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Mapping mining-affected water pollution in China: Status, patterns, risks, and implications

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Abstract: Mining-affected water pollution poses a serious threat to human health and economic prosperity globally. The human toxicity and ecosystem impacts induced by mining activities have achieved considerable public, scientific, and regulatory attention. In this study, a comprehensive database of 8433 water samples from 211 coal mines and 87 metal mines in China was established to reveal the national status and spatial heterogeneity of mining-affected water pollution, human health risks, and their potential multifaceted challenges. The results show that the concentrations of sulfate, Fe, Mn, Al, and several trace elements in the mining-affected water of metal mines are generally higher than those of coal mines, especially in acid water ($pH < 6.5$). In terms of spatial distribution, the gridded data demonstrates that the southern regions in China, especially Guizhou, Guangdong, Fujian, Jiangxi, Hunan, and Guangxi provinces/autonomous regions, are the hotspots of mining-affected water pollution (*i.e.*, low pH as well as high sulfate, Fe, Mn, and heavy metals). The unacceptable carcinogenic risks caused by poor-quality surface water and groundwater are observed in 51.52% (for adults) and 29.29% (for children) of the mining areas. Moreover, severe non-carcinogenic risks are also identified in 68.07% and 80.67% of mining areas for adults and children, respectively. Overall, the acidic and metal-rich water exhibits a widespread and detrimental impact in China, especially in the southern regions, posing significant risks to planetary health by degrading surface water and groundwater quality, destroying biodiversity, and threatening human well-being. This study provides a thorough set of scientific data on surface water and groundwater quality in mining areas to guide policymakers in designing differentiated management strategies for the sustainable development of coal and metal mines.

Keywords: Mining-affected water pollution; Spatial patterns; Risk assessment; Adverse effects; Differentiated management.

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1 Introduction

The extraction and processing of coal and metalliferous mineral resources, essential materials for global socio-economic development, have caused detrimental impacts on aquatic ecosystems, soil ecosystems, living organisms, and human health worldwide (Blowes et al., 2014; Li et al., 2014; Havig et al., 2017; Ighalo et al., 2022). Mine drainage and leachate from active and abandoned mines achieve a global concern, as those continue to release harmful substances into underlying geological materials or adjacent water bodies for decades, inevitably leading to the degradation of both surface water and groundwater quality (Acharya and Kharel, 2020; Ighalo and Adeniyi, 2020). In particular, the environmental risks induced by acid mine drainage (AMD) have been ranked second only to global warming and ozone depletion (Moodley et al., 2018; Ai et al., 2023). Mining-affected water is generally characterized as metalliferous. Certain metals (*e.g.*, Cu, Fe, Mn, and Zn) are of great importance to human metabolism but can become toxic when present at high levels in surface water and groundwater (Wei et al., 2022). Other non-essential heavy metals (HMs), including As, Cd, Cr, Hg, Ni, and Pb, lack nutritional or beneficial effects for humans. They can be toxic even at low concentrations and are therefore recognized as carcinogenic, mutagenic, and teratogenic. In addition, the persistence, toxicity, mobility, and non-biodegradability of HMs potentially form an enduring environmental footprint that jeopardizes ecosystems (Dippong et al., 2024). Consequently, there is a growing demand in mining areas for assessing the status of pollution and associated risks, as well as developing more effective management strategies and policies to mitigate those detrimental impacts.

Exploring the heterogeneity, risks, and threats of mining-affected water pollution is desirable but remains challenging. More recently, an increasing number of studies have been focused on the mining-affected water pollution from coal mines in major coal-producing countries (Sun et al., 2013; Acharya and Kharel, 2020; Dong et al., 2022; Ai et al., 2023; Hou et al., 2024; Kumar et al., 2024). For instance, Acharya and Kharel (2020) provided an in-depth overview of the formation and effects of AMD from coal mining in the United States, reviewed prediction and treatment methods, identified key research gaps, and explored the challenges and opportunities that AMD posed for scientists and researchers. Ai et al. (2023) developed a conceptual model to illustrate the formation and evolution of AMD in the coal mines from a perspective of life-cycle while identifying the critical governing factors and treatment technologies of AMD across abandoned mines in major coal-producing countries, including China, the United States, the United Kingdom, Australia, and India. In fact, coal and metal mines have different priority pollutants and levels of pollution due to variations in geological conditions and mineral extraction methods (Yu et al., 2024). Comparative studies of the status, heterogeneity, risks, and impacts of water pollution in coal and metal mines achieved the limited concerns so far, which are essential for developing remediation strategies and implementing risk-based management to achieve sustainable development in mining areas associated with the mineral economy. Moreover, comparative studies play an important role in designing differentiated management practices.

To our knowledge, previous studies have provided a solid basis for the soil pollution status of HMs and their related health risk at the national or global scale (Li et al., 2014; Liu et al., 2020; Hou et al., 2023; Shi et al., 2023). For example, Shi et al. (2023) revealed the spatiotemporal distribution of soil HM concentrations based on studies conducted between 1977 and 2020 and assessed the ecological and human health risks considering different land use types at the national scale. Yu et al. (2024) provided a more comprehensive analysis of pollution characteristics, spatial

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distribution, major influencing factors, and probabilistic health risks of soil potentially toxic elements based on data from 110 coal mines and 168 metal mines across China. However, systematic studies on water pollution status and risks have yet to be undertaken at a national or even broader scale, as current research only focused on water pollution and risks in specific mining areas (He et al., 1998; Xiao et al., 2003; Wang et al., 2019; Chen et al., 2020; Wang et al., 2023). Therefore, it is necessary to implement deep mining of massive hydrochemical data and establish a nationwide database that can identify the spatial heterogeneity of mining-affected water pollution and support risk assessment.

China, the second-largest economy worldwide, has various and extensive mineral resources (Li et al., 2014). It has been demonstrated that there are 171 types of mineral resources in China, with proven reserves accounting for 12% of the world's mineral resources (Hu et al., 2009). Furthermore, China is one of the largest global producers and consumers of metals and metalloids, such as iron, manganese, zinc, lead, antimony, and tin (Gunson and Jian, 2001). China's coal reserves of 143,197 million tons (Mt) rank fourth worldwide, while production of 2,971 Mt is the highest (Blowes et al., 2014; Ai et al., 2023). In recent years, China has put forward a series of monitoring, prevention, management, and remediation measures to improve water quality and ensure water supply safety. However, the detrimental impacts triggered by mining activities on the aquatic environment have not been well managed. In China, approximately 12,000 coal mines have been closed since 2010 in order to address the issue showing the lower economic profits and higher environmental burden (Ma et al., 2020). These policies recover water storage in the mine region while the acidity, sulfate, and dissolved metals derived from the intricate geochemical reactions from the weathering products of exposed sulfides can subsequently migrate and transform in the

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recovering groundwater, making the water systems highly vulnerable to disruption.

Therefore, the objectives of the study are: (i) to establish a high-quality and national database containing basic water quality information for typical coal and metal mines; (ii) to reveal spatial heterogeneity of mining-affected water and evaluate health risks posed by potentially toxic elements from coal and metal mines drainage in China; and (iii) to highlight the negative impacts and discuss the management implications in the differentiated policy for different mine types (coal or metal) and multiple mining phases (active or abandoned). Exploring the spatial heterogeneity of mining-affected water in China is of great importance to achieve deep insights for designing the targeted and promising mitigation strategies at the different spatial scales, which is critical to implementing the optimal trade-offs between green mining and human health.

2 Data and methodology

2.1 Data mining and processing

In this study, we compiled the dataset of surface water and groundwater affected by mining activities in China collected from the published literature over the past decades, which was mainly collected from mainstream online bibliographic databases, such as China National Knowledge Infrastructure, China Wanfang Literature Database, Web of Science, Elsevier, Springer, Wiley, Taylor & Francis, and the Multidisciplinary Digital Publishing Institute. The screening keywords were 'China', 'coal mine', 'metal mine', 'acid mine drainage', 'mine water', 'surface water', 'groundwater', 'hydrochemistry', and 'heavy metals'. All retrieved literature was downloaded by 2024/4/25, and the irrelevant studies were eliminated based on their abstracts, data, and full-text content.

2.2 Quality assessment

To ensure the reliability of the data, the collected literature was assessed for quality based on the following criteria: (i) adhering to strict quality assurance/quality control procedures during sampling, storage, and laboratory testing to ensure consistency, precision, and accuracy of results; (ii) extracting the sampling year (if not stated, the received or published date of the manuscript was adopted); (iii) extracting the latitude and longitude coordinates of the sampling site, mine or the county-level city in which they are located; and (iv) extracting the concentration of the featured component or statistical values (minimum value, mean value and maximum value) based on the original data.

2.3 Database establishment

To assess the national extent of mining-affected water pollution, a comprehensive database of 8433 data (6175 coal mine data and 2258 metal mine data) derived from 298 mines was established, including 211 coal mines and 87 metal mines (*i.e.*, antimony mine, copper mine, gold mine, hematite mine, iron mine, lead-zinc mine, molybdenum mine, polymetallic mine, pyrite mine, rare earth mine, thallium-mercury mine, tin mine, tungsten mine, and uranium mine). The spatial distribution of the sampling sites used in the study and the data classification at the provincial level are displayed in Fig. 1. The typical mine lists are shown in Table S1. The detailed information we collected, including the sample ID, province, county/mine name, latitude (N), longitude (E), mine type, mine status (active or abandoned), sampling year, sampling month, sample type, basic physiochemical characteristics (pH, temperature (T), electrical conductivity (EC), oxidation reduction potential (ORP), dissolved oxygen (DO), and total dissolved solids (TDS)), major

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cation/anion ions (Na^+, K^+, Ca^{2+}, Mg^{2+}, Cl^-, SO_4^{2-}, HCO_3^-, NO_3^- and F^-), Fe, Mn, Al, HMs (Cr, Ni, Cu, Zn, As, Cd, Hg, and Pb) and data source.

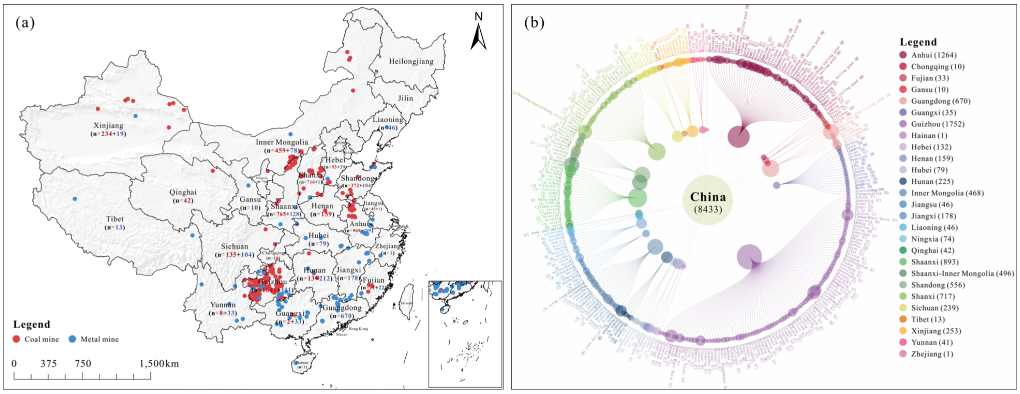

Figure 1. The information on data sources for this study, including (a) spatial distribution of the sampling sites, and (b) data classification at the provincial level. In Fig. 1a, the values in the brackets represent the sample size (n), specially, the red and blue numbers are the sample size of coal mines and metal mines for different provinces, respectively. In Fig. 1b, the size of the inner circle represents the sample size at the provincial level, while that of the outer circle represents the sample size at the specific mine level (the letters C and M in the brackets represent coal mines and metal mines, respectively). In the legend, the value in the bracket represents the sample size of the different provinces.

2.4 Risk assessment

Human exposure to metals can occur through various pathways, including ingestion and dermal contact with contaminated water. Therefore, these two pathways are considered to assess the potential human risks, *i.e.*, non-carcinogenic risks (NCRs) and carcinogenic risks (CRs), for adults and children. The model developed by the U.S. Environmental Protection Agency (US EPA) is employed for risk assessment in this study (US EPA, 2004; 2011):

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$$ADI_{ing} = \frac{C_w \times IR \times EF \times ED}{BW \times AT} \quad (1)$$

$$ADI_{der} = \frac{C_w \times K_p \times ET \times SA \times EF \times ED \times CF}{BW \times AT} \quad (2)$$

where ADI_{ing} and ADI_{der} are the average daily intake by ingestion and dermal adsorption (mg/kg·d), respectively; C_w is the metal concentration in the mining-affected water (mg/L); IR is the ingestion rate (L/d); EF is the exposure frequency (d/yr); ED is the exposure duration (yr); K_p is the permeability coefficient of skin (cm/h); ET is the exposure time (h/d); SA is the exposed skin surface area (cm^2); CF is the conversion factor (L/cm^3), which is set to 0.001 in the study; BW is the body weight (kg); and AT is the averaging time (d).

The hazard quotient (HQ) and hazard index (HI) are used to determine the NCRs to human health (Dippong et al., 2024). The HQ to residents (adults and children) from metal exposure via ingestion (HQ_{ing}) and dermal absorption (HQ_{der}) are quantified:

$$HQ_{ing} = \frac{ADI_{ing}}{RfD_o} \text{ and } HQ_{der} = \frac{ADI_{der}}{RfD_{der}} \quad (3)$$

$$HI = \sum HQ_i = HQ_{ing} + HQ_{der} \quad (4)$$

where HI is the hazard index, which is the sum of HQ. HI > 1 indicates potential adverse effects on human health, whereas HI < 1 suggests no NCR is present; RfD_o is the reference dose for oral intake; and RfD_{der} is the reference dose for dermal exposure, which can be calculated by:

$$RfD_{der} = RfD_o \times ABS_{GI} \quad (5)$$

where ABS_{GI} is the gastrointestinal digestion coefficient (unitless).

The CRs to residents from ingestion and dermal absorption of mining-affected water are determined using Eqs. (6) and (7):

$$CR_{ing} = ADI_{ing} \times SF \text{ and } CR_{der} = ADI_{der} \times SF \quad (6)$$

$$TCR=\sum CR_i = CR_{ing} + CR_{der} \quad (7)$$

where TCR is the total CR, if TCR > 10^{-4}, there is a significant risk to humans, and if $10^{-6} \leq$ TCR $\leq 10^{-4}$, the risk is generally acceptable; CR_{ing} and CR_{der} are the CRs induced by ingestion and dermal contact with mining-affected water, respectively; SF is the slope factor. The detailed values of the parameters in the above formula (Eqs. (1) - (7)) are represented in Tables S2 and S3.

3 Results and analysis

3.1 Overview of mining-affected water in China

All the data were collected from 26 provinces in China, while no data met the data extraction principles in Beijing, Tianjin, Shanghai, Heilongjiang, Jilin, Hong Kong, Macao, and Taiwan. Guizhou, Anhui, Shaanxi, Shanxi, and Guangdong provinces have relatively large sample sizes at the provincial level, with Guizhou and Anhui having the most data, accounting for 20.78% and 14.99% of the database, respectively (Fig. 1). The spatial distributions of the sample sizes for each component at the 0.5° grid scale are depicted in Fig. S1. Generally, most of the data show the basic information of the water sample, such as pH values and the concentrations of major cation and anion ions. In addition, many studies have focused on the status of trace elements in the mining areas which shows that the number of sampling sites reaches 2241, 2265, 1401, 952, 691, 1563, 1575, 1451, 1627, 280, and 1425 for the concentrations of Fe, Mn, Al, Cr, Ni, Cu, Zn, As, Cd, Hg, and Pb, respectively. Hence, the Fe, Mn, Cd, Cu, and Zn are the hotspots for mine water research.

The pH values and multi-component concentrations of mining-affected water in China are presented in Fig. 2. It should be highlighted that the mean values may result in the overestimation, as some extremely high values are observed in the surveyed mines, such as the Baiyin copper mine, Bainiuchang polymetallic mine, Zijinshan copper mine, and so on. Consequently, median values are chosen to represent the national characteristics of the mining-affected water pollution in this section.

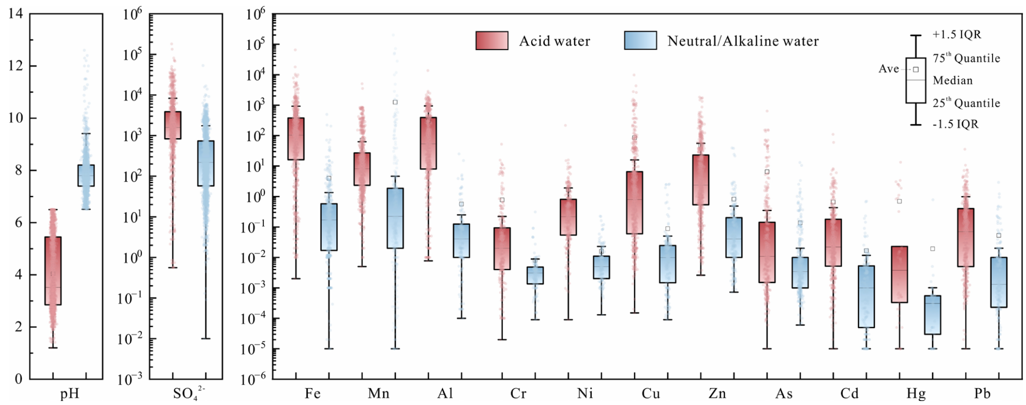

Figure 2. Boxplots of the pH values and multi-component concentrations (mg/L) of mining-affected water in China.

The pH values of acid water (*i.e.*, pH < 6.5) range from 1.20 to 6.50, with a median (interquartile range, IQR) of 3.52 (2.85, 5.45) (CV = 34.39%). In comparison, neutral/alkaline water has pH values between 6.51 and 12.60, with a median of 7.80 (IQR: 7.40, 8.20) (CV = 8.99%). Generally, the SO_4^{2-} concentration of acid water is higher than that of neutral/alkaline water (Figs. 2-3), with the former ranging from 0.56 to 181000 mg/L (25th percentile = 834.33 mg/L, median = 1580.16 mg/L, 75th percentile = 3864.08 mg/L, and CV = 222.70%) and the latter from 0.01 to 52915 mg/L (25th percentile = 52.87 mg/L, median = 181.27 mg/L, 75th percentile = 558.73 mg/L, and CV = 264.02%). Furthermore, the results indicate that the detectable medians of the multi-metal concentrations (mg/L) in acid water follow this order: Fe (103.30) > Al (53.90) > Mn (8.1080) > Zn (2.3685) > Cu (0.8010) > Ni (0.2166) > Pb (0.0700) > Cd (0.0220) > Cr (0.0200) > As (0.0108) > Hg (0.0038), while that of the neutral/alkaline water is Mn (0.2164) > Fe (0.1700) >

Zn (0.0391) > Al (0.0350) > Cu (0.0100) > Ni (0.0050) > As (0.0034) > Cr (0.0031) > Pb (0.0012) > Cd (0.0004) > Hg (0.0003).

3.1.1 Contents of acid mining-affected water in China

The multi-component concentrations of mining-affected water in both coal and metal mines are displayed in Figs. 3 and S2. It is obvious that the concentrations of sulfate, Fe, Mn, Al, and several trace elements in the mining-affected water of most metal mines are higher than those of coal mines, especially in mining-affected water with low pH (< 6.5). For acid mining-affected water, the pH of coal mines is approximately 1.90 - 6.50 (with a median of 4.50), while the pH of metal mines is approximately 1.20 - 6.50 (with a median of 3.10). The medians (IQR) of SO_4^{2-} are 1381.59 (871.41, 1954.73) mg/L and 2982.00 (778.15, 10200.00) mg/L for coal mines and metal mines, respectively. In conjunction with Fig. S3, it can be seen that the detectable medians of multi-metal concentrations (mg/L) in coal mining-affected water are 77.41 (Fe), 12.87 (Al), 3.50 (Mn), 0.4211 (Zn), 0.1796 (Ni), 0.0431 (Cu), 0.0080 (Cr), 0.0036 (Cd), 0.0034 (As), 0.0023 (Pb), and 0.0004 (Hg), respectively. Additionally, the detectable medians of multi-metal concentrations (mg/L) in metal mining-affected water are 152.00 (Al), 113.77 (Fe), 15.82 (Mn), 7.200 (Zn), 1.7325 (Cu), 0.2142 (Ni), 0.1498 (Pb), 0.0500 (Cr), 0.0383 (Cd), 0.0281 (As), and 0.0090 (Hg), respectively.

3.1.2 Contents of non-acid mining-affected water

Similarly, for non-acid mining-affected water, the pH values of coal mines are about 6.51 - 11.51 (with a median of 7.82), while those of metal mines are about 6.51 - 12.60 (with a median of 7.70). The medians (IQR) of SO_4^{2-} are 193.51 (48.97, 582.70) mg/L and 157.41 (60.23, 425.33)

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mg/L for coal mines and metal mines, respectively. As shown in Fig. S3, the results indicate that the detectable medians of multi-metal concentrations (mg/L) in coal mines are in the order of Fe (0.2500) > Mn (0.0204) > Al (0.0200) > Zn (0.0048) > Ni (0.0040) > Cr (0.0022) > As (0.0016) > Cu (0.0010) > Pb (0.0003) > Hg (0.0001) > Cd (0.0000), respectively. Additionally, the detectable median concentrations (mg/L) of Mn, Zn, Al, Fe, Cu, Pb, Ni, Cr, As, Cd, and Hg in metal mines are 0.7612, 0.0692, 0.0575, 0.0484, 0.0196, 0.0068, 0.0065, 0.0042, 0.0040, 0.0017, and 0.0003, respectively.

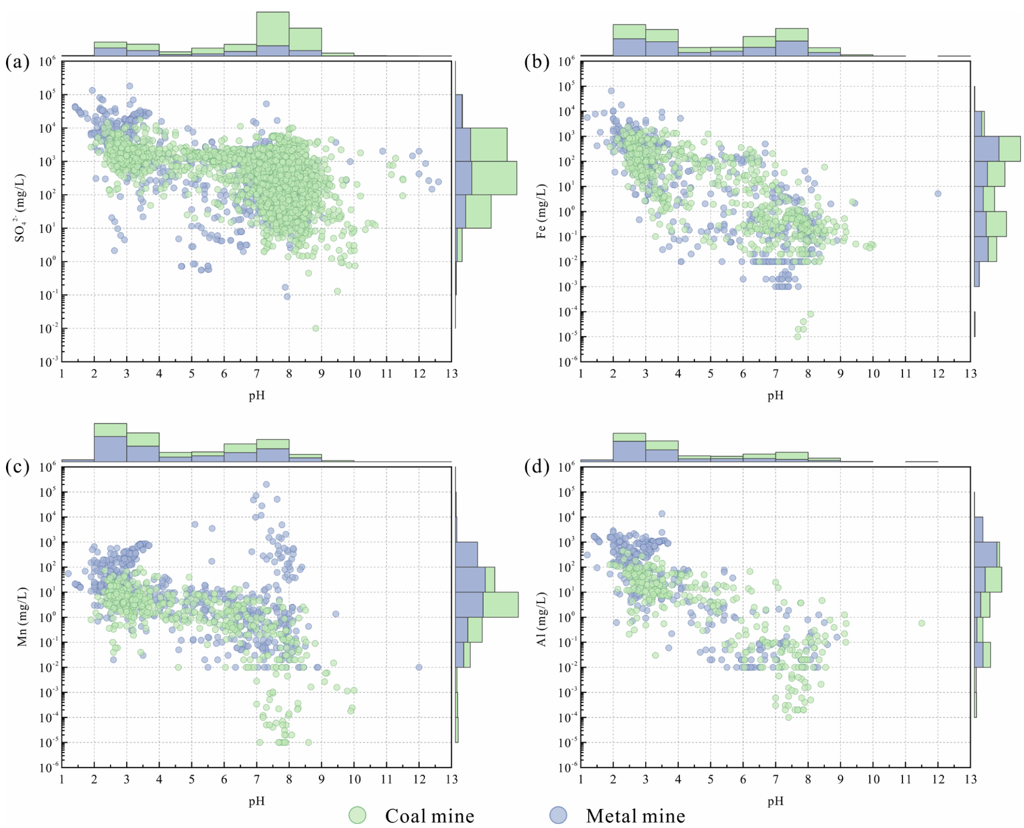

Figure 3. The respective relationships of pH versus (a) SO_4^{2-}, (b) Fe, (c) Mn, and (d) Al in coal and metal mines.

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3.2 *Spatial patterns of mining-affected water pollution in China*

The coal mines surveyed in the study are mainly located in the northern and southwestern regions, which together account for approximately 70% of the national coal production. This localized distribution aligns closely with the pattern of coal-mining belts in China. The southwestern and southern regions of China, rich in metallic mineral resources and with complex geological conditions, have been subject to frequent or unregulated mining activities for many years. Conversely, the western and northern regions are relatively poorly endowed with metal resources. (Yu et al., 2024). The mining-affected water is divided into 4 types in the study based on the multi-component characteristic, *i.e.*, with low pH, with high sulfate, with high Fe and Mn, and with high HMs. Given that mining activities have posed great threats to the surface water and groundwater, the classification criteria of each component in the text are based on the data distribution, but more importantly, we refer to the Environmental Quality Standards for Surface Water (GB 3838-2002) and the Standard for Groundwater Quality (GB/T14848-2017) in China. The categories of water quality in the above documents are listed in Tables S4 and S5. The spatial distributions of mining-affected water pollution are exhibited in Figs. 4 and S4 to reveal contaminated hotspots. Overall, there is a decreasing trend in pollution levels of mining-affected water from southeast to northwest China is observed, showcasing distinct regional patterns.

3.2.1 *Mining-affected water with low pH*

Low pH mining-affected water, with pH values < 6.5 and generally between 2.0 and 4.0, is mainly distributed in southern China (Fig. 4a), especially Fujian, Guangdong, Guizhou, Hubei, Hunan, Jiangxi, and Yunnan provinces. Notably, the mining-affected water (at 0.5° grid scale) in

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Fuquan City of Guizhou province has a pH as low as 1.90. There are notable correlations between the different types of mining-affected water, *e.g.*, acid coal mine water is marked by high levels of sulfate, Fe, and Mn. Furthermore, acid water from metal mines not only shows elevated levels of sulfate, Fe, and Mn but also contains significantly higher concentrations of HMs. The water sample is caused by acid mine drainage, which is generally associated with the extraction and processing of sulfur-bearing metalliferous ore deposits (*e.g.*, pyrite, chalcopyrite, pyrrhotite, and sphalerite) and sulfide-rich coal in China, with sulfur mass fractions ranging from 0.3% to 5.0% (Blowes et al., 2014; Feng et al., 2014).

3.2.2 Mining-affected water with high sulfate

It is evident that there is a spatial consistency in the distribution of high-sulfate mining-affected water and acid water (Fig. 4b). Sulfate concentrations exceed 250 mg/L in 102 grids, accounting for 73.91% of the total number of grids (138, with available data) in China. Besides, the hotspots of high sulfate mining-affected water pollution are simultaneously observed in Anhui, Hebei, Shandong, Shannxi-Inner Mongolia, and Xinjiang provinces/autonomous regions, where the water samples' pH values are generally above 6.5. There are two pathways to produce non-acid high-sulfate water: (i) by pyrite oxidation followed by natural neutralization, and (ii) by dissolution of sulfur-bearing and gypsum minerals. For instance, the Ordovician limestone aquifer is composed of dolomite, which is the primary source of sulfate in southwest Shandong (*e.g.*, Hongshan-Zhaili mines), Anhui (*e.g.*, Huainan-Huaibei mines) and other mining areas. The above-mentioned spatial heterogeneities found in our study are in good agreement with the results of Feng et al. (2014).

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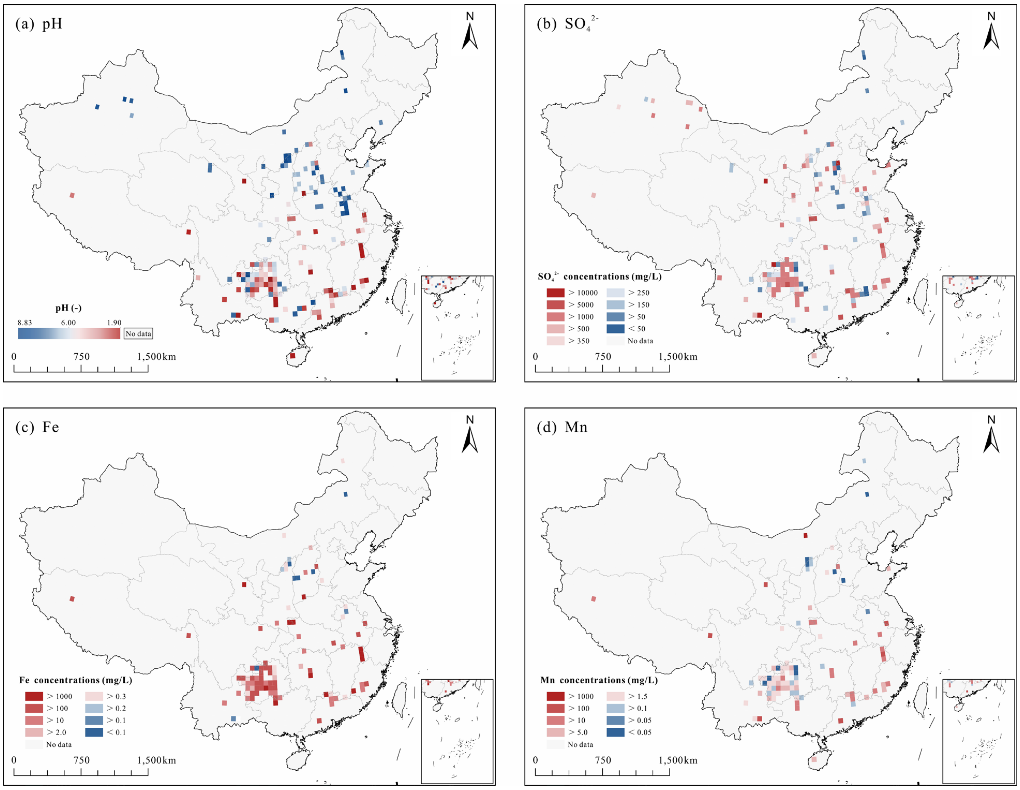

Figure 4. Spatial distributions of (a) the pH values; and the means of single component concentrations (mg/L) showing respective (b) SO_4^{2-}, (c) Fe, and (d) Mn, in mining-affected water on the 0.5° grid.

3.2.3 Mining-affected water with high Fe and Mn

Nationally, the concentrations of Fe and Mn in water affected by mining are widely over 0.3 mg/L and 0.1 mg/L, respectively. As displayed in Fig. 4c, the Fe pollution hotspots are mainly located in the Fujian (*e.g.*, Zijinshan copper mine), Guangdong (*e.g.*, Lechang lead-zinc mine), Gansu (*e.g.*, Baiyin copper mine), Hunan (*e.g.*, Shaodong coal mine), Jiangxi (*e.g.*, Dexing/Yongping copper mines) and Shannxi (*e.g.*, Baihe pyrite mine) provinces, where the concentrations even exceed 1000 mg/L. The results shown in Fig. 4d suggest that Guangdong (*e.g.*,

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Yunfu pyrite mine), Gansu (*e.g.*, Baiyin copper mine), Inner Mongolia (*e.g.*, Bayan Obo pyrite mine), Jiangxi (*e.g.*, Dexing copper mine), Tibet (*e.g.*, Yulong copper mine) and Yunnan (*e.g.*, Bainiuchang polymetallic mine) provinces/autonomous regions are the severe pollution area of Mn. The presence of Fe and Mn in mine drainage primarily stems from ore composition and oxidation reactions. It is acknowledged that many metal ores naturally contain Fe- and Mn-bearing minerals, and when these minerals come into contact with mine water, some metal ions are dissolved, especially in areas with fast-flowing water. Moreover, Fe and Mn are easily oxidized under acidic conditions, which enhance their solubility and lead to higher concentrations.

3.2.4 Mining-affected water with high heavy metals

In terms of mining-affected water with high concentrations of HMs (*i.e.*, Cr, Ni, Cu, Zn, As, Cd, Hg, and Pb), the spatial hotspots are similar, covering the Yangtze River basin and the provinces of Fujian, Gansu, Guangdong, and Guangxi (Figs. S4). These provinces are recognized as key regions for non-ferrous metal production in China and play a vital role in the industry (Zhang et al., 2016). Particularly, in our study, the Baiyin copper mine site in Gansu province exhibits extreme contamination of Cu, Zn, Cd, Hg, and Pb, while the Bainiuchang polymetallic mine, situated in the southeast Yunnan metallogenic belt, has significant influences on Cr and As. In addition, the Zhongtiaoshan copper mining area has the highest level of Ni contamination in acid mine drainage, with a concentration of 15.0 mg/L.

In connection with the results summarized in Section 3.1, the top four HMs in acid water are Zn, Cu, Ni, and Pb, whereas the top four are Zn, Cu, Ni, and As in non-acid water. According to the review by Yin et al. (2018), the mineral composition of copper deposits is complex and includes associated minerals such as nickel, gold, and sulfur. Approximately 76% of the associated gold, 32.5% of the associated silver, and 76% of the sulfur originate from copper mines in China. This reason can explain the prevalence of Cu, Ni, and sulfate contamination hotspots in China's major copper production bases, *i.e.*, Jiangxi (Dexing/Yongping/Dongxiang, etc.), Tongling (Tongguanshan/Shizishan/Xinqiao, etc.), Daye (Tonglushan/Tongshankou, etc.), Zhongtiaoshan, Baiyin, and other copper bases (Chen et al., 2013). Besides, the Zn and Pb pollution levels are relatively higher in the Yunnan-Guizhou and Guangxi-Guangdong regions (Figs. S4d and S4h), where are abundantly occupied by the lead-zinc ores (*e.g.*, Dachang, Daxin, Wuxu, and Fankou lead-zinc mines, etc.), in which sphalerite (ZnS) is the principal mineral source of Zn and galena (PbS) is the main source of Pb (Blowes et al., 2014). Regarding As contamination in mining water, it has been demonstrated that pyrite can contain substantial amounts of As. Abraitis et al. (2004) and Blanchard et al. (2007) have mentioned that As can substitute for S in the pyrite structure, forming As-S dianion groups. The presence of As in pyrite increases its reactivity and accelerates its dissolution.

3.3 Risks of mining-affected water in China

Ingestion and dermal contact are the two main exposure pathways for residents (adults and children) in the mining areas The risks of exposure and health effects are compounded by the metal's persistence, mobility, and potential for accumulation in the environment. Among the measured metals in the study, Cr, Ni, As, Cd, and Pb are classified as carcinogens by the International Agency for Research on Cancer (IARC). In this section, the CRs of Cr, Cd, and As are assessed due to the lack of the carcinogenic slope factors for the other elements (Fig. 5). Additionally, Fe, Mn, Cr, Ni, Cu, Zn, As, Cd, and Pb are taken into consideration to calculate the cumulative values of the NCRs to the residents (Fig. 6). To highlight the human health risks posed

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by different types of mining-affected water in China, the risk assessment is categorized into two types: T1 and T1. T1 includes the mine drainage, mine water, and leachate water, which can be a significant threat to the surrounding water systems. T2 refers to the surface water and groundwater that have been affected by mining activities.

3.3.1 Carcinogenic risk of mining-affected water

CR or TCR values between 10^{-6} and 10^{-4} are considered acceptable (USEPA, 2004). As illustrated in Fig. 5, the median CR values for different population groups or water categories are generally in the order of As > Cd > Cr. Shi et al. (2018) elucidated that the CR values of diverse HMs in mining areas' soils also follow the aforementioned order. In the T1-type water, the median CR values are all below the upper limit of 10^{-4}, except for As for adults. The corresponding orders are As (1.98×10^{-4}) > Cd (6.40×10^{-5}) > Cr (5.39×10^{-5}) for adults and As (7.24×10^{-5}) > Cd (2.34×10^{-5}) > Cr (1.97×10^{-5}) for children, respectively. The CR values of T2-type water are generally lower than those of T1-type water, with median values of 5.28×10^{-5} (As), 3.14×10^{-5} (Cd), 2.13×10^{-5} (Cr) for adults and 1.93×10^{-5} (As), 1.15×10^{-5} (Cd), 7.82×10^{-6} (Cr) for children. Notably, the median TCR values for adults and children both exceed the upper acceptable limit in the mining areas examined in this study, reaching 3.02×10^{-4} and 1.10×10^{-4}, respectively. In connection with the results displayed in Fig. S5a, the mining areas with non-negligible CRs (TCR > 10^{-4}) account for 68.25 % of adults and 51.47% of children exposed to T1-type water, and 40.27% of adults and 23.31% of children exposed to T2-type water. In terms of spatial distribution (Fig. S6), the results show that TCR levels of T1-type water are unacceptable in 55.00% and 40.00% of the mining areas for adults and children, respectively. For T2-type water, the unacceptable CRs are observed in 51.52% and 29.29% of the mining areas for adults and children, respectively, emphasizing that these areas

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should serve as hotspots for further attention and management.

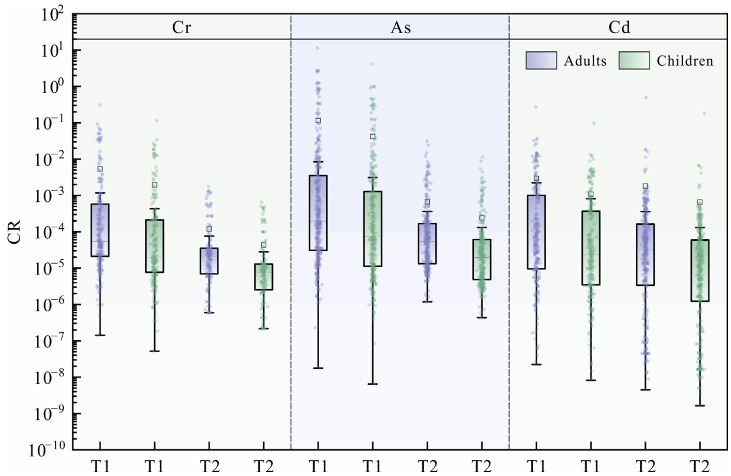

Figure 5. The CR values of Cr, As, and Cd in mining-affected water. T1 includes the mine drainage, mine water, and leachate water, while T2 indicates the mining-affected surface water and groundwater.

3.3.2 Non-carcinogenic risk of mining-affected water

For T1-type water (Figs. 6a and 6b) with high HQ values (HQ > 1), Mn, Fe, and As are the primary contributors, with medians of 6.84, 5.21, and 1.03 for adults, respectively, and 13.26, 9.63, and 1.88 for children, respectively. Additionally, the median values of Cd (1.66) and Pb (1.07) for children are also above the acceptable limit of 1. In T2-type water (Figs. 6c and 6d), the median HQ values for various metals, except for Mn, are all below the USEPA's acceptable threshold of 1 for both adults and children. The medians are in the order of Mn (1.950 for adults, 3.752 for children) > Cd (0.424, 0.812) > As (0.274, 0.500) > Pb (0.196, 0.357) > Cr (0.047, 0.099) > Zn (0.030, 0.055) > Cu (0.022, 0.040) > Fe (0.020, 0.036) > Ni (0.014, 0.025). The results suggest that children exhibit a heightened sensitivity to hazardous effects compared to adults, probably due to the more sensitive parameter settings used for children. In connection with the results displayed in

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Fig. S5b, the mining areas with high HI values (HI> 1) account for 88.35 % of adults and 91.90% of children exposed to T1-type water, and 55.75% of adults and 63.10% of children exposed to T2-type water. As depicted in Fig. S7, the southern regions are mainly occupied by the spatial hotspots of NCRs. For T1-type water, 89.04% of mining areas have unacceptable HI values for adults and 91.78% for children, while those for T2-type water are 68.07% and 80.67%.

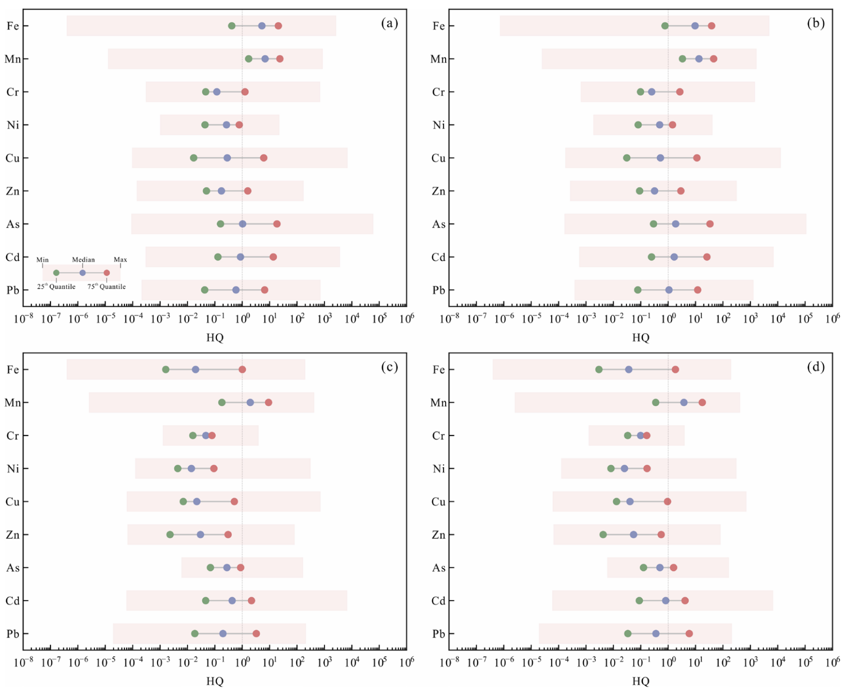

Figure 6. The HQ values of mining-affected water for (a) T1-Adult, (b) T1-Children, (c) T2-Adult, and (d) T2-Children, respectively. T1 includes the mine drainage, mine water, and leachate water, while T2 indicates the mining-affected surface water and groundwater.

4 Discussion

4.1 Effects of mining-affected water pollution in China

It is evident that acidic and metal-rich water is widespread in China, especially in the southern areas (see Fig. 4 above and Fig. S4), these contaminants pose significant risks to planetary health by degrading surface water and groundwater quality, destroying biodiversity, and threatening human well-being. Fig. 7 summarizes the key processes and adverse effects of mining-affected water pollution on the water subsystem, soil subsystem, and human health.

Water subsystem**:** As a vital component of various ecosystems, the water environment faces increasing challenges due to the presence of diverse mining-affected water pollution (as mentioned in Section 3.2). On the one hand, mining activities can contaminate groundwater, making it unfit for irrigation, drinking, and other purposes. It can be seen in Fig. 7 that during the active phase, acid mine drainage forms through a series of physical, chemical, and biological processes associated with the exposure of sulfide minerals to oxygen and water, resulting in the degradation of the groundwater environment (Acharya and Kharel, 2020). In terms of the abandoned period, the weathering products of exposed sulfides can serve as a source of acidity, sulfate, and dissolved metals, which may subsequently migrate and transform within the recovering groundwater (Blowes et al., 2014). On the other hand, acid mine drainage from active and abandoned mines also contaminates water bodies, lowering pH levels and destroying habitats for fish and other aquatic organisms (Ighalo et al., 2021). Toxic metals have the potential to accumulate in the food chain, especially in aquatic organisms, making them one of the most severe contaminants in surface water. Moreover, given that the metals are difficult to biodegrade, their presence has led to detrimental effects on the ecological balance of aquatic ecosystems (Gu et al., 2014; Cui et al., 2021).

Soil subsystem**:** HMs can enter the soil through mining-affected water runoff and tailings leaching, which have been increasingly detected in soil environments worldwide. Excessive HMs can adversely alter the physical and chemical properties of soil, threaten soil organisms (*e.g.*, by disrupting their physiological functions and behaviors), and reduce food production. Moreover, these contaminants can lead to shifts in microbial community structures, affecting the abundance and diversity of key microorganisms. However, the adverse effects of mining-affected water pollution on the soil subsystem are not the focus of our study, as Shi et al. (2023) and Yu et al. (2024) have provided a more comprehensive analysis of the pollution status, risks, and major influencing factors in coal and metal mines across China.

Human health: The results of the human risk assessment presented in Section 3.3 highlight that the CRs and NCRs are severe in China. Moreover, the metals' persistence, mobility, and potential for accumulation of the metals in the environment heighten the exposure risks, intensifying their impacts on health. The eight HMs discussed in this study are all toxic, and once they enter the human body, they can interact with DNA and enzymes, disrupting cellular, endocrine, immune, neurological, and reproductive systems (Shi et al., 2023; Meng et al., 2024). For example, various injuries linked to Cr exposure include nasal irritation and ulceration, skin irritation, and perforation of the eardrum. Acute exposure to Ni can result in damage to the kidneys, liver, and brain, whereas chronic exposure can cause tissue damage. Respiratory problems, dizziness, nausea, and diarrhea are common symptoms induced by elevated Cu concentrations (Gujre et al., 2021). Zn has a significant capacity for bioaccumulation, leading to increased health risks to the immune and nervous systems via the water-food chain (Cui et al., 2021). Chronic exposure to As is associated not only with skin lesions and skin cancer, but also with neurological, respiratory, cardiovascular, and developmental effects (Zhang et al., 2024). Poisoning with Cd can cause damage to the kidneys, bones, lungs, and liver, and can even lead to cancer. (Feng et al., 2022; Liu et al., 2024). Hg can lead to serious neurological disorders in both children and adults (Rui et al.,

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2017). Cardiovascular, central nervous system, kidney, and fertility problems are usually associated with Pb exposure (Shi et al., 2023). Furthermore, it has recently been demonstrated that Fe is linked to pathological disorders such as Alzheimer's and Parkinson's diseases (Sahoo and Sharma, 2023).

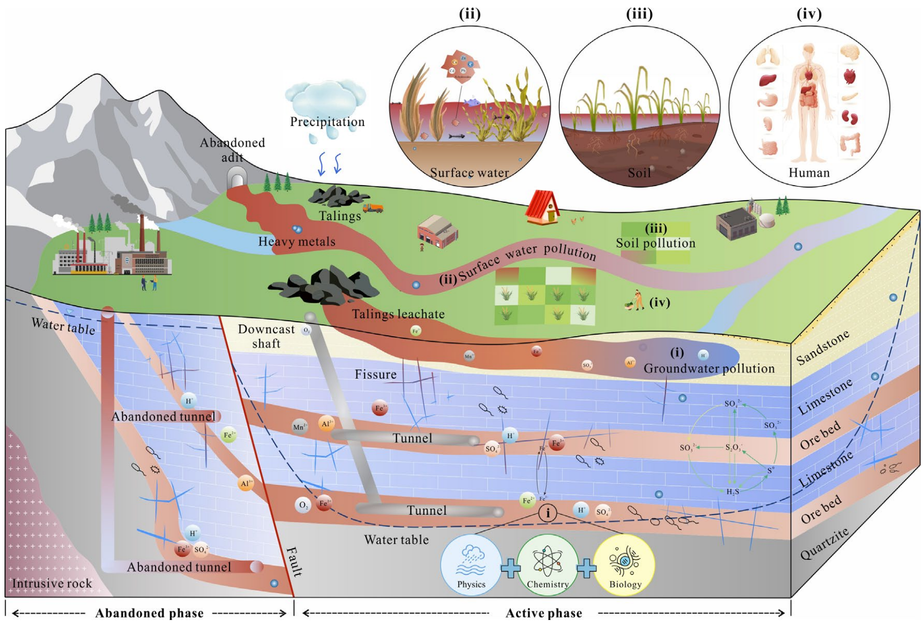

Figure 7. Conceptual model showing the processes and effects of mining-affected water pollution on (i) groundwater subsystem, (ii) surface water subsystem, (iii) soil subsystem, and (iv) human health.

4.2 *Implications for China's future differentiated management*

In the mining areas, the rising HMs contamination and potential health risks in surface water and groundwater call for targeted and forward-looking control strategies in China. In fact, mining regulations differ across provinces and countries, highlighting the need for site-specific frameworks and criteria. Although management may vary by location, priorities must include land use history, mine type, available technology, eco-hydrological conditions, socio-economic factors, multi-stakeholder cooperation, long-term monitoring, effective enforcement of effluent limits, and treatment standards (Acharya and Kharel, 2020). The differentiated management of coal mines and metal mines, active mines and abandoned mines are as follows:

***Coal mine and metal mine*:** The results imply that the water pollution status in metal mines is higher than in coal mines (Figs. 3 and S2). To some extent, policymakers should enhance their focus on regulating metal mining water contamination and devise more effective measures to reduce exposure and manage risks. The results presented in Section 3.1 imply that the characteristic contaminants in the acid water of coal mines are sulfate (with a median of 1381.59 mg/L), Fe (77.41 mg/L), and Mn (3.50 mg/L), while that of metal mines also include a variety of HMs, such as Zn (7.20 mg/L), Cu (1.73 mg/L), Ni (0.21 mg/L), Pb (0.15 mg/L) and so on. Consequently, water quality monitoring and water treatment technologies need to be tailored to address the specific characteristics of the different pollutants in both types of mines, including their sources, transport mechanisms, and environmental impacts. Some studies have demonstrated that precipitation and neutralization are commonly used methods in coal mines, while more complex technologies, such as ion exchange or membrane separation techniques, are required to remove HMs in metal mines.

***Active mine and abandoned mine*:** The differentiated management policies for active and abandoned mines aim to protect both the environment and public health across different stages of mining operations. In active mines, management policies should prioritize preventing and controlling the generation of mine drainage (with low pH, high sulfate and metals), including monitoring and managing potential pollution sources during ore extraction and transportation. Additionally, monitoring should be carried out more frequently to ensure a rapid response to any potential issues. Conversely, in abandoned mines, policies emphasize the remediation and long-

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term monitoring of mine water pollution that has already occurred, with a focus on assessing long-term variations in water quality and the effectiveness of remediation efforts over time. Furthermore, more detailed restoration strategies are needed to rebuild and stabilize ecosystems after mining operations.

Furthermore, sustainable management also plays a pivotal role in addressing the challenges of mining-related water pollution. Emphasis should be directed to multidisciplinary partnerships and cost-effective and eco-friendly treatments, especially integrated treatment approaches that take into account the synergy of source control and end-of-pipe treatment. These elements are crucial for better understanding the complexities of mine drainage, controlling water quality degradation, and minimizing socio-economic damage.

4.3 Reliability, limitations and prospects

In order to reveal the nationwide pollution status, spatial heterogeneity, health risks, and effects of mining-affected water in China, a total of 8433 water samples from 298 mines were integrated. Additionally, the combination of data mining and quality assessment was employed to enhance the reliability of the available data and build a high-quality database. However, there are still some non-negligible limitations or uncertainties in the study. On the one hand, the boundaries of mine sites are rarely clearly defined in the literature we collected, which means that the spatial heterogeneity of mining-affected water pollution cannot be accurately represented. On the other hand, the gridded data imply the southern regions, particularly the provinces/autonomous regions of Guizhou, Guangdong, Fujian, Jiangxi, Hunan, and Guangxi, are mining-affected water pollution hotspots. When compared with the reported sample sizes (Fig. S1), this suggests that these areas are generally high-sampling zones, which may potentially distort the representation of distribution.

Moreover, we cannot uncover the temporal evolution of mining-affected water pollution due to the varying time scales of the data. It's important to note that some gridded data only reflect the historical pollution status of a specific mine (*e.g.*, the Suichang gold mine and the coal mines in the Yudong River Basin) that has undergone successful ecological remediation and achieved good water quality levels after mining activities ceased. If research could be carried out in more coal and metal mines across China, more accurate levels of contamination would probably be found.

Future in-depth research could focus on (i) gathering globally reported data through deep mining and quality control and establishing a high-quality global database to better understand the characteristics of mining-affected water pollution worldwide; (ii) identifying the key factors that govern the transport and transformation of contaminants in surface water and groundwater systems, during active and abandoned periods, and in coal and metal mines; (iii) enhancing the sustainable development of coal and metal mines by AI-driven digital simulations and digital twins, which can provide data-driven insights, optimize remediation endeavors, and advocate proactive measures to safeguard the environment; and (iv) strengthening the studies on the synergistic measures (not only at small-scale experimental sites but also at the mine site scale) to alleviate multifaceted environmental challenges in the mining-affected water and achieve the development of green mining.

5 Conclusions

In this study, a nationwide mining area hydrochemical database, covering 26 provinces/autonomous regions in China, was established based on deep mining of massive reported data to elucidate the extent and spatial distribution pattern of national mining-affected water pollution, health risks of trace metals, and the adverse effects. The main conclusions are as follows:

- Compared to coal mines, most metal mines show elevated concentrations of sulfate (with a median of 2982.00 mg/L), Fe (113.77 mg/L), Mn (15.82 mg/L), Al (152.00 mg/L), and various HMs in mining-affected water, especially in the samples with a low pH (< 6.5).

- The spatial hotspots of mining-affected water pollution are mainly distributed in the southern regions, especially in Guizhou, Guangdong, Fujian, Jiangxi, Hunan, and Guangxi provinces/autonomous regions.

- About mining-affected surface water and groundwater, the mining areas with non-negligible CRs (TCR > 10^{-4}) account for 51.52% (for adults) and 29.29% (for children). Furthermore, 68.07% (for adults) and 80.67% (for children) of the mining sites confront with NCRs (HI > 1).

In summary, the current study provides unique insights into the nationwide water pollution posed by mining activities, and shrinks the knowledge gap on inadequate attention to comparative studies in previous studies, aiming at revealing the status, heterogeneity, risks, and impacts of water pollution in coal and metal mines. Moreover, the established high-quality database and the results obtained from the study are pragmatic in guiding policymakers to develop targeted and forward-looking water pollution control strategies for the development of green mining and human health protection.

Data availability. The detailed data information can be found in Table S1.

Author contributions. ZYY, JS, JFW, and YY conceptualized the manuscript and its scope. ZYY, DGL, and YYS collected the data. ZYY prepared the initial manuscript with contributions from all co-authors. JS, JFW, YY, YYS, and JCW revised the manuscript.

Competing interests. The authors declare that they have no conflict of interest.

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