing a decrease in the cost.

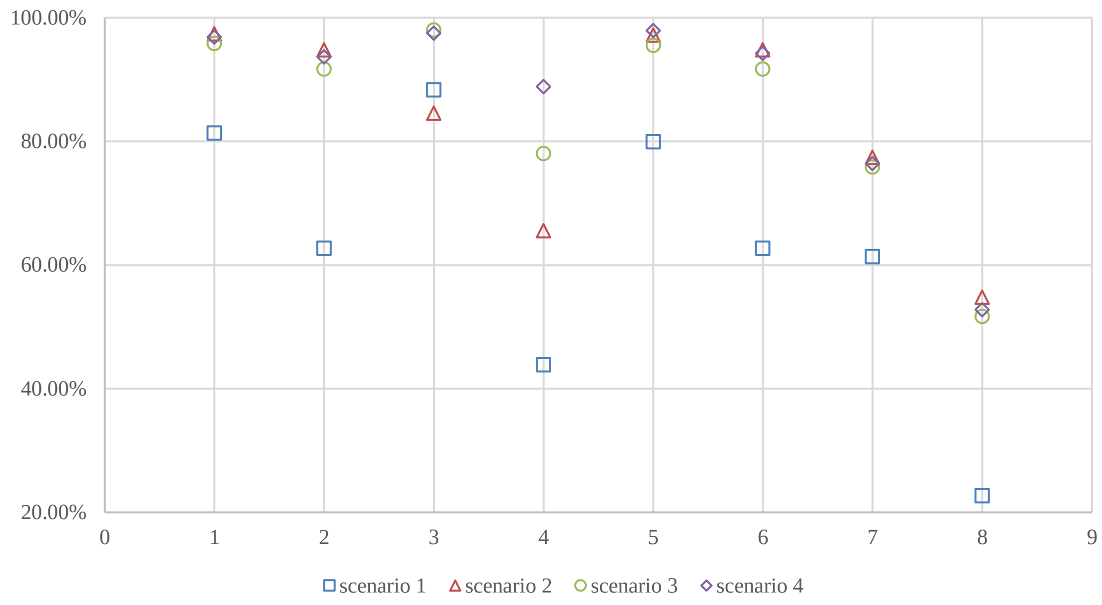

Fig 3-7. Result of $VR_{i,t}$ (variation ratio of NPV) of case 1~8 in sensitivity analysis (case 1: unit cost -20%; case 2: unit cost -40%; case 3: soil depth +20%; case 4: soil depth +40%; case 5: removal efficiency -20%; case 6: removal efficiency -40%; case 7: time span -20%; case 8: time span -40%)

The variation of NPV in cases 1–8 compared with that in case 0 is shown in **Fig 3-7**. The results showed how much the output varies with different variations of input parameters. Among the four parameters selected for the sensitivity analysis, time span had the strongest influence on the output. The influence of cost and RE was relatively weak. Among all the four scenarios, soil replacement was most affected by the variation in parameters, indicating the economic feasibility of soil replacement may be less stable than that of other technologies. As for independent cases, compared to case 0, the cost of soil replacement increased in cases 13 and the removal rate of soil replacement decreased in cases 14. Both NPV and BR decreased to a certain extent. In case 15, a lower removal rate is applied for soil washing, the NPV and BR also decreased. In case 16, the NPV of phytoremediation decreased due to lower removal efficiency, but the remediation duration also decreased, leading to a lower cost and higher BR.

3.4. Discussion

3.4.1 Health benefits of agriculture soil remediation

CBA for soil remediation is a significant research topic. Many researchers contributed to this research field, but the benefit types differed for each study. The present research focused on health benefit, whereas some previous studies focused on other types of benefit. For example, Lavee et al. (2012) assessed the benefit on the basis of the property value of industrial areas in Israel. The results indicated that the one-time benefit had a BR of over 1400%. Wan et al. (2016) assessed the cost and benefit of phytoremediation of agricultural land on the basis of ecosystem benefit, human income, and cash crops. According to the estimation, in a 75-year time span, the BR could reach 746%. The BR of health benefits assessed in the present study was between 122.86%–729.78% for four different alternatives. Comparison of these types of benefits showed that the BR of health benefit in this research was relatively low. However, the benefit level was still unneglectable compared with the levels of other types of benefits reported by previous research. Therefore, health benefit is an important type of benefit, and it should be included in the CBA for soil remediation.

Some other previous researchers have discussed the health benefit of soil remediation for different types of polluted sites. For example, Huysegoms et al. (2019) assessed the health benefit of soil remediation at a dry cleaning facility. They found that remediation is not economically feasible, with approximately −64% and −71% BR for chemical and natural remediation technology, respectively. Zheng et al. (2019) assessed remediation for river sediment, with an estimated cost of TWD 4.7 billion and a benefit of TWD 6.3 billion. The total benefit overruns the cost. However, 81.6% of the total benefit came from ecosystem improvement, and health benefit is a minor type of benefit. In the present research, health benefit considerably exceeded the remediation cost. Even soil replacement, which is the least economically feasible technology in this research, generated a 122.86% benefit rate. The benefit rate of agricultural soil remediation was much higher than that of remediation in other types of sites. One of the main reasons that cause this difference is the exposure route defined by the function of the target area. Comparison of the industrial area and river sediments showed that agricultural soil affects the HM concentration in crops consumed by residents. The exposure dose by dietary intake is much higher than the direct ingestion or inhalation of soil particles (Lian et al., 2019). Therefore, HM contamination of agricultural soil has a higher impact on human health, and it increases the potential health benefit obtained by soil remediation. Moreover, site-specific factors heavily influence the economic feasibility of soil remediation, and health benefit is much more important in the remediation of agricultural lands than other types of contaminated sites.

In the sensitivity analysis, a comparison among the four selected parameters showed that the variation in time span had the strongest influence on the output among other parameters. In cases 9 and 10, a further variation of time span was discussed, showing that short-term health benefit is not economically feasible. In cases 11 and 12, negative NPV values also appeared, indicating a strong influence on the discount rate.

For independent cases, cases 13 and 14 shows scenario definitions using different cost value and refilled soil heavy metal level. The cost in case 13 ($460/ton) is about 1.7 times of the original cost ($270/ton). The result caused a relatively low NPV and BR. But the result remained positive, indicating the analysis result is robust within this cost variation range. In case 14, the difference is not obvious between using China and Yunnan soil background value. So, as long as the refilled soil is fresh, uncontaminated, the result of soil replacement will not be strongly influenced. In case 15, the lower removal efficiency of soil washing is applied, this variation has a strong influence on the result, reducing over a half of the NPV and BR. Soil washing is no longer the most economic feasible technology, with a BR lower than solidification/stabilization and phytoremediation. In case 16, compared with other benefits, such as property value (Lavee et al., 2012), human income, and ecosystem service function (Wan et al., 2016), the health benefit assessed in the present research is a yearly benefit that is highly time-dependent. Therefore, time span and discount rate, of which the influence increases with time, could strongly influence the output. These parameters require rigorous definition in practice. The result of these independent cases remained positive, supporting the robustness of the analysis result of this research. But the variation can be violent, indicating that the parameters such as removal rates and costs requires well rounded consideration.

3.4.2 Identification of potentially best technology

As Huysegoms et al. (2019) argued, even though the assessment shows that the remediation is not economically feasible, it does not mean that the decision-makers should wait for a more efficient technology to be developed. Economic feasibility is not the only criterion for selecting remediation technologies. In the present research, even though soil washing was the most economic feasible technology, it could not be considered the best technology for the target site. In the target area, the HM element with the highest health impact was Pb, indicating Pb should be the priority in soil remediation. However, soil washing, even with the highest BR, had a low RE of Pb. Soil replacement had a high HM removal rate, but the economic feasibility was much lower than that in other technologies due to the high cost. So, these two technologies are difficult to be considered the best technology. For the other two technologies, the NPV and BR of phytoremediation were higher than stabilization/solidification. Therefore, on the basis of the consideration of HM removal and economic feasibility, phytoremediation was identified as the potentially best technology for the target area of this research.

In accordance with the mechanism of the remediation process, the remediation technologies are categorized into physical/chemical and biological processes. Due to the function of agricultural lands and the nature of HMs, some technologies are not applicable to the target site, such as surface capping and thermal treatment. In this research, the alternative technologies were limited to three physical/chemical processes (soil replacement, stabilization/solidification, and soil washing) and one biological process (phytoremediation). The analysis showed that phytoremediation is potentially the best technology for the target site on the basis of cost and removal effectiveness. However, despite cost and removal effectiveness, phytoremediation has other advantages against physical/chemical processes. Phytoremediation introduces fewer artificial chemical substances into the soil system than chemical processes and causes less disturbance to the soil physical structure than physical processes. In addition, biological remediation was reported to be able to improve the agricultural soil fertility (Cui et al., 2017). The main disadvantage of phytoremediation is time consumption. Phytoremediation may cost 5–20 years for a polluted site (Entry et al., 1997). This high time consumption prevents commercial utilization of HM phytoextractors in practice. However, researchers are proposing approaches to accelerate the phytoremediation process, such as chelating agent enhancement and plant genetic manipulation (Liu et al., 2018). These approaches may decrease the time consumption of phytoremediation.

A notable detail is that the removal rate of phytoremediation is based on the assumption that the HM level could decrease to the background level. In case 13, the variation in RE caused the change in remediation duration, which further caused NPV and BR to change differently. The estimation of the economic feasibility of phytoremediation was more complicated than that of other technologies because of the change in remediation duration. The removability of hyper extractors could strongly influence the result, as discussed in the sensitivity analysis. Therefore, in practice, before applying phytoremediation, further laboratory or field-scale experiment is required to discuss the removal ability in the target site. However, even based on assumptions and secondary data, the result of this research is still expected to provide a strategic reference for local decision-makers, and the methodology of assessing the health benefit of agriculture soil remediation is applicable for different sites in future studies.

3.5. Conclusion

Several previous studies have been conducted to determine the economic cost-effectiveness of field-scale soil remediation actions for industrial sites. However, such knowledge is still limited to agricultural soil. In this research, CBA was conducted for the agricultural land in a typical Pb/Zn mining town in China. Four commonly applied soil remediation alternatives were discussed, and benefit estimation on focused on health benefits. For all the alternatives, the health benefit of soil remediation exceeded the remediation cost, indicating that the remediation is economically feasible. Health benefit was found to be more important in the remediation of agricultural soil than in other remediation sites. Compared with other benefits, the health benefit was at an unneglectable level. This result indicated the significance of including health benefits in CBA for soil remediation. Considering HM removal ability and economic feasibility, phytoremediation was identified as the potentially best technology for the target agriculture soil. Sensitivity analysis showed that within a 40% chance of major factors could not change the NPV to negative. The time span of the assessment and the discount rate may strongly influence the estimation result, indicating that the assessment of health benefits is highly dependent.

Data availability and the hypothetical scenario definition are the main limitations of this study. With the developed methodology, more sufficient site-specific data are expected to improve the quality of the result in future case studies. However, this research serves as an attempt to provide cost-effective information on soil remediation for HM-polluted agriculture, which was not fully studied previously. It could provide field-scale information and strategic reference for remediation application in the target area. The methodology applied for health benefit estimation is independent on-site and scenario definition, thus applicable for future studies.

CHAPTER 4: COST-BENEFIT ANALYSIS OF SOIL REMEDIATION UNDER ACCIDENTAL SCENARIO

In Chapter 2, a methodology is developed for DALY quantification. In Chapter 3, this methodology is applied in cost-benefit analysis for soil remediation alternatives under normal scenario. In this Chapter, the major assessment framework resembles Chapter 3, but we focus on accidental scenario. The scenario and parameter definition are based on the hypothetical accidental scenario. The health impact estimated in this Chapter is also compared to Chapter 3 to reveal the impact of accident on the environment and residents' health.

4.1. Introduction

Previous researchers have to some extent discussed the economic feasibility of soil remediation under normal scenario. But the discussion has rarely been extended to accidental scenarios. Considering the target mining area in this research, tailing dam failure is selected as the target accident of this analysis.

Tailing dam is a common structure in mining areas, for which the purpose is to store solid and liquid wastes generated by mining activities, keeping them at a relatively safer state (Dong et al., 2020). Tailing dams have a higher risk of failure than normal reservoir dams due to the relatively more fragile structure. This caused heavy metal release potential to the environment (Liu et al., 2020). In recent years, the number of tailing dam failures in developing countries has been increasing (Lyu et al., 2019). In China, there were still more than 8869 mine tailings ponds active by the end of 2015, including 1425 tailings ponds close to residential areas and other important infrastructures within 1 km downstream (Dong et al., 2020; Liu et al., 2020). These tailing dams pose a high risk of tailing dam failure in China.

Research on tailing dam failure mainly focuses on the mechanism and severe consequence of the accident. Lyu et al. (2019) summarized the common reasons for tailing dam failures by reviewing past events. Other researchers discussed the consequences of past major events such as Aznalcóllar tailing dam failure in Spain (Hudson-Edwards et al., 2003) and Samarco dam collapse in Brazil (Davila et al., 2020). But few studies extended the discussion to the soil remediation process following the dam failure. The consequence of tailing dam failure is severe and remediation will be necessary for the recovery of ecosystem function and residents' safety. Identical to normal scenario, economic feasibility of soil remediation is significant for accident control. Information of soil remediation feasibility under accidental scenario is required by decision makers.

The resident always suffers the biggest impact in a tailing dam failure event. Other than the flooded land and suspension of the mining process, the health impact on the residents is another critical consequence and may have a long-term influence on the area. The studies mentioned above did include the health risk and economic point of view. However, to the author's knowledge, no study is available on the financial impact considering the health impact of tailing dam failure, as well as the corresponding remediation process.

Considering the research gaps, the objective of this research is 1) to assess the resident exposure of HMs and the corresponding health impact under the hypothetical tailing dam failure scenario; 2) to conduct a cost-benefit analysis for the remediation process based on the reduced health impact. This research fills the research gap by investigating the cost-effectiveness of soil remediation for tailing dam failures. The analysis procedure in this research provides a basic methodology for cost-benefit analysis based on health benefits under accidental scenarios. The result of this research is expected to provide reference on tailing dam emergency management for the local government and policymakers.

4.2. Material and method

4.2.1 Research area

The target area is agricultural land in Jinding Town. Jinding Town (26º25' N, 99º24' E) is located in Lanping county, Yunnan province, China. The altitude is about 2300 m. Under a monsoon climate, the average temperature is 11.7 ºC, and the annual average precipitation is 1002.4 mm (Li et al., 2019). Jinding Pb/Zn mine locates in the center of Jinding Town. Jinding deposit is the largest Pb/Zn deposit in China and the world's youngest sediment-hosted super giant Pb/Zn deposit (Xue et al., 2007). This area has had a mining history for several decades and has been polluted by mining activities.

Fig 4-1. (a): Research area (picture obtained from google earth). The two tailing dams are surrounded by the thick white line. The agricultural land surrounded by the thin white line is the target area of this research. The Bijiang River flows from north to south, crossing the area. (b): Households in target residential area (244 households in total, 2.62 people per household (Statista, 2022))

The target land was selected as the total area of agricultural land in the entire Jinding Town is 9.3 km^2. However, not all these agricultural lands will be possibly affected by a tailing dam failure, especially the lands located upstream from the tailing dams. Therefore, in this paper, the target area is selected as agricultural land in front of the tailing dam in the Jinding mining area (**Fig 4-1 a**), which has a high possibility of being flooded if a tailing dam failure happens. The target area is approximately 28 hectares. There are two tailing dams in the Jinding mining area. The target agricultural land is located in front of the two tailing dams and is the closest agricultural land to the tailing dams in this mining area. The shortest distance from the tailing dam to the target area is approximately 0.47 km. Under the hypothetical tailing dam failure event, the target land is assumed to be thoroughly and evenly affected by the tailings released.

4.2.2 Scenario definition

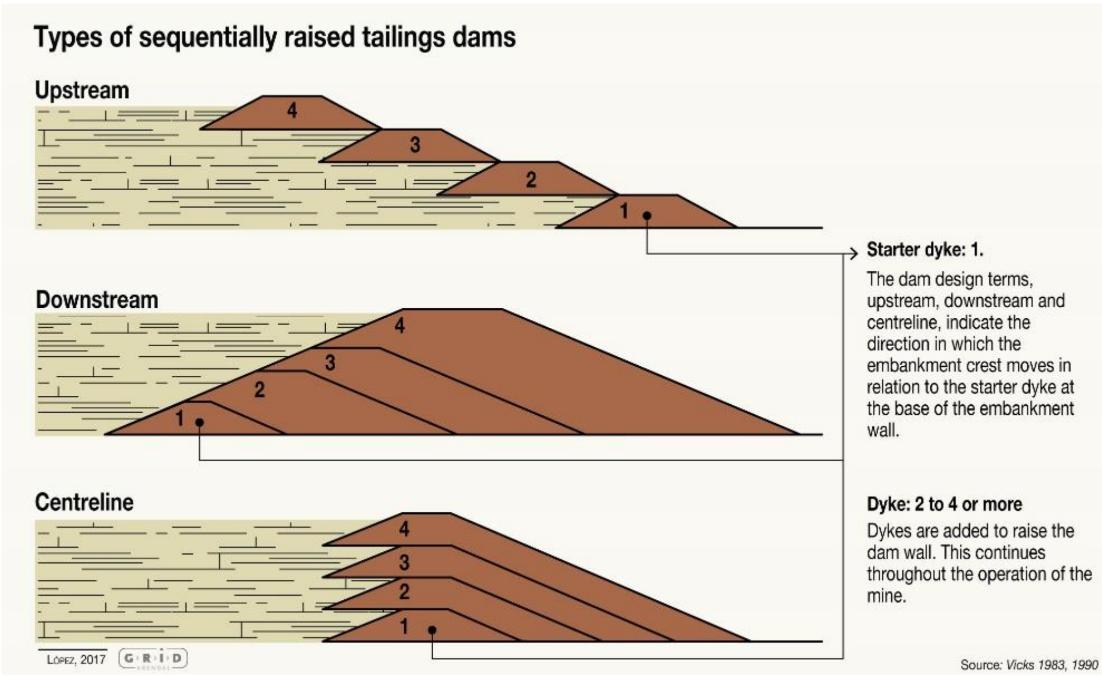

Fig 4-2. Three types of tailing dam structure (Thygesen, 2017; Vick, 1990).

The major reasons for tailing dam failure are structural instabilities, floodings and earthquakes (Lyu et al., 2019). There are three types of building structures for a tailing dam. Upstream, downstream, and centerline (**Fig 4-2**). Among these three types of structures, the upstream dams have a high phreatic line and poor stability. However, due to the lower requirement of construction materials, this is still a relatively common structure of the existing tailing dams (Lyu et al., 2019). The tailing dam in Jinding mining area also utilizes this structure, indicating a relatively higher risk of failure. **Fig 4-3** shows the hazard map of earthquake and flooding. The risks of earthquake and flood in southeast China, where the target area is located, are relatively higher than other areas in China. The risk level is at medium level at the target land, indicating a considerable chance of disaster happening. Therefore, judging from the structure feature of the target tailing dam and the disaster possibility of the target area, we consider it reasonable to assume accidental failure scenario for the target tailing dam. In this research, we focus on the scenario of tailing release and remediation process, so the actual reason of the tailing dam failure is not discussed.

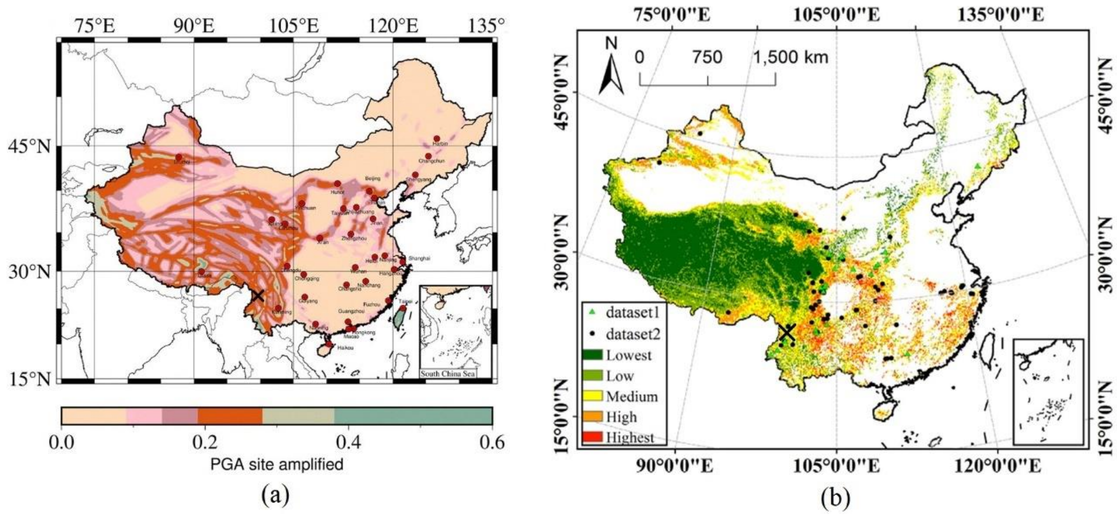

Fig 4-3. Hazard map and target area location. (a: earthquake hazard map (Ji et al., 2021) ; b: flooding hazard map (Zhao et al., 2018). The X mark is the target area.)

Since the target dams have not experienced a failure, we choose a historical tailing dam failure event as a reference. The dam failure scenario is defined based on the Aznalcóllar tailing dam failure in Spain. Similar to the Jinding mine, the Los Frailes mine, where Aznalcóllar tailing dam is located, is also a large-scale Pb-Zn mine (which produces Cu and Ag as well), so the heavy metal contents of the tailings are expected to be similar. Also, the scale of the two mining areas is similar. The designed capacity of Los Frailes mine is 173 kt (Pb+Zn), and the designed production of Jinding Pb/Zn mine is approximately 220 kt (Pb+Zn). Therefore, the amount of tailing is also expected to be similar. Aznalcóllar tailing dam failure is a major dam failure event that was proved to have caused severe consequences. Considering the mining scale and the metal types, the release potential of the Aznalcóllar mine is expected to be similar to the Jinding mine. The Aznalcóllar tailing dam was designed to hold 32.5 million m^3 of tailings, and the maximum height was 32 m. The Aznalcóllar tailing dam failure happened on Apr. 25th, 1998. At the time of failure, the tailing pond contained approximately 15 million m^3 of tailing. Approximately 1.3 million m^3 of fine pyrite tailings and 5.5 million m^3 of wastewater are released into the surrounding environment. The tailings make their way to about 40 km downstream. The tailings released in the accident are concentrated with HMs; the mass content is 0.6%~0.7% As, 0.8%~1.0% Pb, and 0.003%~0.004% Cd. (Mcdermott & Sibley, 2000).

4.2.2.1 Accidental release

Since the two tailing dams in the Jinding mining area are located close to each other, we assume a single-point release. The amount and magnitude of release in this scenario reference the Aznalcóllar tailing dam failure. In this scenario, we assume that about 80% of the 1.3 million m^3 released tailings settled within 12 km of the mine site on approximately 800 hectares of riverbanks and agricultural land (Mcdermott & Sibley, 2000).

4.2.2.2 Physical and chemical properties of the tailings

For a more straightforward calculation, the target area is assumed flat, and the tailings are evenly distributed in the flooded area. Due to the flooding, the soil is expected to mix with the tailings. In Barcelos et al. (2020)'s research, the author assumed the tailings are mixed 1:1 with the soil due to flooding (**Fig 4-4**). In this research, we also apply this assumption. The thickness of the mixture of tailing and soil (H) is calculated as follows:

$$\mathrm{H} = (80\% \times 1.3 \times 10^6\mathrm{m}^3/(8 \times 10^6\mathrm{m}^2)) \times 2 = 0.26(\mathrm{m}) \quad (4\text{-}1)$$

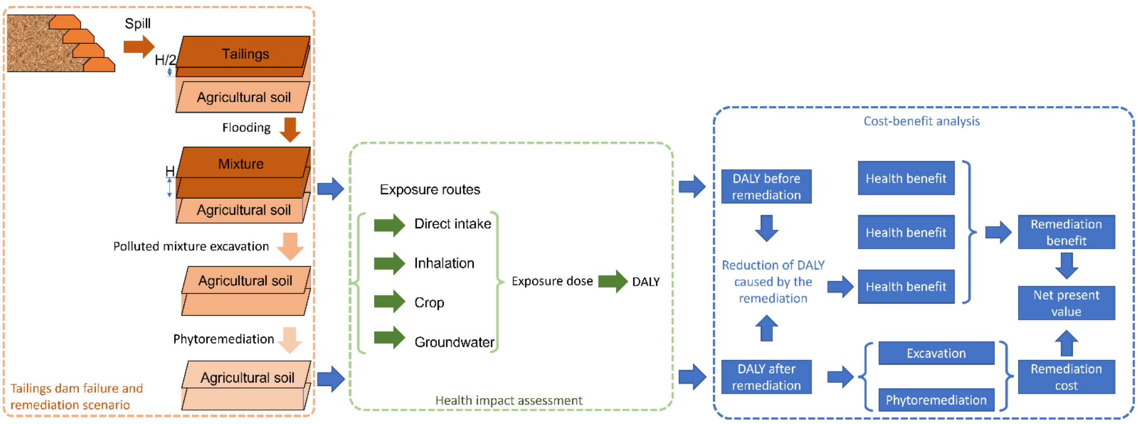

Fig 4-4. Scenario definition and analysis procedure of the present research. The tailings are released by a hypothetical tailing dam failure and mixed with surface soil. Remediation starts immediately after the failure. HM exposure and DALY level are estimated both before and after the remediation. The reduction of DALY is applied for the cost-benefit analysis for the remediation.

Table 4-1. Parameters applied for scenario definition.

Parameters	Description	Values	Reference
$c_{T,1} \pm \sigma c_{T,1}$	As concentration in tailings (mg/kg)	6500 ± 500	a
$c_{T,2} \pm \sigma c_{T,2}$	Cd concentration in tailings (mg/kg)	35 ± 5	a
$c_{T,3} \pm \sigma c_{T,3}$	Pb concentration in tailings (mg/kg)	9000 ± 1000	a
$c_{0,1} \pm \sigma c_{0,1}$	Surface soil As concentration before dam failure (mg/kg)	19.62 ± 7.56	b
$c_{0,2} \pm \sigma c_{0,2}$	Surface soil Cd concentration before dam failure (mg/kg)	5.03 ± 2.90	b
$c_{0,3} \pm \sigma c_{0,3}$	Surface soil Pb concentration before dam failure (mg/kg)	138.97 ± 58.03	b
$c_{1,1} \pm \sigma c_{1,1}$	As concentration of soil and tailings mixture right after dam failure (mg/kg)	3259.81 ± 253.78	a,b
$c_{1,2} \pm \sigma c_{1,2}$	Cd concentration of soil and tailings mixture right after dam failure (mg/kg)	20.02 ± 3.95	a,b
$c_{1,3} \pm \sigma c_{1,3}$	Pb concentration of soil and tailings mixture right after dam failure (mg/kg)	4569.49 ± 529.02	a,b
H	Thickness of the mixture of tailings and soil right after dam failure (m)	0.26	a
$c_{2,1} \pm \sigma c_{2,1}$	Surface soil As concentration after remediation (mg/kg)	19.62 ± 7.56	b
$c_{2,2} \pm \sigma c_{2,2}$	Surface soil Cd concentration after remediation (mg/kg)	0.30 ± 0.17	b
$c_{2,3} \pm \sigma c_{2,3}$	Surface soil Pb concentration after remediation (mg/kg)	80.00 ± 33.41	b
$c_{BG,1}$	Background As concentration of Yunnan soil (mg/kg)	18.4	c
$c_{BG,2}$	Background Cd concentration of Yunnan soil (mg/kg)	0.218	c
$c_{BG,3}$	Background Pb concentration of Yunnan soil (mg/kg)	40.60	c
$c_{ST,1}$	Chinese standard of As concentration in agricultural soil (mg/kg)	40.00	b
$c_{ST,2}$	Chinese standard of Cd concentration in agricultural soil (mg/kg)	0.30	b
$c_{ST,3}$	Chinese standard of Pb concentration in agricultural soil (mg/kg)	80.00	b
BM_{PV}	BM of Pteris vittata. (t/year)	0.0036	d
BM_{SA}	BM of Sedum alfredii H. (t/year)	0.00072	e
D	Depth of the remediated soil for phytoremediation (m)	0.2	b
ρ	Soil density (t/m^3)	1.3	f
$c_{P,3,PV}$	cP of Pb in Pteris vittata. (mg/kg)	1373	g
$c_{P,2,SA}$	cP of Cd in Sedum alfredii H. (mg/kg)	680	g
$c_{P,3,SA}$	cP of Pb in Sedum alfredii H. (mg/kg)	708	g

Reference: a: (Mcdermott & Sibley, 2000), b: (Jin, 2017a); c: (Cheng et al., 2018); d: (Chen, 2002); e: (Yang et al., 2002); f: (Tuyishimire et al., 2022); g: (Wu et al., 2007b)

We define soil HM concentration $c_{i,j}$ (i for different periods: i=0 for before dam failure; i=1 for after dam failure and before remediation; i=2 for after remediation; and j for HMs: j=1 for As, j=2 for Cd and j=3 for Pb) and the standard deviation ($\sigma c_{i,j}$) of $c_{i,j}$. For the HM concentration, only the range of HM content is available, so we assume the mid-value as average and the variation from the mid-value as standard deviation. For example, the As content in tailing is 0.6%~0.7%, and the assumed As concentration is 0.65±0.05%, which equals 6500±500 mg/kg. Since the concentration of tailings in the Jinding mining area is not available, the concentration and standard deviation in the mixture of tailings are estimated as the average of the content of the tailings (Mcdermott & Sibley, 2000) and the mixed soil (Jin, 2017a) ($c_{T,j}$ and $\sigma c_{T,j}$ are the heavy metal concentration and standard deviation of tailing).

$$c_{1,j} = (c_{T,j} + c_{0,j})/2;\ \sigma c_{1,j} = (\sigma c_{T,j} + \sigma c_{0,j})/2 \quad (4\text{-}2)$$

4.2.2.3 Clean-up process after the accidental release

The tailing dam failure caused massive release of tailing and heavy metals, so remediation process is required. The heavy metal concentration in soil (mixture of soil and tailing) will be extremely high after the accidental release, so some remediation technologies may not be effective. In this research, we apply excavation to the target site right after the accident. In the Aznalcóllar dam failure event, the clean-up operation started immediately after the spill and continued until January 1999. The clean-up operation mainly involved the removal of spilt tailings and 4.6 million m^3 of polluted soil (Gallart et al., 1999; Hudson-Edwards et al., 2003). Few further remediation actions are recorded. The clean-up action recovered the HM level to the pre-spill level, but still higher than the pre-mining level (Del Río et al., 2004; Morillo et al., 2005). In the present research scenario, excavation of the mixture of soil and tailing starts right after the dam failure.

Since the excavation can recover the HM concentration in surface soil back to pre-spill level, but not pre-mining level, considering the mining history in the target area, an additional period of remediation process is applied in this research. Considering heavy metal removal rate and economic feasibility, phytoremediation, which is identified as the potentially best technology, is applied to the target soil after excavation. In this topic, because the accidental scenario requires urgent remediation, the remediation target is defined as the Chinese standard of agricultural soil, which is relatively higher than the background value than the 2nd topic, and thus will require relatively less time.

Comparing to the Chinese standard of agricultural soil, Cd and Pb content exceeded this standard, but As content is below the standard level. Therefore, phytoremediation only targets Cd and Pb, and As is not reduced. In the 2nd remediation period, we assume Cd and Pb concentration reduces to the Chinese standard of agricultural soil, and As concentration remains constant. Wan et al. (2016) conducted a case study of phytoremediation with two species of hyperextractor plants: Sedum alfredii H. and Pteris vittata. The study showed the ability to reduce Cd and Pb in agricultural soil. The remediation duration of phytoremediation is estimated by the extraction ability of the two plant species and the amount of HMs to be extracted. Assuming the extraction rate of the hyper extraction plant is constant, as reported by Chandra et al. (2017):

$$RD_j = \lceil (c_{2,j} - c_{1,j}) \times D \times \rho / (c_P \times BM) \rceil \quad (4\text{-}3)$$

RD_j: remediation duration of the jth HM (j=2 for Cd and j=3 for Pb). $c_{2,j}$: pre-spill level of jth HM in the agricultural soil (mg/kg); $c_{1,j}$: soil concentration of jth HM after remediation; c_P: concentration of HM in plant tissue (mg/kg); BM: annual plant biomass production (t/m^2); D: depth of the remediated HM (m); ρ: soil bulk density (t/m^3).

Cd is extracted by Sedum alfredii H.

$$RD_2 = \lceil (c_{2,2} - c_{1,2}) \times D \times \rho / (c_{P,2,SA} \times BM_{SA}) \rceil = \lceil 2.41 \rceil = 3 \text{ years} \tag{4-4}$$

Pb is extracted by both Pteris vittata. and Sedum alfredii H.

$$RD_3 = \lceil (c_{2,3} - c_{1,3}) \times D \times \rho / (c_{P,3,SA} \times BM_{SA} + c_{P,3,PV} \times BM_{PV}) \rceil = \lceil 3.73 \rceil = 4 \text{ years} \tag{4-5}$$

Therefore, the final remediation period is the longest period of both HMs, defined as 4 years. In Wan et al. (2016)'s study, the removal efficiency of Cd and Pb for 2 years is 85.8% and 30.4%, respectively. In the present research, the removal efficiency is 94.04% in 3 years for Cd and 57.57% in 4 years for Pb. Wu et al. (2007a) showed that Sedum alfredii H. and Pteris vittata could endure a concentration of over 1000 mg/kg Cd and Pb in the soil, which is far beyond the soil HM level in both the present research and Wan et al. (2016)'s study. Therefore, the two species are considered proper for remediation in the present research.

In summary, the remediation strategy for this hypothetical dam failure event consists of 2 periods. The first period is removing the tailing and soil mixture (26 cm depth) and reducing surface soil concentration to pre-spill level. The second period is phytoremediation, reducing the soil concentration (20 cm depth) to the Chinese standard of agricultural soil (Jin, 2017a) (**Table 4-1**). The excavated soil and polluted plant biomass produced during the remediation process are transferred to permitted off-sites and disposal facilities for further monitoring or processing to prevent pollutant release to the environment.

4.2.2.4 Temporal change and mass flow of the scenario

The timeline and mass flow for this scenario are shown in **Fig 4-5**. As a result of the remediation, 99.40% of As, 98.50% of Cd, and 98.24% of Pb are reduced from the flooded agricultural site. All the tailings are supposed to be removed, and additional HMs in the soil that exceeds the Chinese standard of agricultural soil are also extracted from the site.

The mass flow is calculated according to the definition of the scenario. The accidental release is the HM that enters the area with the tailings released. The long-term release is the HM released by mining activities and is accumulated in the target area, which explains the reason for a higher concentration than the background value in the soil before the tailing dam failure. The stock of HM in the soil is the background content of HM from the natural source. Excavation and phytoremediation remove the HM in the polluted area after the tailing dam failure happens. The deposit of HM in the soil is the HM that is not removed and remains in the soil but is at a safer level.

$$Flow_{\text{accidental release}} = \rho \times A \times (H/2) \times c_T \tag{4-6}$$

$$\text{Flow}_{\text{long-term release}} = \rho \times A \times (D + H/2) \times (c_0 - c_{BG}) \tag{4-7}$$

$$\text{Stock}_{\text{soil}} = \rho \times A \times (D + H/2) \times c_{BG} \tag{4-8}$$

$$\text{Flow}_{\text{excavation}} = \rho \times A \times H \times c_1 \tag{4-9}$$

$$\text{Flow}_{\text{phytoremediation}} = \rho \times A \times D \times (c_0 - c_{ST}) \tag{4-10}$$

$$\text{Deposit}_{\text{soil}} = \text{Flow}_{\text{accidental release}} + \text{Flow}_{\text{long-term release}} - \text{Flow}_{\text{excavation}} + \text{Flow}_{\text{phytoremediation}} \tag{4-11}$$

ρ is the density of soil; A is the area of target agricultural land; H is the thickness of the mixture of tailings and flooded soil; D is the depth of soil for phytoremediation; c_T is the HM concentration in tailings; c_0 is the original HM concentration in soil; c1 is the HM concentration of soil and tailings mixture; c_{BG} is background value of Yunnan soil; c_{ST} is the Chinese standard of agricultural soil. The value and reference of these parameters are shown in **Table 4-1**.

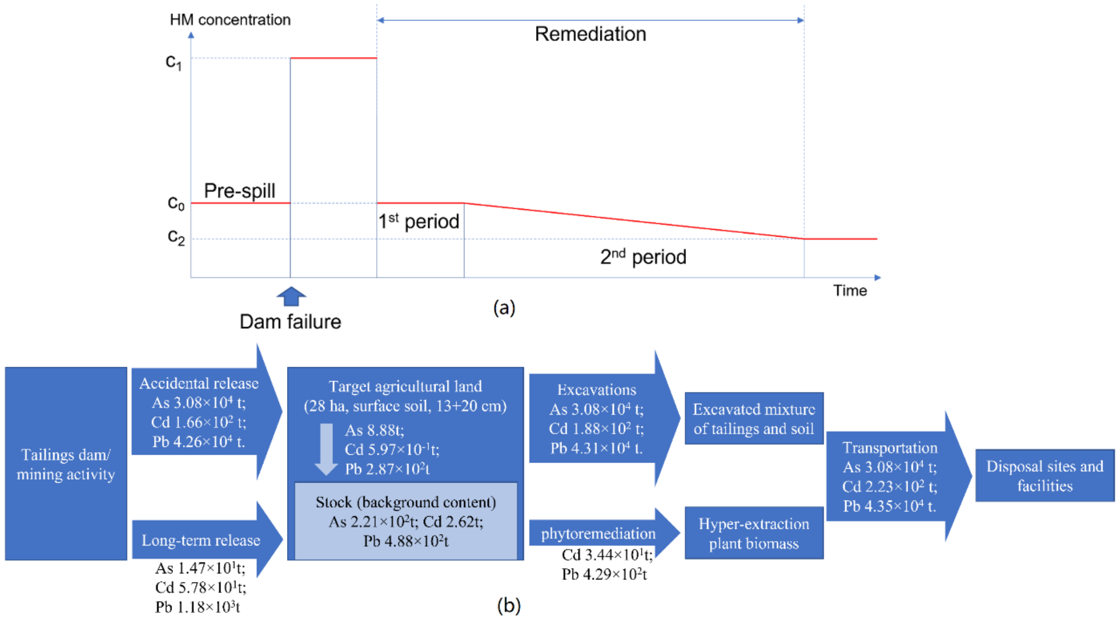

Fig 4-5. The timeline and the mass flow of the scenario in the present research. (a) Timeline of the variation in surface soil HM concentration; (b) the mass flow of the three HMs in this scenario.

4.2.3 HM exposure estimation

In this research, we consider four main exposure routes: direct soil ingestion; soil particle inhalation; crop consumption, and groundwater consumption (**Fig 4-4**). All these four exposure routes are affected by the HM concentration of surface soil. The exposure dose via direct soil ingestion and soil particle inhalation is directly affected by the HM concentration. The soil HM concentration will influence the concentration in agricultural products and groundwater, further influencing the other two exposure routes. The average daily dose is estimated as $ADD_{i,j,k}$, i for different periods: i=0 for before dam failure; i=1 for after dam failure and before remediation; i=2 for after remediation; and j for HMs: j=1 for As, j=2 for Cd and j=3 for Pb and k is for each exposure route (k=1 for soil ingestion, k=2 for soil particle inhalation, k=3 for groundwater consumption and k=4 for crop consumption). Due to the massive release of tailing and the mixing of tailing and surface soil, the target agricultural may temporarily lose its ability of growing crops until the remediation is carried out. Also the groundwater may also be severely influenced by the tailing water released. So the two exposure routes, crop consumption, and groundwater consumption are excluded from the exposure assessment before remediation.

For soil ingestion; soil particle inhalation, and groundwater consumption, the heavy metal exposure dose is calculated by the GERAS-1 (Geo-Environmental Risk Assessment System, tier-1) model simulation (Kawabe et al., 2003). GERAS-1 is an onsite screening exposure assessment model developed based on another risk assessment model, CSOIL (Brand et al., 2007). The GERAS-1 model can assess human exposure to HMs and organic matter at the site where the pollutants are directly released. GERAS-1 estimates the pollutant concentration based on the fugacity of pollutants between soil solid particles, soil pore-water, and soil pore-air. The concentration in the exposure media (atmosphere, crops, groundwater) is estimated by the transfer process from soil to those exposure media. Finally, the exposure dose is estimated with the concentration in the exposure media according to the human exposure route, activity pattern, and living environment. Detailed information (equations) on the models can be found in the supplemental material and (Brand et al., 2007; Kawabe et al., 2003).

Table 4-2. Parameters applied for GERAS-1 simulation (because no specific temporal definition is applied for the scenario, default normal temperature is applied for temperature).

Soil parameters	Values	Exposure parameters	Unit	
			Adults	Children
Air fraction (%)[a]	0.21	Time outside (weekday) (h/d)[d]	0	2
Water fraction (%)[a,b]	0.2	Time inside (weekday) (h/d)[d]	24	22
Solid fraction (%)[a,b]	0.59	Time outside (weekend) (h/d)[d]	4	5
pH (-)[b]	6.2	Time inside (weekend) (h/d)[d]	20	19
Temperature (K)	300	Years living (y)[d]	64	6
Organic carbon (%)[c]	0.017	Body weight (kg)[e]	62	20
Clay content (%)[b]	0.18	Soil direct ingestion (g/d)[e]	50	85
Density (g/cm^3)[a]	1.33	Groundwater consumption (L/d)[d]	2	1
		Inhalation rate (m^3/d)[e]	16.1	8.8

Reference: a: (Tuyishimire et al., 2022); b: (Tang et al., 2014); c: (He et al., 2009); d: default settings in GERAS-1 (Brand et al., 2007; Kawabe et al., 2003); e: (Duan et al., 2015; Zhao et al., 2016).

The main crop in the target area is wheat (Ministry of Civil Affairs of PRC, 2016, **supplemental material S3**). In GERAS-1, the intake of crops is estimated separately as leaf crops and root crops. However, either of these two categories is suitable for wheat. Therefore, the exposure dose by wheat intake was calculated as follows (US EPA, 2002):

$$\mathrm{ADD_{i,j,4}} = \mathrm{c_{i,j}} \times 1000 \times \mathrm{BCF_k} \times \mathrm{IngR} \times \mathrm{EF} \times \mathrm{ED}/(\mathrm{BW} \times \mathrm{AT}) \tag{4-12}$$

Where $ADD_{i,j,4}$ is the average daily dose of crop intake (μg/kg-BW/day), $c_{i,j}$ is the HM concentration in soil (mg/kg), IngR is the wheat ingestion rate (g/day), EF is the exposure frequency (day/year), ED is the exposure duration (year), BW is body weight (kg), and AT is the average exposure time (day). The BCF_k is the ratio of the kth HM concentration in plants and soil (Liu et al., 2007). An average bio-concentration factor of wheat under different soil HM concentrations reported by (Wang et al., 2017) was applied in this research (**Table S2**).

As a result of the exposure assessment, all exposure dose of the four exposure routes is summed as a total average daily dose ($ADD_{i,j}$).

$$\mathrm{ADD_{i,j}} = \textstyle\sum_{k=1}^{4} \mathrm{ADD_{i,j,k}} \tag{4-13}$$

The standard deviation of the average daily dose ($\sigma ADD_{i,j,k}$) is required for DALY calculation. In this research, we assume that the daily dose and HM concentration are changed proportionally. Therefore, we have ($\sigma c_{i,j,k}$ is the standard deviation of $c_{i,j,k}$):

$$\sigma\mathrm{ADD_{i,j}}/\sigma\mathrm{c_{i,j}} = \mathrm{ADD_{i,j}}/\mathrm{c_{i,j}} \tag{4-14}$$

4.2.4 Health impact

DALY is a health impact indicator that quantifies health risk by combining the dose-response relationship with the impact of diseases or disabilities on the human body. Compared to traditional indicators applied in CBA for soil remediation (Huysegoms et al., 2018; Stewart & Purucker, 2011), DALY is expected to create a stronger connection between exposure dose and end-point effect. Thus presenting a more comprehensive health impact level for cost-benefit assessment. Additionally, DALY can be monetized by an average financial burden and can be compared with remediation cost in the cost-benefit analysis. Therefore, we choose DALY as the indicator for health impact quantification. The methodology for quantifying DALY based on exposure dose and exposure duration was developed by Murray (1994). This research defines DALY as years life lost (YLL) and years lost to disability (YLD).

$$\mathrm{DALY} = \mathrm{YLL} + \mathrm{YLD} \tag{4-15}$$

The exposure is not expected to cause a fatal effect. Therefore, in this research, YLL is assumed as zero. The yearly $\mathrm{DALY}_{i,j}$ is calculated as:

$$\mathrm{DALY}_{i,j} = \text{Disease Probability } \left(\mathrm{DP}_{i,j}\right) \times \text{Disease Weight } (\mathrm{DW}_j) \tag{4-16}$$

Disease probability is defined by exposure dose and dose-response relationship:

$$\mathrm{DP}_{i,j} = \int \mathrm{f}_{i,j}(\mathrm{x})\phi_j(\mathrm{x})\, \mathrm{dx} \tag{4-17}$$

x: exposure dose (μg/kg-BW/day); $f_{i,j}(x)$: possibility distribution of exposure dose of the jth HM in the ith period; $\phi_j(x)$: possibility distribution of dose-response relationship of the jth HM. This research assumes that $f_{i,j}(x)$, and $\phi_j(x)$ follow a lognormal distribution.

$$\mathrm{f}_{i,j}(\mathrm{x}) = \left(1/\mathrm{x}\sigma_{i,j}\sqrt{2\pi}\right)\exp\left(-\left(\ln\mathrm{x} - \mu_{i,j}\right)^2/2\sigma_{i,j}^2\right) \tag{4-18}$$

$$\phi_j(\mathrm{x}) = \left(1/\mathrm{x}\sigma_j\sqrt{2\pi}\right)\exp\left(-\left(\ln\mathrm{x} - \mu_j\right)^2/2\sigma_j^2\right) \tag{4-19}$$

For $f_{i,j}(x)$, $\sigma_{i,j}$ and $\mu_{i,j}$ could be derived from the mean value ($ADD_{i,j}$) and standard deviation ($\sigma ADD_{i,j}$) of exposure dose (**Table 4.3**).

$$\mu_{i,j} = \mathrm{ADD}_{i,j}^2/\left(\sqrt{\mathrm{ADD}_{i,j}^2 + \sigma\mathrm{ADD}_{i,j}^2}\right) \tag{4-20}$$

$$\sigma_{i,j} = \exp\left(\sqrt{\ln\left(\left(\sigma\mathrm{ADD}_{i,j}/\mathrm{ADD}_{i,j}\right)^2 + 1\right)}\right) \tag{4-21}$$

For $\phi_j(x)$, σ_j and μ_j are defined by:

$$\mu_j = \ln(\mathrm{GM}_{res}) \tag{4-22}$$

$$\sigma_j = \ln(\mathrm{GSD}_{res}) \tag{4-23}$$

GM_{res}: the geometrical mean of the dose-response relationship; GSD_{res}: the geometrical standard deviation of the dose-response relationship. For all HMs, GSD_{res} are defined by the difference in metabolizing rate (GSD_m) and sensitivity (GSD_s) in the human body, 1.4 and 2.7 (μg/kg-BW/day), respectively (Masuyama, 1977; Nordberg & Strangert., 1976).

$$(\ln(\mathrm{GSD_{res}}))^2 = (\ln(\mathrm{GSD_m}))^2 + (\ln(\mathrm{GSD_s}))^2 \qquad (4\text{-}24)$$

Table 4-3. Parameters applied for DALY calculation.

Parameters	Description	Values	Reference
$ADD_{1,1} \pm \sigma ADD_{1,1}$	Average daily dose of As after dam failure (μg/kg-BW/d)	$1.07 \pm 0.08 \times 10^{-2}$	a
$ADD_{1,2} \pm \sigma ADD_{1,2}$	Average daily dose of Cd after dam failure (μg/kg-BW/d)	$1.76 \pm 0.35 \times 10^{-4}$	a
$ADD_{1,3} \pm \sigma ADD_{1,3}$	Average daily dose of Pb after dam failure (μg/kg-BW/d)	$5.68 \pm 0.68 \times 10^{-3}$	a
$ADD_{2,1} \pm \sigma ADD_{2,1}$	Average daily dose of As after remediation (μg/kg-BW/d)	$6.43 \pm 2.48 \times 10^{-5}$	a
$ADD_{2,2} \pm \sigma ADD_{2,2}$	Average daily dose of Cd after remediation (μg/kg-BW/d)	$4.41 \pm 2.55 \times 10^{-5}$	a
$ADD_{2,3} \pm \sigma ADD_{2,3}$	Average daily dose of Pb after remediation (μg/kg-BW/d)	$1.73 \pm 0.72 \times 10^{-4}$	a
$NOAEL_1$	NOAEL for As (μg/kg-BW/day)	0.80	b
$NOAEL_2$	NOAEL for Cd (μg/kg-BW/day)	10.00	c
$NOAEL_3$	NOAEL for Pb (μg/kg-BW/day)	8.96	d
DW_1	Disease weight applied for As	0.056	e
DW_2	Disease weight applied for Cd	0.098	e
DW_3	Disease weight applied for Pb	0.090	e

Reference: a: obtained from GERAS-1 simulation; b: (U.S. EPA, 1988); c: (U.S. EPA, 1987); d: (Kobayashi, 2006); e: (Lopez et al., 2006).

Nakanishi et al. (2003) et al. reported a definition of GMres of mercury by assuming that the no observable adverse effect level ($NOAEL_j$) could induce a 0.5% possibility of disease. In the present research, this assumption is applied to the target HMs:

$$0.005 \fallingdotseq \phi_j(\mathrm{GM_{res}}/\mathrm{GSD_{res}^{2.58}}) = \phi_j(\mathrm{NOEAL_j}) \qquad (4\text{-}25)$$

$$\mathrm{GM_{res}} = \mathrm{NOEAL_j} \times \mathrm{GSD_{res}^{2.58}} \qquad (4\text{-}26)$$

NOAEL applied in this research is 0.8 μg/kg-BW/day for As (U.S. EPA, 1988) and 10 μg/kg-BW/day for Cd (U.S. EPA, 1987). NOAEL for Pb ($NOAEL_3$) is calculated from the NOAEL blood Pb level (50 μg/L). For adults (Norihiro Kobayashi, 2006):

$$\text{NOAEL blood Pb level} = 0.09 \times \mathrm{NOAEL_3} \times \mathrm{BW} \qquad (4\text{-}27)$$

The NOAEL intake level of Pb ($NOAEL_3$) is 8.96 μg/kg-BW/day. The DW_j of 3 target elements is (Lopez et al., 2006): DW_1 (Hyperpigmentation caused by As exposure) is 0.056; DW_2 (End-stage renal disease caused by Cd exposure) is 0.098; DW_3 (mild mental retardation caused by Pb exposure) is 0.090.

4.2.5 Cost/benefit analysis

In this research, costs and benefits are estimated separately for each part, and the net present value (NPV) is calculated as the total benefit with the total cost extracted.

$$\mathrm{NPV} = \mathrm{B_{total}} - \mathrm{C_{total}} \tag{4-28}$$

The cost is calculated from the unit cost of soil excavation and phytoremediation. For soil excavation:

$$\mathrm{C_{se}} = \mathrm{A} \times \mathrm{H} \times \rho \times \mathrm{UC_{se}} \tag{4-29}$$

C_{se}: cost of soil excavation; A: area of remediated land; h: the thickness of the tailing/soil mixture layer; ρ: soil density; UC_{se}: the unit cost of soil excavation.

For phytoremediation:

$$\mathrm{C_{pr}} = (\mathrm{IC_{pr}} + \mathrm{AC_{pr}} \times \mathrm{RD}) \times \mathrm{A} \tag{4-30}$$

C_{pr}: cost of phytoremediation; IC_{pr}: the initial cost of phytoremediation; AC_{pr}: annual cost of phytoremediation; RD: remediation duration; A: area of remediated land. The total cost of remediation (C_{total}) is calculated as follows:

$$\mathrm{C_{total}} = \mathrm{C_{se}} + \mathrm{C_{pr}} \tag{4-31}$$

In this research, the benefit of remediation consists of three parts: health benefit caused by the reduction of HM exposure (B_H), the benefit of recovering the ecosystem (B_E), and economic benefit caused by the production of agricultural products (B_A) (Wan et al., 2016). These categories of benefits mainly focus on the residents' merit, which is this paper's primary concern. Therefore, the other categories, such as suspension of production and land price, mainly involving mining enterprises and local government, are not included. Among these benefits, B_E is a one-time benefit, B_H and B_A are annual benefits generated with time. Considering the accidental scenario, the time span for benefits assessment of B_H and B_A is set as 10 years, a relatively short time span.

For B_H:

$$B_H = FB \times \Delta DALY \times P \times 10 \quad (4\text{-}32)$$

FB: average financial burden of DALY; ΔDALY: reduction of DALY. P: population; The population directly affected by the target agricultural area is defined as the population in the residential area next to the target area (**Fig 4-1 b**). The population is estimated by the number of households and the average population in each household.

Table 4-4. Parameters applied for cost-benefit analysis.

Parameters	Description	Values	Reference
UC_{se}	Unit cost of soil excavation ($/t)	270	a
IC_{pr}	Initial cost of phytoremediation ($/hm^2)	34684.5	b
AC_{pr}	Annual cost of phytoremediation ($/hm^2)	20345.4	b
FB	Average financial burden of DALY ($/year of DALY)	108600	c
P	Population affected by target agricultural land (persons)	639	d
T	Time-span for the estimation of annual benefit (years)	10	e
V	Average value of ecosystem in China ($/hm^2)	959.8	f
Y	Average yield of wheat (t/ha)	1.88	g
wp	Average price for wheat ($/kg)	0.48	g

Reference: a: (FRTR, 2012); b: (Wan et al., 2016); c: (WHO, 2010); d: (Statista, 2022); e: (CRC CARE, 2018); f: (Xie et al., 2003); g: (Weicong et al., 2014).

ΔDALY is calculated by the total DALY before (i=1) and after remediation (i=2):

$$\Delta DALY = \sum_{j=1}^{3} DALY_{1,j} - \sum_{j=1}^{3} DALY_{2,j} \quad (4\text{-}33)$$

For B_E:

$$B_E = A \times V \quad (4\text{-}34)$$

V: the average value of the ecosystem in China.

For B_A, the main crop in the area is assumed to be wheat:

$$B_A = Y \times wp \times A \times 10 \quad (4\text{-}35)$$

Y: average yield of wheat; wp: average price for wheat.

The total benefit is the sum of the three categories of benefit.

$$B_{total} = B_H + B_E + B_A \tag{4-36}$$

The parameters applied for cost-benefit analysis are shown in **Table 4-4**.

4.2.6 Sensitivity analysis

In this research, a sensitivity analysis is conducted for the calculation process. The result of this research heavily depends on the definition of the hypothetical dam failure scenario. If the scenario changes, the assumptions and parameters may be influenced, and the result may vary. The main purpose of the sensitivity analysis is to discuss the robustness and the influence of parameter change on the output of the analysis procedure. In this research, a sensitivity analysis is conducted. The method of sensitivity analysis is nominal range sensitivity (Frey & Patil, 2002). The major assumptions in this research are applied to the definition of the tailing release scenario, the scale (population and area) of target land and the hypothesis of the remediation process. Considering these major assumptions, variation of input values for parameters are substituted for important indicators in the present research to discuss the influence of variation in input factors on the result of NPV (Eqn 26). Only one of the chosen parameters changes each time, while the other parameters stay constant (CRC CARE, 2018; Frey & Patil, 2002). This methodology is defined by Walker & Fox-Rushby (2001) as qualitative analysis. By changing each input parameter, the result of NPV also changes, and the variation of the new NPV value compared to the original result is defined as the sensitivity to the corresponding change in the parameter. The extent of variation is set as ±20% and ±40% (Huysegoms et al., 2019a).

$$S_{t\theta} = NPV_{t\theta}/NPV \tag{4-37}$$

$S_{t\theta}$: sensitivity of NPV to the change θ in input parameter t; $NPV_{t\theta}$: the result of NPV under change θ in input parameter t; θ: change in input parameters (±20% or 40%); t: number of the changed parameter.

Parameters 1~3 is selected according to the major assumptions in the scenario definition for tailings release. Parameter 1 is the input HM concentration of tailings; parameter 2 is the mixing ratio of tailings and soil; parameter 3 is the release amount of tailings.

Parameters 4 is selected according to the definition of the target area and population. We assume that the area of the agricultural land and the population influenced by it is proportional and is expected to change simultaneously.

Parameters 5 and 6 are selected for the cost-benefit analysis of the remediation process. Parameter

5 is the variation in the soil depth for phytoremediation in the 2nd period of the remediation process, and parameter 6 is the time span for benefits (health benefit and price of agricultural products). Additionally, not only ±20% and ±40%, time spans of 30 years (Huysegoms et al., 2019a) and 75 years (life expectancy) are also applied to discuss long-term effectiveness. NPV is estimated for the variation of each parameter, and the variation ratio in NPV of each variation compared to the original result is also calculated and discussed.

4.3. Results

4.3.1 Reduction of health impact

4.3.1.1 HM exposure dose

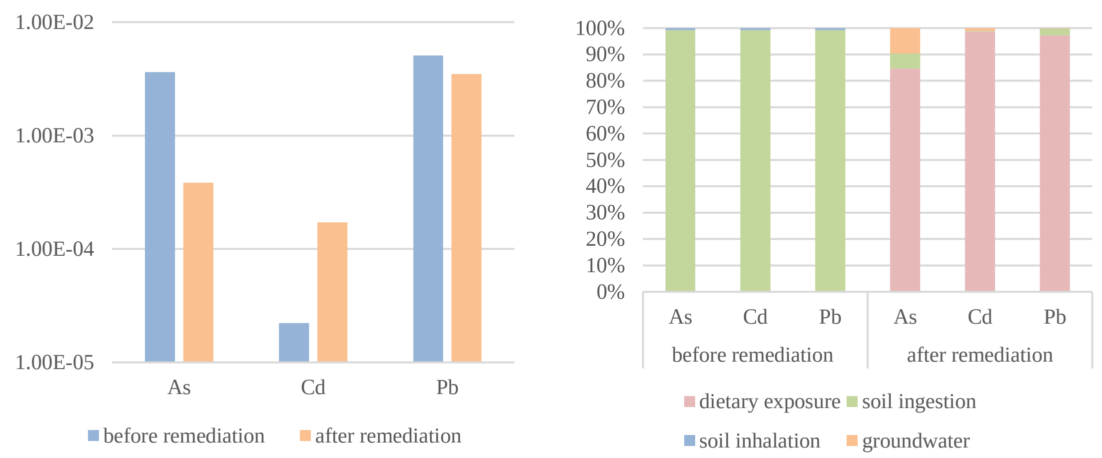

Fig 4-6. The simulation result of exposure dose (obtained by GERAS-1) a: exposure dose; b: percentage of each exposure route.

Before remediation, The exposure dose of HM is 3.62×10^{-3} (mg/kg-BW/d) for As; 2.22×10^{-5} (mg/kg-BW/d) for Cd; and 5.07×10^{-3} (mg/kg-BW/d) for Pb (**Fig 4-6**). For exposure assessment, only soil ingestion and soil inhalation are included. Exposure via soil ingestion is much higher than soil inhalation. This result indicates that it is important action for the resident to avoid contact between soil and food or oral areas, such as washing hands before eating. After the remediation is applied, the exposure dose is 3.83×10^{-4} (mg/kg-BW/d) for As; 1.72×10^{-4} (mg/kg-BW/d) for Cd; and 3.48×10^{-3} (mg/kg-BW/d) for Pb. The As and Pb exposure dose after remediation is reduced, however the Cd exposure dose is increased. This is due to the dietary intake via the consumption of crops after the remediation. Dietary exposure is the overwhelming exposure route for Cd exposure after the remediation. For As and Pb, dietary exposure is also the most important exposure route, but groundwater consumption and soil ingestion have certain contribution. The result of exposure analysis indicates that the remediation action is effective, but the exposure of Cd may remain as a problem.

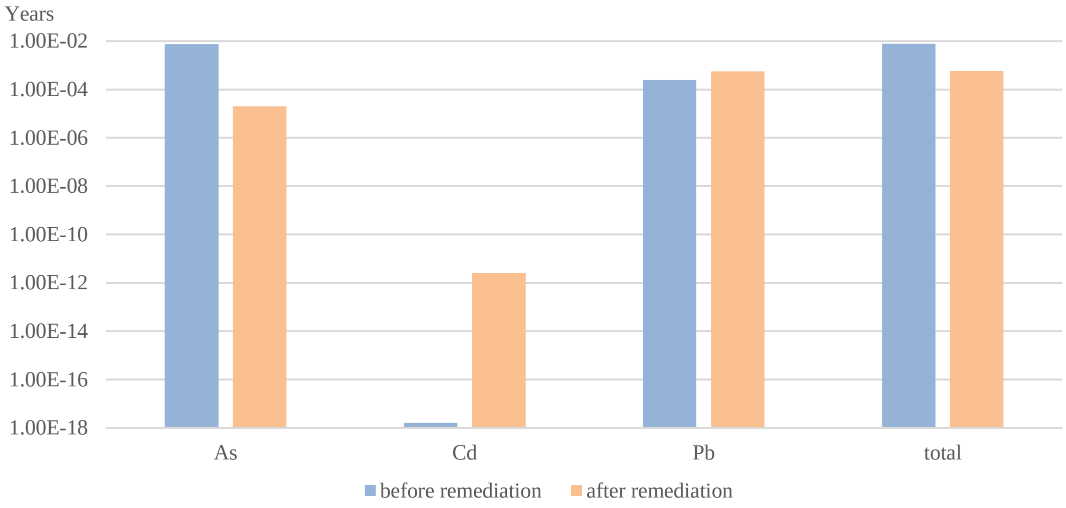

Fig 4-7. The assessment result of DALY. The DALY level decreases significantly after remediation. As and Pb contribute higher DALY than Cd.

4.3.1.2 DALY

The total DALY caused by HM exposure before remediation is 7.77×10^{-3} years per year of exposure (**Fig 4-7**). The DALY of As exposure is the most important, taking 96.83% of the total DALY. Cd is less harmful than the other two metals in this scenario and does not contribute much to the total health impact. After the remediation, the total DALY is reduced to 5.84×10^{-4} years of DALY per year of exposure. The remediation successfully reduces 92.94% of the original DALY, indicating the remediation is effective. Pb contributes the most (96.51%) to the total DALY after remediation. Compared to the research mentioned in the previous chapter that discussed the health risk of residents, the DALY level estimated in the present research reveals the actual expectation of impact on the residents' health, as well as the reduction of the impact due to the remediation process. This reduction of impact (DALY) level is essential for the cost-benefit analysis of the remediation process.

4.3.2 Cost-benefit analysis

The result is shown in **Fig 4-8**. The total cost for remediation is \$ 2.88×10^{7}. Approximately 88.72% of the remediation cost comes from soil excavation. The average cost of remediation in this paper is \$ 1.03×10^6/ha, higher than the recovery and clean-up price (CNY 7.5×10^5/ha) evaluated by Liu et al. (2020). This result is also higher than the average cost of soil excavation (\$ 7.02×10^5/ha) estimated for agricultural soil under normal scenario in chapter 3. The difference to chapter 3 may be caused by the depth of excavated polluted materials, defined by the amount of tailing released. In this paper, the thickness of excavated soil layer is higher than chapter 3. Also, in this research, remediation involves two periods. The cost of phytoremediation also contributes a part to the total cost.

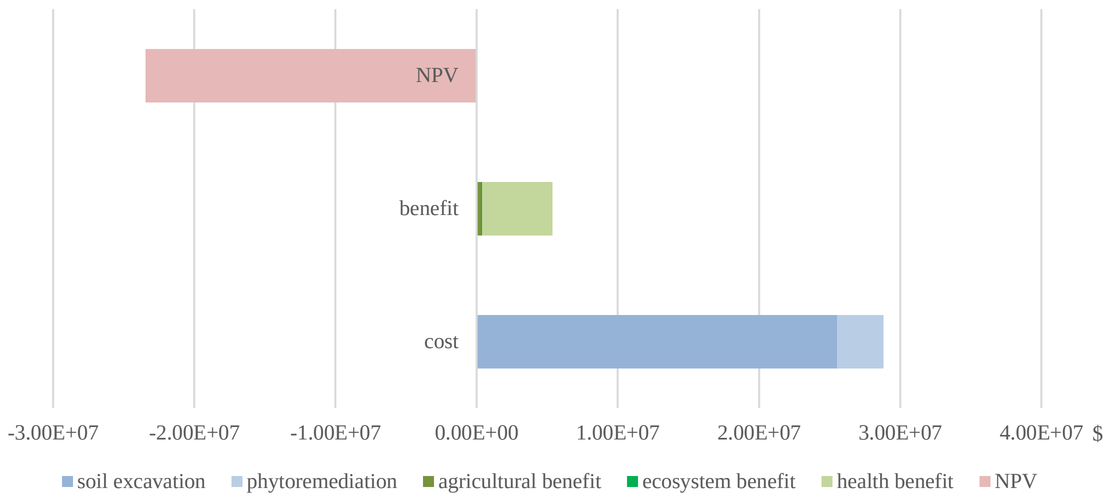

Fig 4-8. Result of cost-benefit analysis. Soil excavation takes up most of the cost, and health benefit takes up most of the benefit. The total cost exceeds the total benefit, so the net present value (NPV) is negative. The NPV bar shows the scale of the negative NPV value.

The total benefit estimated in this research is \$ 5.36×10^6, which is lower than the total cost. Health benefit is the most important benefit, taking 93.19% of the total benefit. In Wan et al. (2016)'s research, income loss prevented by remediation is also evaluated, but this category is not included in the present research due to a lack of information. However, the benefit value of income loss prevention is much less than other categories, which is not expected to strongly influence the result.

The analysis result showed a minus value of the net present value of \$ -2.34×10^7. The benefit counts for 18.59% of the total cost, and the benefit rate (NPV/total cost) is -81.41%. This result indicates that the remediation action may not be economically feasible for the target area. The negative rate is mainly due to the high cost of soil excavation. Researchers that conducted cost-benefit analyses for soil remediation showed different results. Some studies showed beneficial remediation actions for industrial areas (Lavee, 2012) or river sediments (Zheng et al., 2019). Some studies showed negative NPV values for industrial soil remediation (Huysegoms et al., 2019a). The reason for these differences is the variation in cost/benefit considerations and scenario settings.

Nonetheless, all these studies are under normal scenarios instead of accidental ones. Studies are unavailable on cost-benefit analysis of soil remediation for tailing dam failures. In the present research, we discuss the economic feasibility of soil remediation for tailing dam failures, and the result shows that remediation of agricultural soil under a tailing dam failure event may not be economically beneficial.

In this case, however, the negative NPV is reasonable. The consequence of a tailing dam failure is accidental, acute, and much more severe than long-term pollution. It is reasonable that a tailing dam failure will require a large amount of effort and budget to be repaired. Additionally, a negative NPV does not mean it is better not to conduct the remediation. Since the impact of a tailing dam failure is exceptionally high, clean-up action immediately after the failure is necessary. To avoid the economically infeasible remediation action, it will be better for the local government to put stricter regulation and supervision on the tailing dam management, preventing the failure before it happens. Additionally, since the remediation is economically infeasible, the local government should pay attention to the emergency budget and consider the fine on the mining industry under dam failure scenarios.

It is worth noticing that the benefit of health risk reduction and cash crops is time-dependent. The time span considered in this research is 10 years. If the time span increases, the benefit may exceed the cost. The time span will depend on the consideration of the local government in practice. The influence of time span is discussed in the sensitivity analysis.

4.3.3 Sensitivity analysis

Parameters selected for sensitivity analysis are selected according to the major assumptions in the scenario definition and analysis process. The variation in the tailing release scenario, the scale of the target area, and the cost-benefit analysis are discussed for their impact on the final result (NPV value). The result of the sensitivity analysis ($S_{t\theta}$) is shown in **Fig 4-9**. The y-axis shows the variation of output compared to the original result. The Numbers 1~6 on the x-axis represent the 6 parameters. Parameter 1 is the input HM concentration of tailings; parameter 2 is the mixing ratio of tailings and soil; parameter 3 is the release amount of tailings; parameter 4 is the area of the target agricultural land and the number of residents affected by the target residential area; parameter 5 is variation in the soil depth for phytoremediation; parameter 6 is the time span (only change of 20% and 40% is shown in the figure, variations of 30 years and 75 years are excluded.) Since the original NPV is a minus value, a variation rate below 100% is an increase in the NPV value. Therefore, among all the input parameters, HM concentration, Number of residents, and the time span positively correlate with the result, while the other five parameters negatively correlate with the result.

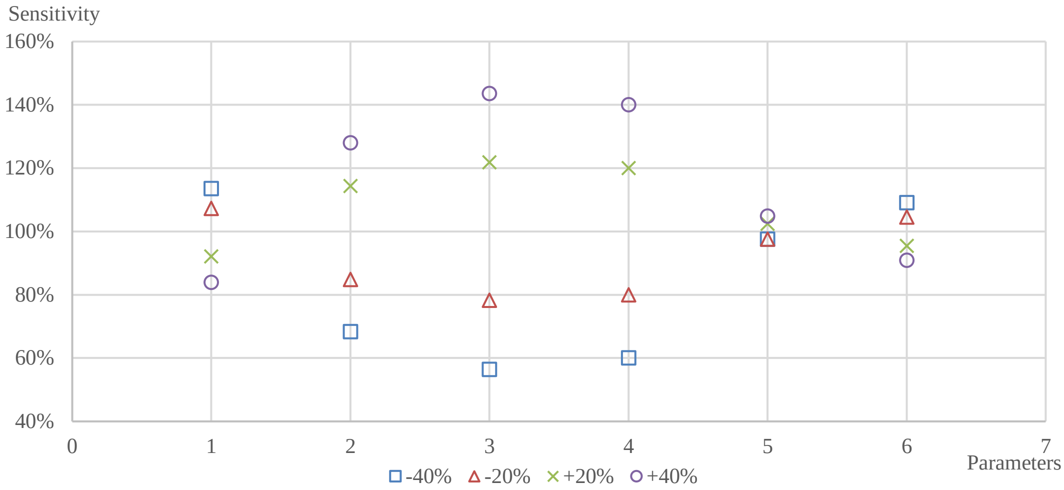

Fig 4-9. Result of sensitivity analysis (S_{t0}).

The release amount of tailings (parameter 3) has a relatively heavy influence on the result, indicating tailings release is a critical factor in a dam failure event. Soil depth for phytoremediation (parameter 7) has minimum influence on the result. Soil depth only influence the remediation duration and the cost of phytoremediation. However, the cost of phytoremediation is much less than soil excavation and only takes a small part of the total cost, so changes in parameter 7 have a limited influence on the total cost. Among all the parameters, with a 40% change in the input parameter, none of the parameter output a positive NPV value. Most changes in the output are less than 40%. This result proves the robustness of the conclusion of this research.

Considering the time span, changing the time span in the ±40% range does not produce a positive value, but the NPV value for a 75-year time span produces positive NPVs ($1.12×10^7$). Health benefit, as well as agricultural products, is a yearly benefit. Therefore, it is highly time-dependent. This research selected the time span for an accident event as a short time span (10 years). The analysis parameter may be adjusted in practice according to the requirement of the local policymaker, and since the time span can influence the result heavily, this parameter should be paid attention to.

4.4. Comparison between normal and accidental scenario

4.4.1 Health impact under normal scenario and accidental scenario

The resident will face different health impact levels under normal scenario and accidental scenario. The dam failure can greatly increase the heavy metal content in the environment, causing the health risk to increase. In this research, the health impact of soil heavy metal under normal scenario and accidental scenario is estimated and compared.

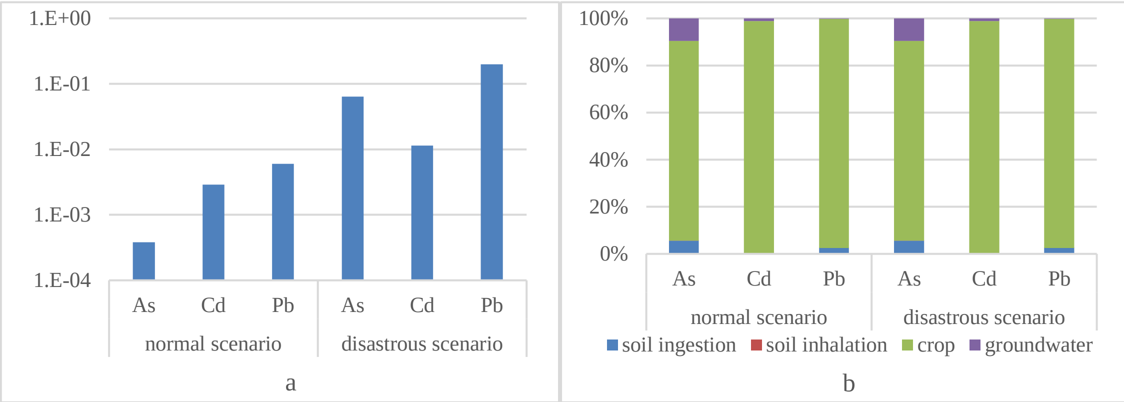

Fig 4-10. Comparison of exposure dose and contribution ratio of exposure routes of four exposure routes (a. exposure dose; b. contribution ratio of exposure routes)

Fig 4-10 shows the comparison of exposure dose under normal and accidental scenarios with all the four exposure routes included. The estimation of exposure dose via each exposure route is proportional to the soil concentration, so no difference between normal and accidental scenario is detected in the ratio of each exposure route for each heavy metal. Dietary intake from crops is the overwhelming exposure route for all the three heavy metals. Soil ingestion and groundwater have a little contribution for As and Pb exposure and soil inhalation is neglectable. The accidental exposure dose is 165, 3 and 31 times higher than normal scenario for As, Cd and Pb, respectively. The exposure dose will be significantly increased if the resident still consumes the crops and groundwater in the flooded area. Therefore, during the recovery period, the local government should provide food and water with safety insurance and avoid the exposure via the crop and groundwater route.

Fig 4-11 shows the comparison of exposure dose estimated in chapter 3 (normal scenario) and chapter 4 (accidental scenario without crops and groundwater intake). The exposure dose of Cd and Pb under accidental scenario is lower than normal scenario, especially Cd, which is 0.8% of the normal scenario. This result addressed the importance of the excluded exposure routes under normal scenario. As under accidental scenario is still higher than normal scenario after exclusion of crops and groundwater, indicating the importance of the two included exposure routes. This result corresponds to the contribution ratio in **Fig 4-10**. For accidental scenario without crops and groundwater intake, the main exposure route is soil ingestion, contributing almost all the heavy metal exposure after the accident. Therefore, if the food and drinking water is ensured by the local government, it is important for the residents to avoid ingestion of soil to protect themselves from high heavy metal exposure. For example, avoid touching the contaminated soil and washing hands before eating food.

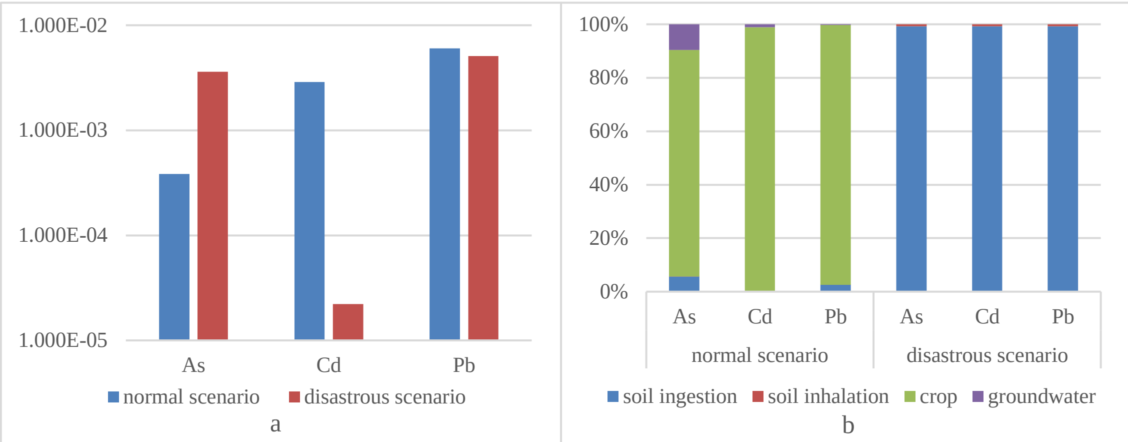

Fig 4-11. Comparison of exposure dose and contribution ratio of exposure routes with crops and groundwater intake excluded under accidental scenario (a. exposure dose; b. contribution ratio of exposure routes)

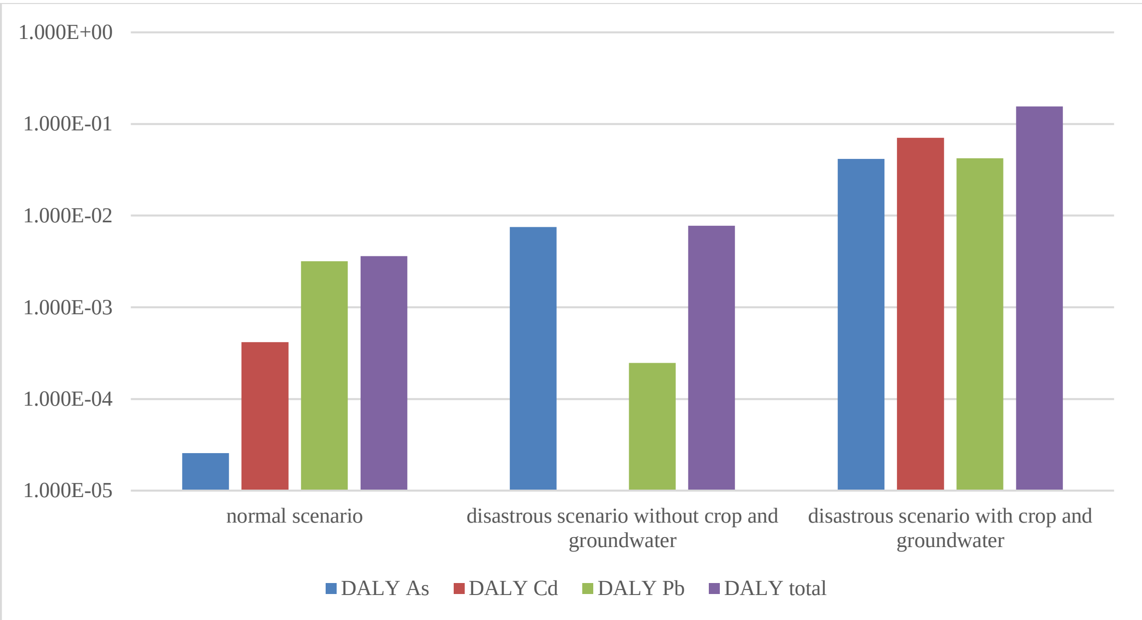

Fig 4-12. Assessment of DALY under normal and accidental scenarios

The health impact revealed by DALY (**Fig 4-12**) shows that including the crops and groundwater intake will cause a significant high DALY after tailing dam (41.8 times higher than normal scenario). Even with the crops and groundwater intake excluded, and the exposure dose of Cd and Pb lower than normal scenario, the DALY is still 1.15 times higher than normal scenario. The main DALY comes from Pb exposure under normal scenario, but As become the critical heavy metal under accidental scenario (if crops and groundwater are included, the critical heavy metal is Cd). Therefore, the prioritized heavy metal of remediation under normal and accidental scenarios are different. For example, if soil washing is applied under normal scenario, leachate with higher removability of Pb should be selected, leachate with better removability of As will be more effective under accidental scenario.

4.4.2 Remediation alternatives

In Chapter 3, four alternative remediation technologies are discussed and compared for removal efficiency and economic feasibility. As a result, phytoremediation is identified as a suitable alternative for relatively low level heavy metal contamination removal. However, for high level heavy metal contamination scenario in Chapter 4, the requirement of removal ability is highly questioned for remediation strategy. Removal efficiency of normal technologies such as soil washing and solidification may be less than enough for recovering the function of agricultural soil. Therefore, for accidental scenario, soil replacement is selected because the ability of completely removal of excessive heavy metal. The soil concentration after remediation will disregard the contamination extent and is defined by the fresh soil refilled. Different to Chapter 3, because the remediation is urgent, the refilled soil is taken from the unaffected agricultural soil in the same area to avoid the time consumption of long-term transportation. The soil concentration is also assumed to be reduced to the original level of local agricultural soil

In Chapter 3, we already discovered that the original soil concentration will place certain health threat on the residents. Therefore, after the soil excavation, an additional removal process is applied. According to the analysis in Chapter 3, phytoremediation is the potentially best remediation alternative for the agricultural soil in target area. Therefore, phytoremediation is applied to the 2nd step of remediation process. Because phytoremediation is relatively time consuming, due to the requirement of urgent remediation, the remediation target is set as a relatively higher heavy metal level (the Chinese soil standard instead of Chinese background level). The phytoremediation takes 4 years, which is much shorter than the 8 year remediation period in Chapter 3.

By comparing the remediation strategy under normal and accidental scenarios, we can conclude that under different scenarios, the remediation target and criteria for remediation alternative selection is also different. This result again addresses the importance of selecting proper remediation technology in practice, and the importance of cost-effectiveness information that can provide reference for the decision making.

4.4.3 Economic feasibility of remediation

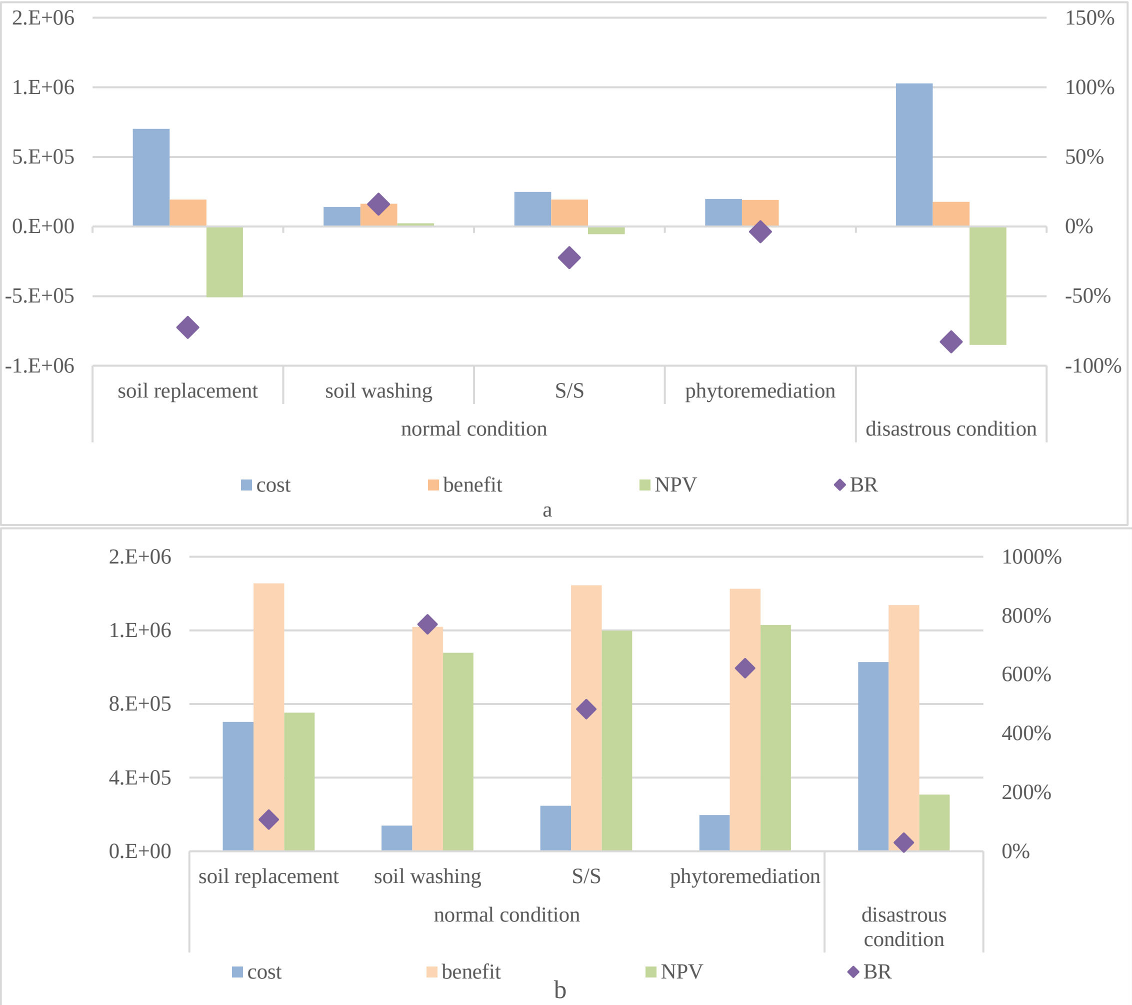

Fig 4-13. Comparison of Economic feasibility under normal and accidental scenario (a.10 year benefit; b. 75 year benefit)

In Chapter 3 and Chapter 4, the time span for benefit estimation is different. In Chapter 3, a 75-year (lifetime) impact is estimated based on long-term pollution and chronic health impact. In Chapter 4, we applied a relatively short 10-year time span for accidental heavy metal release scenario and acute exposure. The influence of time span on the analysis result is discussed in the sensitivity analysis in the two chapters respectively, but the cost and benefit in normal and disastrous scenario is not compared yet.

The 10-year and 75-year cost-benefit analysis of average area under normal and accidental scenarios is shown in **Fig 4-13**. Because the scale of study area is different between chapter 3 and 4, the economic feasibility is analyzed based on average values. For 10-year benefit, only soil washing under normal scenario exceeded the cost and yielded a positive NPV value, all the other scenarios including accidental scenario yielded negative NPV value. But for 75-year benefit, all the five scenarios yielded positive NPV value. Health benefit is a highly time dependent type of benefit. Therefore changing the time period of the assessment greatly influences the result. This indicates the health benefit may be underestimated in remediation projects that focus on short-term benefit and local government should pay attention to the residents' health when discussing the economic feasibility of remediation.

Remediation under accidental scenario has a relatively low economic efficiency. The main reason is the exclusion of the critical exposure route, dietary intake. Lower exposure level before remediation yielded a lower reduction rate in exposure dose and health impact. There are also some minor reasons. Remediation included two phases which caused the total cost to increase. Also the remediation target is different, the exposure level after remediation in chapter 4 is a little higher than chapter 3. These minor differences did not have significant influence on the final result.

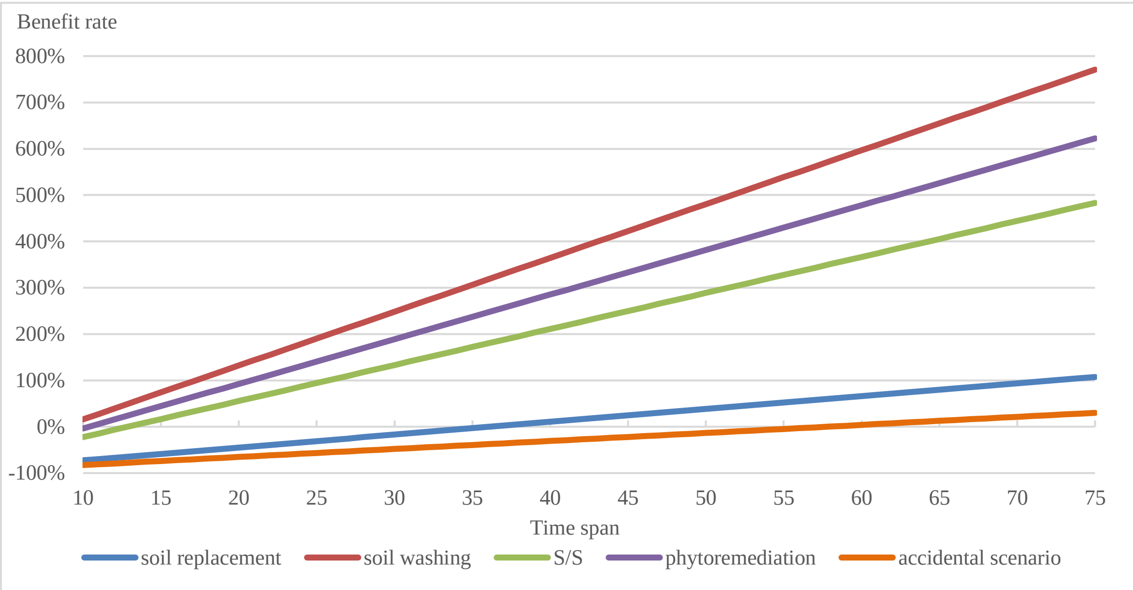

Fig 4-14. Benefit rate under different time span

The variation of benefit rate with change in time span under normal and disaster scenario is shown in **Fig 4-14**. Because the yearly benefit is the same for each scenario, the net benefit and benefit rate are proportional to the time span. Therefore, the variation is expressed with straight lines. The benefit under accidental scenario is lower than normal scenarios. For the short-term (10 years) benefit, only the benefit of soil washing is positive, while for long-term (75 years) benefit, benefit is positive under all the scenarios. Therefore, there exists threshold values for the benefit rate to be converted from negative to positive. This threshold time span value is 36 years for soil replacement, 13 years for solidification/stabilization, and 11 years for phytoremediation under normal scenarios.

The threshold time span value is 58 years for remediation under accidental scenario. Judging from this result, time span has a high influence on the benefit estimation. Therefore, time span is an important issue in cost-benefit analysis for remediation based on health benefit. The time consideration of benefit analysis needs to be considered carefully in practice.

4.5. Conclusion

In this research, we develop a hypothetical tailing dam failure scenario for the Jinding Pb/Zn mine and estimate the health impact of the HM exposure and the economic feasibility of the corresponding remediation process. The health impact is high after the tailing dam failure, and remediation action is necessary. However, the remediation process may not be economically feasible. Because remediation is necessary after tailing dam failure, local governments should put more effort into tailing dam management to avoid the unbeneficial remediation process. The result of this research provides information on the cost-effectiveness of soil remediation under accidental scenarios, which was rarely discussed in previous studies. The methodology applied in this research has relatively low site dependence. Sensitivity analysis has proved certain robustness of the analysis approach, indicating the applicability of the methodology to different tailing dam sites.

The major limitation of this research is the high dependence on assumptions. This limitation, however, is difficult to avoid. For predictive studies on hypothetical tailing dam failure events, the failure was not yet occurred. Therefore, little accurate information about the actual failure event can be available. To improve the reliability of this type of study, simulation models of tailing dam failure with wide applicability and high accuracy are required. With improvements in scenario definition and data availability, a major improvement in the result of predictive studies is expected in the future.

CHAPTER 5: CONCLUSION

5.1 Research conclusion

The main conclusion of the thesis is summarized as follows:

(1) In this thesis, a quantification methodology of DALY is developed and tested. DALY provides a more comprehensive estimation of health impact than traditional indicators by combining the disease possibility and disease severity. A test study of average Hg risk assessment in China indicates that DALY is capable of evaluating health impact caused by heavy metal exposure. The health impact assessed by DALY is monetized and applied in cost-benefit analysis for agricultural soil remediation processes. The result shows that health benefit is an unneglectable type of benefit and is expected to be included in future studies.

(2) This research provides field scale information on economic feasibility of soil remediation for agricultural soil, which have seldomly been discussed by researchers. The efficiency and economic feasibility of soil remediation is discussed under normal and accidental scenarios. For the target site, Jinding Pb/Zn mining area, phytoremediation is the most feasible remediation alternative under normal scenario. Under dam failure scenario, a combination of soil replacement and phytoremediation can effectively reduce the health impact on residents. The recommended remediation alternative can provide reference for local decision makers. The different remediation strategy under normal and accidental scenarios also indicates that the selection of remediation alternative highly depends on site-specific parameters. Such as soil type, heavy metal concentration and remediation expectation. Proper remediation strategy is important for the success and efficiency of remediation projects. Therefore, information on health effect and economic feasibility of soil remediation is for decision making.

(3) Few studies have discussed the economic feasibility of soil remediation under accidental scenarios. However, if soil system is contaminated in an accident, soil remediation will be one of the most important parts of environmental recovery process. In this research, we focus on the tailing dam failure, which is the most common technical accident in mining areas. The result shows that the remediation may be not economic feasible if only short-term health benefit is considered. A comparison between normal and accidental scenarios shows that the benefit rate under accidental scenario is relatively low. In order to avoid the unbeneficial remediation process, tailing dam failures should be prevented before happening. Also, considering the heavy metal content of the released tailing, the prioritized type of heavy metal is different for normal and accidental scenarios. The result of this research addresses the significance of tailing dam management and provides reference for accident planning.

5.2 Research limitation and future perspective

The major limitation of the research is high dependence on assumptions and previous reports due to lack of site specific data. In the analysis of exposure assessment, assumptions are applied to defined release and exposure scenarios. And in the analysis of economic feasibility, cost and removal efficiency reported by previous researches are applied. These assumptions and utilization of previous results created uncertainties. The sensitivity analysis shows even though the result is considered robust within certain range of input parameter variation, several parameters are identified to have strong influence on the result. In future studies on the economic feasibility of soil remediation projects, better access to site-specific data will decrease the uncertainty of the result and will improve the quality of the results.

Another limitation is that in the cost-benefit analysis, this thesis mainly focuses on health benefit. The purpose of this thesis is to apply DALY and health benefit in cost-benefit analysis. Therefore, the benefit estimation is focused on the health aspect. However, in actual assessment for remediation projects, the included benefit category will be defined considering the demand of local government or decision makers. In this thesis we argued that the health benefit is an unneglectable type of benefit for agricultural soil remediation. Future studies are expected to include the discuss of health aspects.

ACKNOWLEDGMENT

I would like to thank my esteemed supervisor – Prof. Akihiro Tokai for his invaluable supervision, support and tutelage during the course of my PhD degree. Additionally, I would like to express gratitude to all the assistant professors, Ms. Leticia dos Muchangos, Ms. Lisa Ito, Mrs. Hoa Nyugen, Mr. Naoya Kojima, for their treasured support which was really influential in shaping my experiment methods and critiquing my results. My gratitude extends to the Japanese Ministry of Education, Culture, Sports, Science, and Technology (MEXT) and the Chinese Scholarship Council (CSC) for the funding opportunity to undertake my studies at the Department of engineering, Osaka University. I would also like to acknowledge the Environment Research and Technology Development Fund (JPMEERF18S11702) of the Environmental Restoration and Conservation Agency of Japan for supporting my research. Finally, I would like to thank all my family, my friends and lab mates, for a cherished time spent together in the lab, and in social settings, as well as their encouragement and support all through my studies.

LIST OF ABBEREVIATIONS

Heavy Metal (HM)

Mercury (Hg)

Lead (Pb)

Cadmium (Cd)

Arsenic (As)

Copper (Cu)

Zinc (Zn)

Chromium (Cr)

Federal Remediation Technologies Roundtable (FRTR)

Cost-Benefit Analysis (CBA)

US EPA (United States Environmental Protection Agency)

Hazard Quotient (HQ)

Reference dose (RfD)

Margin Of Exposure (MOE)

Lower side of BenchMark Dose (BMDL)

No Observed Effect Levels (NOEL)

No Observed Adverse Effect Levels (NOAEL)

Lowest Observed Adverse Effect Levels (LOAEL)

Disability-Adjusted Life Years (DALY)

Quality-Adjusted Life Years (QALY)

World Health Organization (WHO)

Methylmercury (MeHg)

Total Gaseous Mercury (TGM)

Particulate Boundary Mercury (PBM)

Average Daily Dose (ADD)

Inhalation rate (InhR)

Exposure frequency (EF)

Exposure duration (ED)

Body weight (BW)

Average exposure time (AT)

Ingestion rate (IngR)

Human skin surface area (SA)

skin adherence factor (SL)

Skin Absorption factor (ABS)

Food consumption rate (FC)

Years life lost (YLL)

Years lost to disability (YLD)

Disease probability (DP)

Disease weight (DW)

Geometric mean value (GM)

Geometric standard deviation (GSD)

Uncertainty factor (UF)

Modification factor (MF)

Single parameter (SA)

Pollutant Release and Transfer Register (PRTR)

Life cycle impact assessment (LCIA)

Stabilization/solidification (s/s)

Removal efficiency (RE)

Remediation duration (RD)

Unit cost (UC)

Geo-Environmental Risk Assessment System, tier-1 (GERAS-1)

Bioconcentration factor (BCF)

Financial burden (FB)

Initial optional cost (IC)

Annual operation cost (AC)

Net present value (NVP)

Benefit ratio (BR)

Variation ratio (VR)

Biomass (BM)

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SUPPLEMENTAL MATERIALS

Supplemental material S1: Calculation of DALY developed by Murray (1994)

The original DALY formula for the number of DALYs lost by one individual developed by Murray (1994) is as follows:

$$\int_{x=a}^{x=a+L} DCxe^{-\beta x}e^{-r(x-a)}dx$$

The solution of the definite integral from the age of onset a to a+L where L is the duration of disability or time lost due to premature mortality gives us the DALY formula for an individual:

$$-\left[\frac{DCe^{-\beta a}}{(\beta+r)^2}\left[e^{-(\beta+r)(L)}(1+(\beta+r)(L+a)-(1+(\beta+r)a)\right]\right]$$

D is the disability weight (or 1 for premature mortality), r is the discount rate, C is the age weighting correction constant, β is the parameter from the age-weighting function, a is the age of onset, and L is the duration of disability or time lost due to premature mortality. $Cxe^{-\beta x}$ is the function for age-weighting. In this research, no information is available on proper average age of onset, therefore age-weighting is not considered.

Supplemental material S2: Remediation duration of phytoremediation in Chapter 3.

For scenario 4, since the phytoremediation takes more than 1 year to remove the HM in soil, the length of remediation duration (RD_4) needs to be evaluated. Assuming the extraction rate of the hyper extraction plant is constant, as reported by Chandra et al. (2017):

$$\mathrm{RD}_{4k} = \lceil \mathrm{c}_{0jk} \times \mathrm{RE}_{4k}/\Delta\mathrm{c}_{k} \rceil$$

$$\Delta\mathrm{c}_{k} = \mathrm{cP} \times \mathrm{BM}/(\mathrm{D} \times \rho_{b})$$

Δc_k: yearly reduction rate of heavy metal concentration (mg/kg); cP: concentration of HM in plant tissue (mg/kg); BM: annual plant biomass production (t/m^2); D: depth of the remediated HM (m); ρ_b: soil bulk density (t/m^3).

As is extracted by Pteris vittata.

$$\Delta\mathrm{c}_1 = \mathrm{cP}_{1,\mathrm{PV}} \times \mathrm{BM}_{\mathrm{PV}}/(\mathrm{D} \times \rho_{b}) = 1373 \times 0.0036/(0.2 \times 1.3) = 19.01 \ \ (\mathrm{mg/kg})$$

$$\mathrm{RD}_{4,1} = \lceil \mathrm{c}_{0,j,1} \times \mathrm{RE}_{4,1}/\Delta\mathrm{c}_1 \rceil = \lceil 19.62 \times 42.92\%/19.01 \rceil = \lceil 0.44 \rceil = 1 \text{ year}$$

Cd is extracted by Sedum alfredii H..

$$\Delta\mathrm{c}_2 = \mathrm{cP}_{2,\mathrm{SA}} \times \mathrm{BM}_{\mathrm{SA}}/(\mathrm{D} \times \rho_{b}) = 708 \times 0.00072/(0.2 \times 1.3) = 1.96 \ \ (\mathrm{mg/kg})$$

$$\mathrm{RD}_{4,2} = \lceil \mathrm{c}_{0,j,2} \times \mathrm{RE}_{4,2}/\Delta\mathrm{c}_2 \rceil = \lceil 5.03 \times 98.07\%/1.96 \rceil = \lceil 2.52 \rceil = 3 \text{ year}$$

Pb is extracted by both Pteris vittata. and Sedum alfredii H..

$$\begin{aligned}\Delta\mathrm{c}_3 &= (\mathrm{cP}_{3,\mathrm{SA}} \times \mathrm{BM}_{\mathrm{SA}} + \mathrm{cP}_{3,\mathrm{SA}} \times \mathrm{BM}_{\mathrm{SA}})/(\mathrm{D} \times \rho_{b}) \\ &= (2290 \times 0.0036 + 680 \times 0.00072)/(0.2 \times 1.3) = 15.76 (\mathrm{mg/kg})\end{aligned}$$

$$\mathrm{RD}_{4,3} = \lceil \mathrm{c}_{0,j,3} \times \mathrm{RE}_{4,3}/\Delta\mathrm{c}_3 \rceil = \lceil 138.79 \times 81.27\%/15.76 \rceil = \lceil 7.16 \rceil = 8 \text{ year}$$

Table S1. Parameters applied and result of remediation duration of phytoremediation

Parameter	Value	Unit	Reference
cP of As in Pteris vittata. ($cP_{1,PV}$)	1373	mg/kg	a
cP of Pb in Pteris vittata. ($cP_{3,PV}$)	680	mg/kg	a
cP of Cd in Sedum alfredii H. ($cP_{2,SA}$)	708	mg/kg	a
cP of Pb in Sedum alfredii H. ($cP_{3,SA}$)	2290	mg/kg	a
BM of Pteris vittata. (BM_{PV})	0.0036	t/year	b
BM of Sedum alfredii H. (BM_{SA})	0.00072	t/year	c
D	0.2	m	d
ρ_b	1.3	t/m^3	e
Original soil As concentration ($c_{0,j,1}$)	19.62	mg/kg	f
Original soil Cd concentration ($c_{0,j,2}$)	5.03	mg/kg	f
Original soil Pb concentration ($c_{0,j,3}$)	138.79	mg/kg	f
As removal efficiency (RE_{41})	42.92%	-	d
Cd removal efficiency ($RE_{4,2}$)	98.07%	-	d
Pb removal efficiency ($RE_{4,3}$)	81.27%	-	d

References: a: (Wu et al., 2007); b: (Tongbing Chen et al., 2002); c: (Yang et al., 2002); d: assumption and calculation of this research; e: (Tuyishimire et al., 2022); f: (Jin, 2017b);)

Supplemental material S3: bioconcentration factor (BCF) for wheat

Due to data availability, we are unable to determine the area of agricultural lands for each type of crop. So we cannot consider multiple types of crops grown in the target area. Therefore, we choose to assume that the agricultural land in the target area grows the same crop.

According to Ministry of Civil Affairs of PRC (2016), The main crop species grown in the target area is wheat and corn. Therefore, wheat and corn are our priority choice. For dietary consumption factors, we apply the factors in the "Chinese Exposure Factors Handbook" reported by Duan et al. (2015). In this handbook, the average consumption is only available for rice, flour and "other cereal", which includes corn. Since flour is mostly made of wheat, we choose wheat as the target crop in our research because of data availability.

We apply the bioconcentration factor reported by Wang et al. (2017) (Table S2).

Table S2. Bioconcentration factor (BCF) of wheat applied in the present research (a different value of BCF is applied for a different range of soil concentration. To avoid extreme values, the top group and bottom group is removed from the original result and is not applied in the present research)

	Soil concentration range	BCF
As	0~11.25	0.006
	11.25~15	0.011
	15~	0.009
Cd	0~0.15	0.242
	0.15~0.2	0.269
	0.2~0.4	0.197
	0.4~0.6	0.257
	0.6~1.2	0.328
	1.2~	0.307
Pb	0~26.25	0.031
	26.25~35	0.021
	35~	0.023

(Extracted and modified from Wang et al., 2017)

Supplemental material S4: The GERAS-1 and CSOIL model:

CSOIL 2000 calculates the risks that humans are exposed to if they come into contact with soil contamination. Humans can be exposed to contaminated soil via different exposure routes (soil, air, water and crops). Physical-chemical properties of the contaminant in soil air, soil particles and groundwater also have an influence on the exposure. The GERAS-1 is developed based on CSOIL with additional information of soil property and groundwater consumption in Japan. The calculation between the two models is similar.

Due to difference in physical and chemical property between heavy metals (HM) and organic pollutants, the calculation for the two are different. In this material, information about the calculation of HM exposure in CSOIL is introduced.

In GERAS, the concentration in exposure media are estimated by the following equations (Brand et al., 2007; Kawabe et al., 2003)

S4.1 Equations partition soil, water and air

S4.1.1 Fugacity calculation

♦ Za = 0

♦ Zs = Kp*SD*Zw/Vs

♦ Vs = 1-Va-Vw

Za : fugacity capacity in soil air [mol.m^{-3}.Pa]

Zw : fugacity capacity in soil moisture [mol.m^{-3}.Pa]

Zs : fugacity capacity in soil solid [mol.m^{-3}.Pa]

Kp : partition coefficient soil-water

(Kp value applied in this research is 1800 for As, 5800 for Cd and 150000 for Pb [dm^3/kg])

SD : dry bulk density [1.3 kg/dm^3]

Va : volume fraction soil air [-]

Vw : volume fraction soil moisture [-]

S4.1.2 Mass fraction calculations

♦ Pa = (Za*Va)/(Za*Va+Zw*Vw+Zs*Vs) = 0

♦ Pw = (Zw*Vw)/(Za*Va+Zw*Vw+Zs*Vs) = Vw/(Vw+Kp*SD)

♦ Ps = (Zs*Vs)/(Za*Va+Zw*Vw+Zs*Vs) = 1-PW

Vs : volume fraction soil solid [-]

Pa : mass fraction in soil air [-]

Pw : mass fraction in soil moisture [-]

Ps : mass fraction in soil solid [-]

S4.1.3 Concentrations in air, water and soil

Soil HM concentration (Cs) is the input parameter

HM concentration in pore water is calculated by solubility.

♦ Cpw = Cs*SD*Pw*/Vw

Cpw : HM concentration in pore water [$g.m^{-3}$]

HM concentration is assumed 0 in this model.

♦ Csa = 0

Csa : concentration in soil air [$g.m^{-3}$] or [$mg.dm^{-3}$]

S4.2 Equations soil ingestion, inhalation

S4.2.1 Exposure via soil ingestion

Conventions:

c : child

a : adult

L : lifelong

Fa : relative sorption factor [-]

BWc,a : bodyweight (adult 62, child 20) [kg]

Cs : initial soil concentration (total concentration in gas-, water- and solid phase) [mg.kg-1]

Exposure, the lifelong average exposure [mg.kg-1 bw.d-1]

Lifelong [70 year]

Child

♦ DIc = AIDc*Cs*Fa/BWc

DIc : exposure via ingestion of soil - child [$mg.kg^{-1}$ $bw.d^{-1}$]

AIDc : daily intake soil – child [$1.0 \cdot 10^{-4}$ kg $dw.d^{-1}$]

Cs : initial soil concentration (total concentration in gas-, water- and solid phase) [$mg.kg^{-1}$]

Fa : relative absorption factor soil particle [1.0 -]

BWc : bodyweight child [20 kg]

Adult

♦ DIa = AIDa*Cs*Fa/BWa

DIa : exposure via ingestion of soil – adult [$mg.kg^{-1}$ $bw.d^{-1}$]

AIDa : daily intake soil - adult [$5.0 \cdot 10^{-5}$ kg $dw.d^{-1}$]

Cs : initial soil concentration (total concentration in gas-, water- and solid phase) [$mg.kg^{-1}$]

Fa : relative absorption factor soil particle [1.0 -]

BWa : bodyweight adult [62 kg]

Lifelong average

♦ DIL =(6*DIc+64*DIa)/70

DIL : exposure via ingestion of soil lifelong average [mg.kg^{-1} bw.d^{-1}]

DIc : exposure via ingestion of soil lifelong average - child [mg.kg^{-1} bw.d^{-1}]

DIa : exposure via ingestion of soil – adult [mg.kg^{-1} bw.d^{-1}]

S4.2.2 Exposure via inhalation of soil particles

Child

♦ IPc = Cs*ITSPc*Fr*Fa/BWc

IPc : exposure via inhalation of soil particles - child [mg.kg^{-1} bw.d^{-1}]

Cs : initial soil concentration (total concentration in gas-, water- and solid phase) [mg.kg^{-1}]

ITSPc : inhaled amount of soil particles – child [3.13·10^{-7} kg.d^{-1}]

Fr : retention factor soil particles in lungs [0.75 -]

Fa : relative absorption factor [1.0 -]

BWc : bodyweight child [20 kg]

♦ ITSPc = TSP*frs*AVc*t*tf

TSP : amount of suspended particles in air [mg.m^{-3}]

Indoors 0.75 * 70 = 52.5 µg.m^{-3}

Outdoors 70 µg.m^{-3}

frs : fraction soil particles in air [-]

Indoors 0.8

Outdoors 0.5

AVc : air volume child [0.317m^{3}.h^{-1}]

t : length of time of exposure [h]

Indoors 16 h

Outdoors 8 h

tf : correction factor exposure daily ➔ yearly - child [-]

Indoors 1.322

Outdoors 0.357

Adult

♦ IPa = Cs*ITSPa*Fr*Fa/BWa

IPa : exposure via inhalation of soil particles – adult [mg.kg^{-1} bw.d^{-1}]

Cs : initial soil concentration (total concentration in gas-, water- and solid phase) [mg.kg^{-1}]

ITSPa : inhaled amount of soil particles – adult [8.33·10^{-7} kg.d^{-1}]

Fr : retention factor soil particles in lungs [0.75 -]

Fa : relative absorption factor [1.0 -]

BWa : bodyweight adult [62 kg]

♦ ITSPa = TSP*frs*AVa*t*tf

TSP : amount of suspended particles in air [$mg.m^{-3}$]

Indoors 0.75 * 70 = 52.5 $\mu g.m^{-3}$

Outdoors 70 $\mu g.m^{-3}$

frs : fraction soil particles in air [-]

Indoors 0.8

Outdoors 0.5

AVa : air volume adult [0.833 $m^3.h^{-1}$]

t : length of time of exposure [h]

Indoors 8 h

Outdoors 8 h

tf : correction factor exposure daily ➔ yearly – adult [-]

Indoors 2.856

Outdoors 0.143

Lifelong average

♦ IPL = (6*IPc+64*IPa)/70

IPL : exposure via inhalation of soil particles lifelong average [$mg.kg^{-1}$ $bw.d^{-1}$]

IPc : exposure via inhalation of soil particles – child [$mg.kg^{-1}$ $bw.d^{-1}$]

IPa : exposure via inhalation of soil particles – adult [$mg.kg^{-1}$ $bw.d^{-1}$]

S4.3 Exposure via crop consumption

1.3.1 HM concentration in crop

♦ Cpr = Cs*BCFr

Cpr : average concentration in root crops [$mg.kg^{-1}$]

Cs : initial soil concentration (total concentration in gas-, water- and solid phase) [$mg.kg^{-1}$]

BCFr: bioconcentration factor for roots [0.009 for As, 0.31 for Cd, 0.017 for Pb -]

♦ Cps = Cs*BCFs+ Cdp

Cps : average concentration in shoot crops [$mg.kg^{-1}$]

Cdp : concentration obtained from deposition [$mg.kg^{-1}$]

BCFs: bioconcentration factor for shoots [-]

♦ ln(BCFr/BCFs) = 2.67-1.12*ln (Kp)

♦ Cdp = TSpo*DRo*frs*Cs*[fin/(Yv*fEi)]*{1-(1-exp[-fEi*te]/(fEi*te)}

TSpo : particle concentration in atmosphere [$mg.m^{-3}$]

DRo : deposition rate [m/day]

frs : fraction atmospheric particles captured by crops [-] fin : deposit disturbance factor [-]

Yv : crop yield in 1 square meter [kg.m-2]

fEi : weather factor [1/day]

te : growing time for crops [day]

Child

♦ DCCc = (Qfvkc*Cpr)+(Qfvbc*Cps)*fv*fa/BWc

Qfvkc : root crop consumption – child [0.0595 kg fw.d^{-1}]

Qfvbc : leafy crop consumption - child [0.0583 kg fw.d^{-1}]

fv : fraction contaminated crop [0.1 -]

fa : relative sorption factor [1.0 -]

BWc : bodyweight child [20 kg]

Adult

♦ DCCa = (Qfvka*Cpr)+(Qfvba*Cps)*fv*fa/BWa

Qfvka : root crop consumption – adult [0.122 kg fw.d^{-1}]

Qfvba : leafy crop consumption – adult [0.139 kg fw.d^{-1}]

BWa : bodyweight adult [62 kg]

Lifelong average

♦ VI= (6*VIc+64*VIa)/70

VI : exposure via ingestion of crops lifelong average [mg.kg^{-1} bw.d^{-1}]

VIc : exposure via ingestion of crops – child [mg.kg^{-1} bw.d^{-1}]

VIa : exposure via ingestion of crops – adult [mg.kg^{-1} bw.d^{-1}]

S4.4 Exposure via groundwater consumption

S4.4.1 Concentration in groundwater

♦ Cgw = df*Cpw

Cgw: HM concentration in groundwater [g.m^{-3}]

df: dilution factor [0.1 -]

Child

♦ DIgwc = Qdw*Cgw*fga/BWc

DIgwc : exposure via groundwater consumption – child [mg.kg^{-1} bw.d^{-1}]

Qdw : groundwater consumption children [1L/day]

fga : absorption factor groundwater [1.0 -]

BWc : bodyweight child [20 kg]

Adult

♦ DIgwa = Qdw*Cgw*fga/Bwa

DIgwa : exposure via groundwater consumption – adult [mg.kg-1 bw.d-1]

BWa : bodyweight adult [62 kg]

Lifelong average

♦ DIgw= (6*DIgwc +64*DIgwa)/70

DIgwa : exposure via groundwater consumption lifetime average [$mg.kg^{-1}$ $bw.d^{-1}$]

S4.5 Total exposure

Child

♦ TEc = DIc+IPc+DCCc+DIgwc

Tec : total exposure child [$mg.kg^{-1}$ $bw.d^{-1}$]

Adult

♦ TEa = DIz+IPz+DCCz+DIgwa

TEa : total exposure adult [$mg.kg^{-1}$ $bw.d^{-1}$]

Lifelong average

♦ TE = (6*TEc+64*TEa)/70

TE : total exposure lifetime average [$mg.kg^{-1}$ $bw.d^{-1}$]