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ORIGINAL PAPER

Assessment of environmental pollution and human health risks of mine tailings in soil: after dam failure of the Córrego do Feijão Mine (in Brumadinho, Brazil)

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Abstract The dam failure of the Córrego do Feijão Mine (CFM) located in Minas Gerais State, Brazil, killed at least 278 people. In addition, large extensions of aquatic and terrestrial ecosystems were destroyed, directly compromising the environmental and socioeconomic quality of the region. This study assessed the pollution and human health risks of soils impacted by the tailing spill of the CFM dam, along a sample perimeter of approximately 200 km. Based on potential ecological risk and pollution load indices, the enrichments of Cd, As, Hg, Cu, Pb and Ni in soils indicated that the Brumadinho, Mário Campos, Betim and São Joaquim de Bicas municipalities were the most affected areas by the broken dam. Restorative and reparative actions must be urgently carried out in these areas. For all contaminated areas, the children's group indicated an exacerbated propensity to the development of carcinogenic and non-carcinogenic diseases, mainly through the ingestion pathway. Toxicological risk assessments, including acute, chronic and genotoxic effects, on people living and working in mining areas should be a priority for public management and mining companies to ensure effective environmental measures that do not harm human health and well-being over time.

Keywords Cancerous diseases · Environmental tragedy · Geochemical indices · Metal(loid)s · Soil pollution

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Introduction

The growing worldwide demand for ore has led to intensive exploitation of these natural resources. In recent decades, this has triggered several environmental tragedies involving dam ruptures (Buch et al., 2020; Lyu et al., 2019; Rico et al., 2008). Dams are critical infrastructure in mining, and their failures can have catastrophic consequences for the environment and human life. Most collapses have been associated with exceeding the tailings storage capacity in dams, as well as inadequate management and lack of structural maintenance (CPRM 2019a; Buch et al., 2021, 2023). In Brazil, at least 126 mining dams are unstable due to the exceeding of their full capacity, making them vulnerable to failure in the coming years (Buch et al., 2020; Santos & Oliveira, 2021). This has been evidenced in the last 8 years, with the rupture of two iron ore tailing dams: Fundão (on November 5th 2015, in the Mariana Municipality) and Córrego do Feijão Mine-CFM (on January 25th 2019, in the Brumadinho Municipality) both situated in the Minas Gerais-MG State (Brazil) and belonging to VALE S.A. The CFM dam break released about 12 million m^3 of tailings into the environment, killing 278 people (six remain missing) and destroying entire villages, as well as areas of aquatic and terrestrial ecosystems. The mud of tailings traveled around 10 km until the Paraopeba River—a major tributary of the São Francisco River (CPRM, 2019a). In all, the mud traveled more than 300 km, affecting about 26 municipalities (18 of these directly affected). In terrestrial ecosystems, it has been estimated that approximately 1.8 million hectares were impacted by the dam collapse, with 300 hectares of native vegetation affected.

Over time, the persistence, reactivity and bioaccumulation of metals (predominant components in mining tailings) in the terrestrial environment result in ecotoxic effects on flora, fauna and human life, leading to physiological and metabolic stress, as well as imbalances in biodiversity and the performance of its functions within ecosystems (Buch et al., 2021, 2023). Trace elements such as arsenic (As), cadmium (Cd), chromium (Cr), nickel (Ni) and lead (Pb) may be very dangerous contamination components in terrestrial ecosystems, as they are not sensitive to any process of decomposition in soils and remain unaltered (Buch et al., 2023; Kumar et al., 2022a, 2022b). In addition, they are easily transported by chemical, physical, or biological processes to other environmental compartments (Ma et al., 2020). Mercury (Hg), a global pollutant, may change its chemical species through several processes, including the action of microorganisms, which can methylate this element and quickly lead to its bioaccumulation and biomagnification (Buch et al., 2020; Gupta et al., 2022). Only the metal contents of the soil surface horizons do not provide extensive indications of pollution. Geochemical indices may be enlightening when used to indicate anthropogenic interference in soil pollution (e.g., ore tailings deposition) (Barbieri et al., 2015; Islam et al., 2015).

Although indices of human health risks are a basic approach to assessment, they are widely used, and the main global guidelines recommend their applicability for prospecting the potential risks of different materials or chemical substances deposited in ecosystems (EPA 2005). The susceptibility of early-life exposure to carcinogens for target groups can help guide future decision-making in compliance with environmental measures to mitigate or eliminate such hazards to human health (EPA, 2005). Thus, this study assessed the pollution and human health risks of soils impacted by the tailing spill of the CFM dam, along a sample perimeter of approximately 200 km. This evidence will enable the identification of critical areas of pollution, prioritization of remediation actions, and support decision-making in programs for public health and environmental preservation and recovery.

Material and methods

Soil samples characterization

The study areas covered a sampling gradient of approximately 200 km along the Paraopeba River Basin, including sites that were completely affected by the mining mudflow from the CFM dam rupture and partially affected (contaminated after the Paraopeba River overflowed due to intense rainy periods in the region). Riparian soil samples were collected from 11 study areas: Dam 1 and 2; Pinheiros (Pi), Alberto Flores (AF), Mário Campos (MC), Betim (B), São Joaquim de Bicas (SJB), Florestal (F), São José da Varginha (SJV), Paraopeba (Pa), and Pompéu (P). Additionally, forest soil samples (from non-impacted sites by mining activities) belonging to the Cerrado and Atlantic Forest biomes were collected in two reference areas: (i) Parque Estadual da Serra do Rola-Moça (PESRM) located in the Nova Lima municipality in MG State (24 km from the dam rupture) and (ii) Parque Estadual do Sumidouro (PES) situated in an Environmental Conservation Unit at Pedro Leopoldo and Lagoa Santa Municipalities in MG State (130 km from the dam rupture). Descriptions of these areas and their distances from the CFM dam are found in Table S1. The physical-chemical analyses were performed according to EMBRAPA (2011) on ten subsamples (0–20 cm depth) collected for each area, composing a representative grid (20 × 20 m). The soil sampling occurred in February 2022. The concentrations of Al, As, Cd, Cu, Cr, Fe, Pb, Mn, Ni, Sr, Ti and Zn were estimated using Method 3051 (USEPA, 2007) and analyzed by inductively coupled plasma mass spectrometry (ICP-MS) according to USEPA (2001), as well as by atomic absorption spectrometry using cold vapor atomization for Hg according to USEPA (1986). Quality control was established in accordance with reference material certified São Joaquim 2709, provided by the National Institute of Standards and Technology-NIST (US Department of Commerce). The average recovery rates for metals ranged from 85 to 97%.

Geochemical indices

Data from the study areas were used to calculate the geochemical indices. Background values were obtained from the median values of the Paraopeba River Basin, based on a data survey carried out by CPRM (Portuguese acronym for Company of Research of Mineral Resources—Geological Survey of Brazil) in monitoring years (since 2009) that preceded the dam rupture (CPRM). Baseline values were based on actual trace metal concentrations from the two reference areas, which are more dependent of the parent geological/source material.

The enrichment factor (EF) was estimated according to Kemp and Thomas (1976) (Eq. 1) to discriminate natural and anthropogenic sources of metals in soils. Aluminum was used for normalization due to its lithogenic and conservative features.

$$\mathrm{EF} = \frac{\left(M_i/\mathrm{Al}\right)_{\mathrm{sample}}}{\left(M_i/\mathrm{Al}\right)_{\mathrm{Background\ or\ Baseline\ values}}} \tag{1}$$

where Mi is the interest metal.

Pollution load index (PLI) was calculated from the individual metal values of contamination factor (CF) according to Eqs. (2 and 3) described in Qing et al. (2015).

$$\mathrm{CF} = \frac{C_i}{B_i} \tag{2}$$

$$\mathrm{PLI} = \left(\mathrm{CF}_1 \times \mathrm{CF}_2 \times \mathrm{CF}_3 \times \ldots\ldots\ldots \mathrm{CF}_n\right)^{\frac{1}{n}} \tag{3}$$

The Potential Ecological Risk index (PER) was estimated according to Hakanson et al. (1980), based on ecological risk (ER), Eqs. 4 and 5.

$$\mathrm{ER} = \mathrm{TR} \times \mathrm{CF} \tag{4}$$

$$\mathrm{PER} = \sum \mathrm{ER} \tag{5}$$

where TR is the "toxic-response" factor for a given metal (As = 10, Cd = 30, Cr = 2, Cu = 5, Hg = 40, Ni = 5, Pb = 5 and Zn = 1; for Al, Fe and Mn, there are no TR values in the literature, thus not allowing the calculation of ER; CF is the contamination factor.

Human health risk assessment

Due to the contact of target groups (adults and children) with soil contaminated by mine tailings, for each entry situation of metal(loid) (through three exposure pathways: ingestion, dermal contact and inhalation), the absorbed dose was calculated. Average daily intake (ADI) of toxic metals via ingestion (ADIing), inhalation (ADIinh) and dermal contact (ADIderm) for target groups were based on values from the guidelines and Exposure Factors Handbook of US Environmental Protection Agency (USEPA, 1989, 1997, 2001, 2002 and 2011), which were described in Table S2 and determinate by Eqs. 6 to 8

$$\mathrm{ADI}_{\mathrm{ing}} = \frac{\mathrm{CS} \times \mathrm{CoF} \times \mathrm{IR}s \times ExF \times \mathrm{ED}}{\mathrm{BW} \times \mathrm{AT}} \tag{6}$$

where CS is the chemical concentration in a particular exposure medium (mg kg^{-1}), CoF is the conversion factor; IR is the ingestion rate (mg day^{-1}), ExF is the exposure frequency (day per year^{-1}), ED is the exposure duration (year), BW is the body weight of the exposed individual (kg), and AT is the time period over which the dose is averaged (day).

$$\mathrm{ADI}_{\mathrm{in}h} = \frac{\mathrm{CS} \times \mathrm{ExF} \times \mathrm{ED}}{\mathrm{PEF} \times \mathrm{BW} \times \mathrm{AT}} \tag{7}$$

where PEF is the particle emission factor (m^{3}kg^{-1}).

$$\mathrm{ADI}_{\mathrm{Dermal}} = \frac{\mathrm{CS} \times \mathrm{SA} \times \mathrm{AF} \times \mathrm{CoF} \times \mathrm{ABS} \times \mathrm{ExF} \times \mathrm{ED}}{\mathrm{BW} \times \mathrm{AT}} \tag{8}$$

where SA is the exposed skin surface area (cm^2), AF is the adherence factor (mg/cm^2-day), and ABS is the dermal absorption factor (unitless).

The dose absorbed by a person represents the amount of metal(loid) that can affect human health; correlating the human body weight and exposure time, which is based on available toxicological data (Cocârţã et al., 2016).

Non-carcinogenic risk assessment

Non-carcinogenic hazards are typically characterized by the hazard quotient (HQ), according to Eq. 9 (USEPA, 1989):

$$\mathrm{HQ} = \frac{\mathrm{ADI}_i}{\mathrm{RfD}} \tag{9}$$

where RfD (mg kg^{-1} day^{-1}) is the maximum daily dose of a metal from a specific exposure pathway, for both adults and children.

The hazard index (HI) is the sum of HQs and means of the total risk of non-carcinogenic elements via three exposure pathways for a single element, Eq. 10.

$$\mathrm{HI} = \sum \mathrm{HQ} = \sum \frac{\mathrm{ADI}_i}{\mathrm{RfD}} \tag{10}$$

Carcinogenic risk assessment

Carcinogenic risks are estimated by calculating the probability of an individual developing cancer. The slope factor (SF) converts the estimated daily intake of a toxin averaged over a lifetime of exposure directly to the incremental risk of an individual developing cancer (USEPA, 1989), Eqs. 11 and 12.

$$\mathrm{RI} = \mathrm{ADI}_i \times \mathrm{SF}_i \tag{11}$$

$$\mathrm{TRI} = \sum \mathrm{RI} = \sum \mathrm{ADI}_i \times \mathrm{SF}_i \tag{12}$$

where RI is the carcinogenic risk; TRI is the total carcinogenic risk; and SF is the carcinogenicity slope factor over a lifetime (mg kg^{-1} day^{-1}). Carcinogenic risk values ranging from 1×10^{-6} to 1×10^{-4} are defined as an acceptable risk for human health (USEPA, 2001).

Statistical analysis

Analysis of variance (ANOVA) followed by Dunnett's post hoc test ($p < 0.05$) was applied using the software Minitab®17.1.0. Data homoscedasticity and normality were checked by Bartlett's and Kolmogorov–Smirnov's tests, respectively ($p < 0.05$). The Spearman correlation and principal component analyses (PCA) were applied to integrate concentrations of 13 metal(loid)s and environmental and human variables for soil samples from the 13 study areas using multivariate analysis through Canoco 5 software. Matrix cluster analysis was used to classify the samples into distinct groups by integrating all the data, which was generated by statistical packages of open-source software R.

Results

Physicochemical soil parameters

Based on granulometric composition, the texture varied from sandy loam (in the areas of Dam 1 and 2, Pi, AF, MC, B, SJB, F and Pa) and silt loam (in the SJV and P areas). In the reference areas (PESRM and PES), the sandy clay texture prevailed (Table 1). The areas impacted by CFM tailings indicated an acid pH (ranging from 3.5 to 4.2). The lowest values were noted in areas closest to the broken dam. The reference areas showed the highest pH values (4.9 ± 0.4 and 5.7 ± 1.0 for the PESRM and PES, respectively). For these areas, there were also the lowest average values of soil bulk densities (= 1.35 ± 0.3 g cm^{-3} for the PESRM area and of 1.30 ± 0.4 g cm^{-3} for the PES). In impacted areas, the average values of bulk density ranged from 2.05 to 4.21 g cm^{-3}

Table 1 Physicochemical soil parameters from 13 areas, after collapse of Córrego do Feijão Mine dam in Brumadinho Municipality (Minas Gerais State. Brazil)

Areas	Sand %	Silt %	Clay %	BD g/dm^3	pH
Dam 1	66 ± 10.0	18 ± 4.2	16 ± 6.9	3.07 ± 0.5	3.6 ± 0.3
Dam 2	64 ± 8.2	22 ± 2.9	14 ± 5.8	4.21 ± 0.9	3.5 ± 0.8
Pi	62 ± 11.1	19 ± 6.7	19 ± 5.2	3.41 ± 1.7	3.8 ± 0.6
AF	55 ± 7.8	22 ± 4.5	23 ± 6.3	3.27 ± 0.5	4.0 ± 0.8
MC	52 ± 8.6	23 ± 3.9	25 ± 5.5	3.12 ± 0.3	3.5 ± 0.4
B	58 ± 7.3	20 ± 4.2	22 ± 6.4	3.76 ± 0.2	3.2 ± 0.2
SJB	57 ± 8.2	23 ± 6.2	20 ± 6.7	2.74 ± 0.7	4.1 ± 0.2
F	56 ± 8.8	19 ± 4.0	25 ± 4.9	2.41 ± 0.4	4.2 ± 0.2
SJV	44 ± 7.7	24 ± 3.4	32 ± 4.4	2.59 ± 0.9	3.9 ± 0.5
Pa	54 ± 6.4	29 ± 6.5	17 ± 3.8	2.31 ± 0.9	3.7 ± 0.8
P	47 ± 7.9	16 ± 3.9	37 ± 4.9	2.05 ± 0.8	3.8 ± 0.6
PESRM	39 ± 5.5	17 ± 4.7	44 ± 7.6	1.35 ± 0.3	4.9 ± 0.4
PES	29 ± 6.2	26 ± 2.6	45 ± 7.2	1.30 ± 0.4	5.7 ± 1.0

The values are shown as means ($n = 6$). Acronyms: *BD* bulk density; Dam 1 and 2 refers to dam areas; *Pi* Pinheiros; *AF* Alberto Flores; *MC* Mário Campos; *B* Betim; *SJB* São Joaquim de Bicas; *F* Florestal; *SJV* São José da Varginha; *Pa* Paraopeba; *P* Pompeu; *PESRM* Parque Estadual da Serra do Rola-Moça; *PES* Parque Estadual do Sumidouro

(Table 1). The highest bulk densities were related to areas nearby CFM dam areas.

Metal concentrations are shown in Fig. 1. In the affected areas, the order of metal concentrations (from high to low) was Fe > Mn > Al > Cu > Zn > Cr > Pb > Ni > As > Sr > Cd > Hg. The highest values were found in areas of Dam 1 and 2; Pi, AF, MC, B and SJB, whose municipalities are located closer to the broken dam areas. In reference areas, none of the metal(loid)s exceeded the guideline values from Brazilian normative (COPAM-MG and CONAMA) and for others environmental directives recognized worldwide for soil quality assessment. For more comparative details, see Table S2. In impacted areas, the concentrations of As Cd, Cu, Hg, Pb and Zn exceeded the maximum allowable contents of metals in soils defined by COPAM-MG 166 (2011). According to CONAMA 420 (2009), all the areas contaminated by CFM tailings exceeded the limiting values of Cu and almost all areas exceeded the As levels (except the F, SJV, Pa and P areas). However, for Cd, Hg, Pb (except the Dam 1 and Dam 2 areas) and Zn concentrations, no area indicated an exceedance. For both Brazilian normatives (COPAM-MG and CONAMA), the Cr contents were higher than threshold values only in the Dam 1, Dam 2 and B areas. Ni exceeding values were noted in the Dam 1, Dam 2, Pi and AF areas (according to CONAMA 420, 2009) and Dam 1, Dam 2, Pi, AF, MC, B and SJB areas (based on COPAM-MG 166, 2011).

Geochemical indices

The enrichment factor was classified as follows: < 1—background concentration; 1 to 2—depletion to minimal enrichment; 2 to 5—moderate enrichment; 5 to 20—substantial enrichment; 20 to 40—very high enrichment; and > 40—extremely high enrichment (Bam et al., 2011). Overall, the EF values were higher when determined using the baseline values from the reference areas (especially for As, Cd, Cu, Hg, Ni and Pb contents), showing slightly higher EF on the basis of PES values, except for Mn, Sr and Zn contents, which were higher when evaluated by background values (Fig. 2). From the PES baseline values, the Cd and Pb concentrations indicated very high enrichment values. Moderate metal enrichment factors were observed for Cr, Fe and Mn concentrations from the baseline values (PES and PESRN). The EFs of As, Cd and Hg determined from the background values showed substantial enrichment levels, whereas Cu, Mn, Ni, Pb and Zn presented moderate enrichment; Cr presented depletion to minimal enrichment; and Fe contents suggest background concentration, possibly from parental material. In general, the order of average EF values of metals was Cd > Pb > Hg > As > Cu > Cr > Ni > Mn > Fe > Zn.

According to contamination degree, the PLI may be classified as unpolluted (PLI ≤ 1), unpolluted to moderately polluted (1 ≤ PLI ≤ 2), moderately polluted (2 ≤ PLI ≤ 3), moderately to highly polluted (3 ≤ PLI ≤ 4), highly polluted (4 ≤ PLI ≤ 5), or very highly polluted (PLI > 5) (Tomlinson et al., 1980, Chen et al., 2015). The PLIs calculated for metal(loid)s from background values (3.27 to 5.85) showed the lowest values when compared to those from the baseline values, which ranged from 6.71 to 12.6 (Fig. 3). However, all investigated areas were classified as highly polluted environments.

The highest ecological risk values were noted for Cd and Hg (Fig. 4). The sequence of metals (from highest to lowest) was: Cd > Hg > As > Pb > Ni > Cu > Cr > Zn (Fig. 4). PER values were

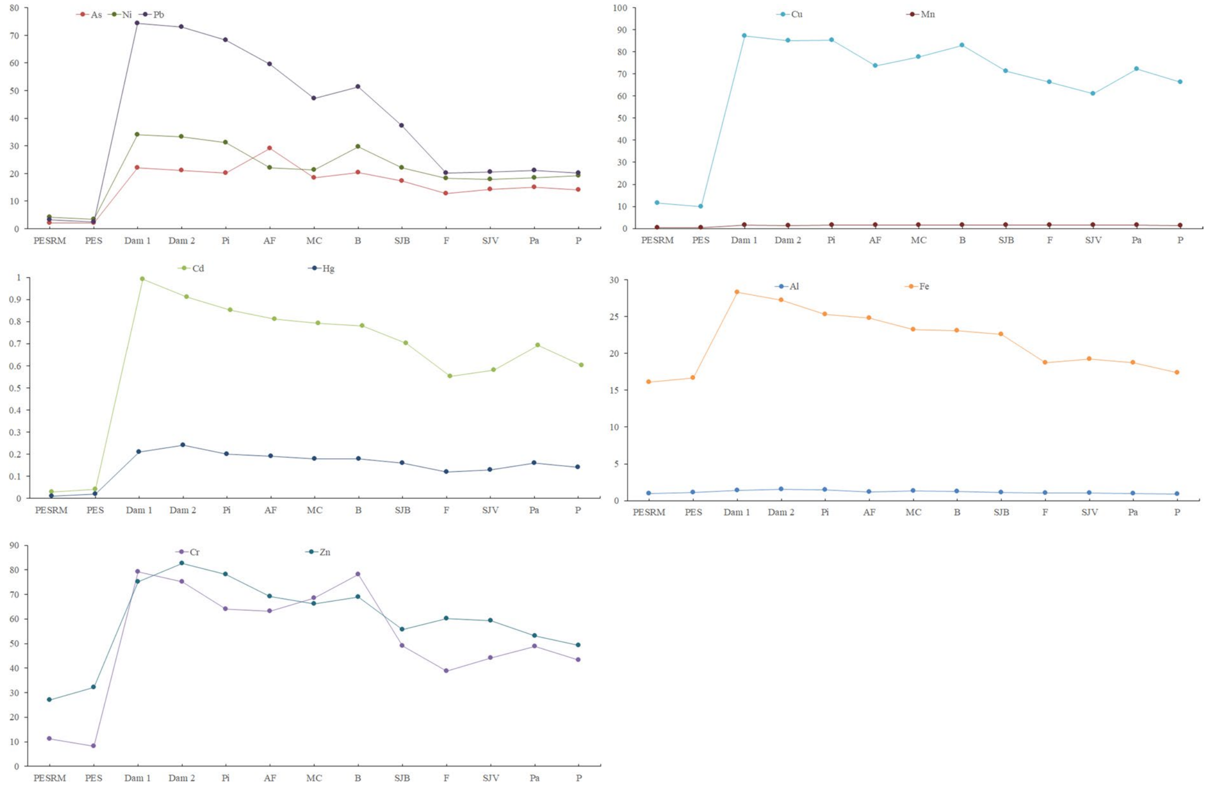

Fig. 1 Total metal(loid) concentrations ($n=6$) from the 13 sampling areas. Dam 1 and 2 refers to dam áreas; *Pi* Pinheiros; *AF* Alberto Flores; *MC* Mário Campos; *B* Betim; *SJB* São Joaquim de Bicas; *F* Florestal; *SJV* São José da Varginha; *Pa* Paraopeba; *P* Pompeu; *PESRM* Parque Estadual da Serra do Rola-Moça; *PES* Parque Estadual do Sumidouro

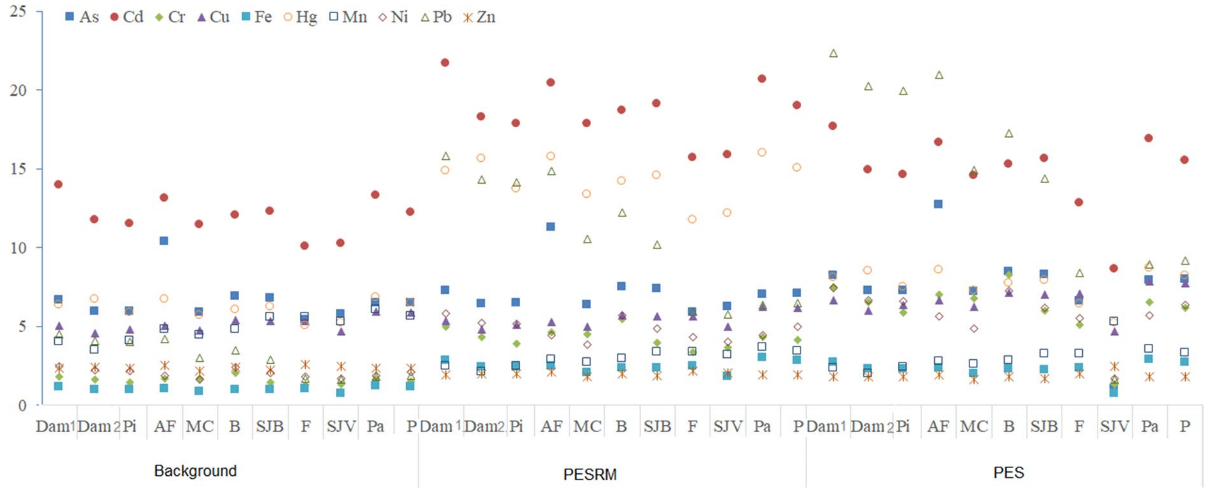

Fig. 2 Enrichment factors of metal(loid)s in soils, based on background and baseline values. Dam 1 and 2 refers to dam áreas; *Pi* Pinheiros; *AF* Alberto Flores; *MC* Mário Campos; *B* Betim; *SJB* São Joaquim de Bicas; *F* Florestal; *SJV* São José da Varginha; *Pa* Paraopeba; *P* Pompeu; *PESRM* Parque Estadual da Serra do Rola-Moça; *PES* Parque Estadual do Sumidouro

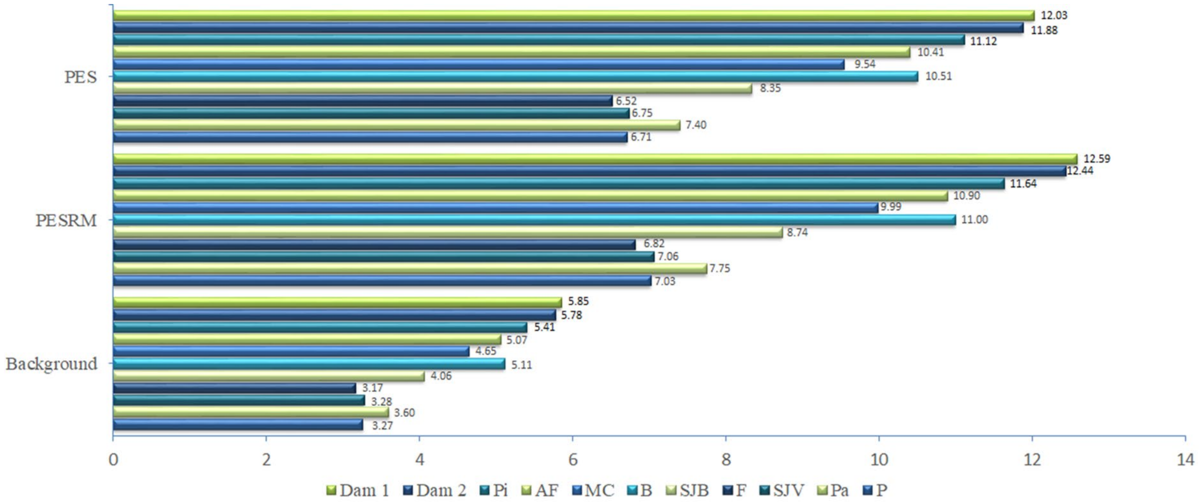

Fig. 3 Pollution load index from the 13 study areas. Dam 1 and 2 refers to dam áreas; *Pi* Pinheiros; *AF* Alberto Flores; *MC* Mário Campos; *B* Betim; *SJB* São Joaquim de Bicas; *F* Florestal; *SJV* São José da Varginha; *Pa* Paraopeba; *P* Pompeu; *PESRM* Parque Estadual da Serra do Rola-Moça; *PES* Parque Estadual do Sumidouro

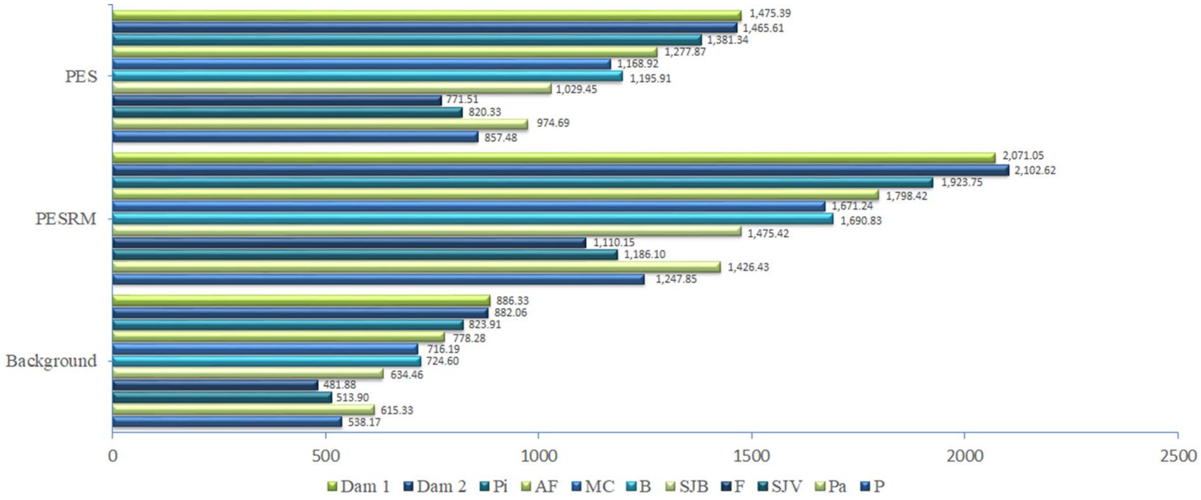

Fig. 4 Index of potential ecological risk from the 13 study areas. Dam 1 and 2 refers to dam áreas; *Pi* Pinheiros; *AF* Alberto Flores; *MC* Mário Campos; *B* Betim; *SJB* São Joaquim de Bicas; *F* Florestal; *SJV* São José da Varginha; *Pa* Paraopeba; *P* Pompeu; *PESRM* Parque Estadual da Serra do Rola-Moça; *PES* Parque Estadual do Sumidouro

considered as: low or slight ($PER < 150$), moderate ($150 \leq PER < 300$), considerable or strong ($300 \leq PER < 600$) and very high or very strong ($PER \geq 600$). PER values based on PES and PESRM baseline values ranged from 772 to 1475 and 1110 to 2103, respectively (Fig. 4). Hence, a very strong level of potential ecological risk may be attributed to all areas impacted by CFM tailings. In comparison with those found by background values, the PERs ranged from 481.9 to 886.3; the SJB, Flo, SJV, Pa and P areas indicated a strong PER; however, in the Dam 1 and 2, Pi, AF, MC and B areas were observed very high PER values ($PER > 600$).

Human health risk assessment

Non-carcinogenic risk (HQ) values greater than 1 means that the exposed population is likely to experience adverse effects related to metals (Faiz et al., 2012; Qing et al., 2015). For all metal(loid)s from all areas (except for reference areas), the HQ values of a specific via exposure pathway were less than 1, in groups of adults and children. In the children's group, for all affected areas, the HQ values of Pb (1.355 to 5.481) exceeded the limits, indicating a high predisposition to non-carcinogenic effects. In the adult's group, the HQ values ranged from 0.00024 (Zn) to 0.805 (Pb) and from 0.0016 (Zn) to 5.481 (Pb) in the children's group (Table 2). The order of the HQs observed for both groups was: Pb > As > Cr > Ni > Hg > Cd > Zn. The highest HQ rates decreased as the areas moved away from the dam, both for adults and children. Considering the three exposure pathways together (through hazard index -HI values, which is the HQ sum) for all metal(loid)s from all affected areas, the children's group presented high health risks to non-carcinogenic effects (HI > 1; from 1.91 to 6.49). HIs in adults did not indicate a propensity to non-carcinogenic diseases (less than 1; from 0.29 to 0.95). In addition, a decrease in the HI values can be observed as the affected areas move away from the dam failure areas (Table 2). In the reference areas, HI values were 0.04 (PES) and 0.05 (PESRM) for adults and 0.028 (PES) and 0.035 (PESRM) for children (Fig. 5). On average, for most of the metals, the main exposure pathway that highest contributed to the hazard quotient (HQ) was the ingestion (94%) followed by the dermal exposure representing 4.5% and the inhalation (less than 1%).

Values of RI lower than 1×10^{-6} suggest there is no significant carcinogenic risk, while RI values $> 1\times10^{-4}$ imply serious and unacceptable carcinogenic risks. To simplify the understanding of this index in the study areas, values can be seen in Table 3. RI values for adults ranged from 2.5×10^{-7} (Pb) to 1.4×10^{-4} (Ni). For children, RI values ranged from 1.7×10^{-6} (Pb) to 6.0×10^{-4} (Ni). Adult TRIs ranged from 1.7×10^{-4} to 3.2×10^{-4}. In children, the TRI values ranged from 7.5×10^{-4} to 1.5×10^{-3} (Fig. 5). The ingestion was the main exposure route (60% and 73% for adults and children, respectively) to the carcinogenic risks, whereas the dermal exposure represented an average of 40% and 27% for adults and children, respectively, and inhalation about less than 1% for both. All study areas except the reference areas presented TRI of metal(loid)s greater than 1×10^{-4}

Table 2 Non-carcinogenic risks of metals for adults and children

Study areas	Adult								Children							
	HQ							HI	HQ							HI
	As	Cd	Cr	Hg	Ni	Pb	Zn		As	Cd	Cr	Hg	Ni	Pb	Zn	
Dam 1	0.105	0.001	0.038	0.001	0.002	0.805	0.00037	0.95	0.715	0.010	0.256	0.007	0.017	5.481	0.003	6.49
Dam 2	0.137	0.001	0.036	0.001	0.002	0.727	0.00040	0.91	0.936	0.009	0.247	0.008	0.017	4.952	0.003	6.17
Pi	0.128	0.001	0.034	0.001	0.002	0.775	0.00037	0.94	0.870	0.008	0.228	0.006	0.015	5.274	0.003	6.40
AF	0.121	0.001	0.033	0.001	0.002	0.605	0.00031	0.76	0.824	0.007	0.222	0.005	0.014	4.115	0.002	5.19
MC	0.081	0.001	0.031	0.001	0.001	0.411	0.00027	0.53	0.554	0.007	0.209	0.006	0.010	2.795	0.002	3.58
B	0.094	0.001	0.035	0.001	0.002	0.495	0.00038	0.63	0.638	0.005	0.241	0.004	0.014	3.371	0.003	4.28
SJB	0.077	0.001	0.023	0.001	0.002	0.299	0.00025	0.40	0.522	0.005	0.153	0.005	0.010	2.035	0.002	2.73
F	0.059	0.001	0.018	0.001	0.001	0.201	0.00029	0.28	0.401	0.006	0.122	0.004	0.009	1.368	0.002	1.91
SJV	0.068	0.001	0.020	0.001	0.001	0.198	0.00026	0.29	0.461	0.005	0.136	0.004	0.009	1.350	0.002	1.97
Pa	0.070	0.001	0.023	0.001	0.001	0.197	0.00025	0.29	0.475	0.005	0.159	0.006	0.009	1.343	0.002	2.00
P	0.066	0.001	0.019	0.001	0.001	0.199	0.00024	0.29	0.448	0.004	0.127	0.005	0.009	1.355	0.002	1.95
PESRM	0.009	0.000	0.006	0.000	0.000	0.035	0.00013	0.05	0.064	0.000	0.041	0.001	0.002	0.237	0.001	0.35
PES	0.009	0.000	0.006	0.000	0.000	0.024	0.00014	0.04	0.063	0.000	0.043	0.000	0.002	0.164	0.001	0.27

HQ is the hazard quotient; *HI* is the hazard index representing the sum of HQs; *Dam 1 and 2* refers to dam áreas; *Pi* Pinheiros; *AF* Alberto Flores; *MC* Mário Campos; *B* Betim; *SJB* São Joaquim de Bicas; *F* Florestal; *SJV* São José da Varginha; *Pa* Paraopeba; *P* Pompeu; *PESRM* Parque Estadual da Serra do Rola-Moça; *PES* Parque Estadual do Sumidouro

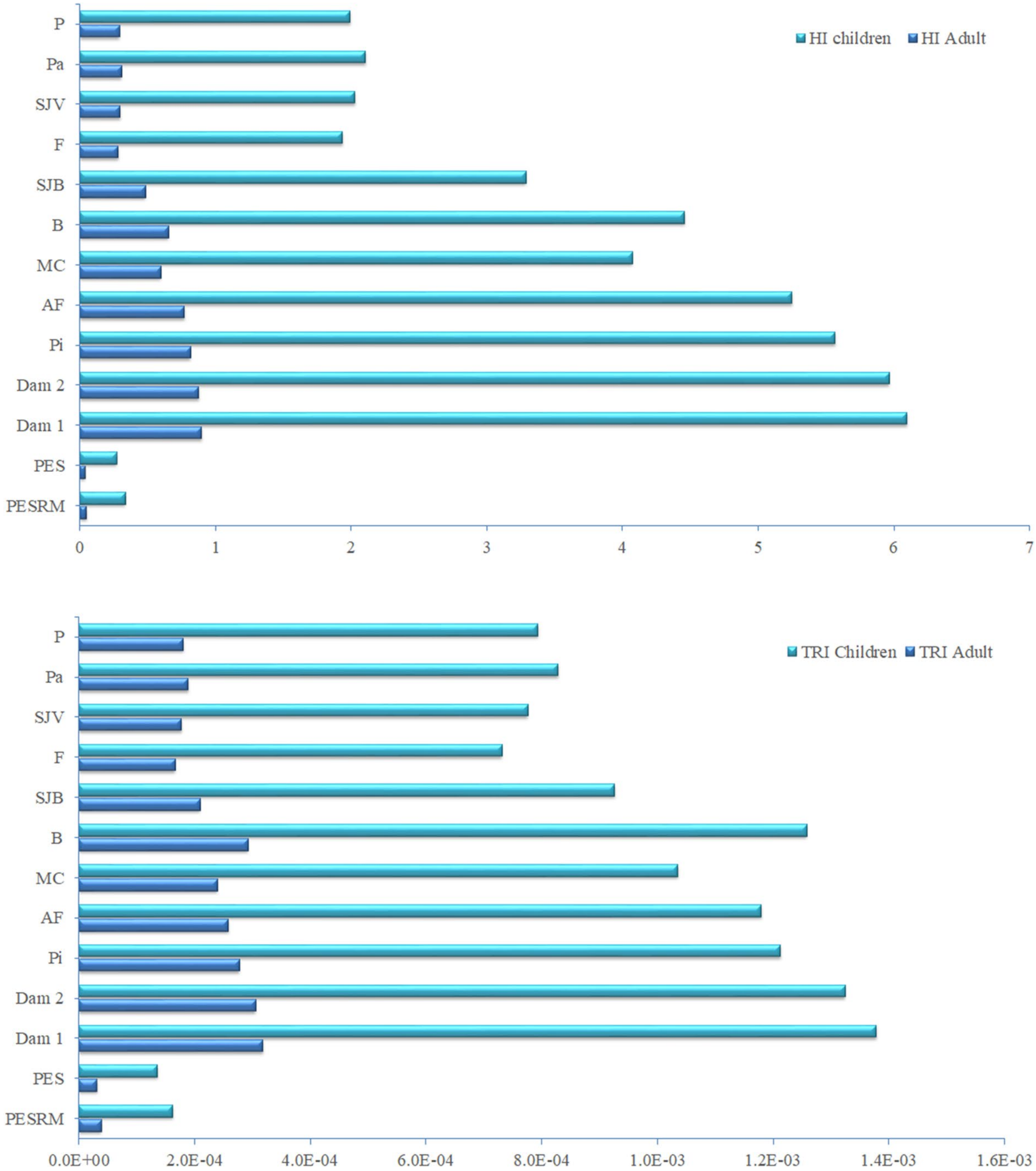

Fig. 5 Hazard index (HI) for the non-carcinogenic risks and the total carcinogenic risk (TRI) of metals, in adults and children. Dam 1 and 2 refers to dam áreas; *Pi* Pinheiros; *AF* Alberto Flores; *MC* Mário Campos; *B* Betim; *SJB* São Joaquim de Bicas; *F* Florestal; *SJV* São José da Varginha; *Pa* Paraopeba; *P* Pompeu; *PESRM* Parque Estadual da Serra do Rola-Moça; *PES* Parque Estadual do Sumidouro

Table 3 Carcinogenic risks of metals for adults and children

Study areas	RI Adult					TRI Adult	RI Children					TRI Children
	As	Cd	Cr	Ni	Pb		As	Cd	Cr	Ni	Pb	
Dam 1	4.9×10^{-5}	9.0×10^{-6}	1.2×10^{-4}	1.4×10^{-4}	9.9×10^{-7}	3.2×10^{-4}	3.2×10^{-4}	6.1×10^{-5}	4.2×10^{-4}	6.0×10^{-4}	6.8×10^{-6}	1.4×10^{-3}
Dam 2	6.4×10^{-5}	8.6×10^{-6}	1.2×10^{-4}	1.4×10^{-4}	9.0×10^{-7}	3.3×10^{-4}	4.2×10^{-4}	5.9×10^{-5}	4.0×10^{-4}	6.0×10^{-4}	6.1×10^{-6}	1.5×10^{-3}
Pi	6.0×10^{-5}	7.0×10^{-6}	1.1×10^{-4}	1.3×10^{-4}	9.5×10^{-7}	3.1×10^{-4}	3.9×10^{-4}	4.8×10^{-5}	3.7×10^{-4}	5.5×10^{-4}	6.5×10^{-6}	1.4×10^{-3}
AF	5.7×10^{-5}	6.7×10^{-6}	1.1×10^{-4}	1.2×10^{-4}	7.4×10^{-7}	2.9×10^{-4}	3.7×10^{-4}	4.5×10^{-5}	3.6×10^{-4}	5.1×10^{-4}	5.1×10^{-6}	1.3×10^{-3}
MC	3.8×10^{-5}	6.1×10^{-6}	9.9×10^{-5}	8.7×10^{-5}	5.1×10^{-7}	2.3×10^{-4}	2.5×10^{-4}	4.2×10^{-5}	3.4×10^{-4}	3.6×10^{-4}	3.4×10^{-6}	1.0×10^{-3}
B	4.4×10^{-5}	4.6×10^{-6}	1.1×10^{-4}	1.2×10^{-4}	6.1×10^{-7}	2.9×10^{-4}	2.9×10^{-4}	3.1×10^{-5}	3.9×10^{-4}	5.2×10^{-4}	4.2×10^{-6}	1.2×10^{-3}
SJB	3.6×10^{-5}	4.5×10^{-6}	7.3×10^{-5}	8.9×10^{-5}	3.7×10^{-7}	2.0×10^{-4}	2.4×10^{-4}	3.1×10^{-5}	2.5×10^{-4}	3.7×10^{-4}	2.5×10^{-6}	8.9×10^{-4}
F	2.8×10^{-5}	5.2×10^{-6}	5.8×10^{-5}	8.0×10^{-5}	2.5×10^{-7}	1.7×10^{-4}	1.8×10^{-4}	3.5×10^{-5}	2.0×10^{-4}	3.4×10^{-4}	1.7×10^{-6}	7.5×10^{-4}
SJV	3.2×10^{-5}	4.3×10^{-6}	6.5×10^{-5}	7.7×10^{-5}	2.4×10^{-7}	1.8×10^{-4}	2.1×10^{-4}	2.9×10^{-5}	2.2×10^{-4}	3.2×10^{-4}	1.7×10^{-6}	7.8×10^{-4}
Pa	3.3×10^{-5}	4.2×10^{-6}	7.6×10^{-5}	8.1×10^{-5}	2.4×10^{-7}	1.9×10^{-5}	2.1×10^{-4}	2.8×10^{-5}	2.6×10^{-4}	3.4×10^{-4}	1.7×10^{-6}	8.4×10^{-4}
P	3.1×10^{-5}	4.1×10^{-6}	6.1×10^{-5}	7.6×10^{-5}	2.5×10^{-8}	1.7×10^{-5}	2.0×10^{-4}	2.8×10^{-5}	2.1×10^{-4}	3.2×10^{-4}	1.7×10^{-6}	7.6×10^{-4}
PESRM	4.4×10^{-6}	3.6×10^{-7}	1.9×10^{-5}	1.6×10^{-5}	4.3×10^{-8}	4.1×10^{-5}	2.9×10^{-5}	2.4×10^{-6}	6.6×10^{-5}	6.9×10^{-5}	2.9×10^{-7}	1.7×10^{-4}
PES	4.4×10^{-6}	1.8×10^{-7}	2.0×10^{-5}	1.5×10^{-5}	3.0×10^{-8}	4.0×10^{-5}	2.9×10^{-5}	1.2×10^{-6}	6.9×10^{-5}	6.4×10^{-5}	2.0×10^{-7}	1.6×10^{-4}

The meaning acronyms: RI is the carcinogenic risk; TRI is the total carcinogenic risk; Dam 1 and 2 refers to dam áreas; *Pi* Pinheiros; *AF* Alberto Flores; *MC* Mário Campos; *B* Betim; *SJB* São Joaquim de Bicas; *F* Florestal; *SJV* São José da Varginha; *Pa* Paraopeba; *P* Pompeu; *PESRM* Parque Estadual da Serra do Rola-Moça; *PES* Parque Estadual do Sumidouro

Table 4 Spearman correlation matrix related to environmental and human variables: metal concentrations (Al, As, Cd, Cr, Cu, Fe, Hg, Mn, Ni, Pb and Zn), granulometry (silt, sand and clay), bulk density (density), pH, and PER after dam collapse of Córrego do Feijão Mine in Brumadinho municipality (Minas Gerais State, Brazil), ($p < 0.05$)

	Al	As	Cd	Cr	Cu	Fe	Hg	Mn	Ni
As	0.782								
p-Value	0.002								
Cd	0.727	0.868							
p-Value	0.005	0.000							
Cr	0.738	0.929	0.845						
p-Value	0.004	0.000	0.000						
Cu	0.705	0.945	0.819	0.973					
p-Value	0.007	0.000	0.001	0.000					
Fe	0.647	0.896	0.780	0.962	0.984				
p-Value	0.017	0.000	0.002	0.000	0.000				
Hg	0.624	0.852	0.792	0.836	0.858	0.877			
p-Value	0.023	0.000	0.001	0.000	0.000	0.000			
Mn	0.430	0.677	0.578	0.699	0.751	0.757	0.772		
p-Value	0.142	0.011	0.039	0.008	0.003	0.003	0.002		
Ni	0.749	0.934	0.912	0.956	0.956	0.940	0.825	0.721	
p-Value	0.003	0.000	0.000	0.000	0.000	0.000	0.001	0.005	
*P*b	0.741	0.907	0.951	0.890	0.863	0.841	0.781	0.567	0.945
p-Value	0.004	0.000	0.000	0.000	0.000	0.000	0.002	0.043	0.000
Sr	0.579	0.824	0.720	0.885	0.912	0.896	0.692	0.748	0.890
p-Value	0.038	0.001	0.006	0.000	0.000	0.000	0.009	0.003	0.000
Ti	0.678	0.839	0.946	0.833	0.830	0.784	0.740	0.676	0.900
p-Value	0.011	0.000	0.000	0.000	0.000	0.002	0.004	0.011	0.000
Zn	0.672	0.846	0.890	0.879	0.852	0.797	0.637	0.492	0.896
p-Value	0.012	0.000	0.000	0.000	0.000	0.001	0.019	0.087	0.000
Sand	0.619	0.828	0.919	0.795	0.825	0.814	0.797	0.704	0.922
p-Value	0.024	0.000	0.000	0.001	0.001	0.001	0.001	0.007	0.000
Silt	−0.296	−0.298	−0.528	−0.278	−0.160	−0.174	−0.258	−0.086	−0.391
p-Value	0.326	0.323	0.064	0.357	0.601	0.569	0.394	0.780	0.187
Clay	−0.503	−0.792	−0.768	−0.798	−0.878	−0.883	−0.852	−0.815	−0.867
p-Value	0.079	0.001	0.002	0.001	0.000	0.000	0.000	0.001	0.000
Dens	0.766	0.874	0.951	0.890	0.841	0.813	0.822	0.641	0.918
p-Value	0.002	0.000	0.000	0.000	0.000	0.001	0.001	0.018	0.000
*p*H	−0.557	−0.715	−0.729	−0.823	−0.787	−0.792	−0.727	−0.570	−0.757
p-Value	0.048	0.006	0.005	0.001	0.001	0.001	0.005	0.042	0.003
HI Ad	0.780	0.940	0.852	0.978	0.951	0.929	0.839	0.735	0.956
p-Value	0.002	0.000	0.000	0.000	0.000	0.000	0.000	0.004	0.000
HI ch	0.799	0.956	0.874	0.973	0.956	0.934	0.872	0.748	0.962
p-Value	0.001	0.000	0.000	0.000	0.000	0.000	0.000	0.003	0.000
TRI Ad	0.452	0.791	0.742	0.720	0.824	0.802	0.806	0.732	0.786
p-Value	0.121	0.001	0.004	0.006	0.001	0.001	0.001	0.004	0.001
TRI Ch	0.771	0.956	0.857	0.984	0.967	0.940	0.830	0.715	0.967
p-Value	0.002	0.000	0.000	0.000	0.000	0.000	0.000	0.006	0.000
*P*ER	0.669	0.883	0.883	0.900	0.883	0.894	0.939	0.598	0.878
p-Value	0.012	0.000	0.000	0.000	0.000	0.000	0.000	0.031	0.000

Table 4 (continued)

	Al	As	Cd	Cr	Cu	Fe	Hg	Mn	Ni			
*P*LI	0.757	0.960	0.856	0.982	0.977	0.966	0.895	0.700	0.955			
p-Value	0.003	0.000	0.000	0.000	0.000	0.000	0.000	0.008	0.000			
	Pb	Zn	Sand	Silt	Clay	Density	pH	HI Ad	HI ch	TRI Ad	TRI ch	PER
As												
p-Value												
Cd												
p-Value												
Cr												
p-Value												
Cu												
p-Value												
Fe												
p-Value												
Hg												
p-Value												
Mn												
p-Value												
Ni												
p-Value												
*P*b												
p-Value												
Sr	0.802											
p-Value	0.001											
Ti	0.914											
p-Value	0.000											
Zn	0.885											
p-Value	0.000											
Sand	0.916	0.795										
p-Value	0.000	0.001										
Silt	−0.551	−0.295	−0.516									
p-Value	0.051	0.328	0.071									
Clay	−0.724	−0.699	−0.869	0.155								
p-Value	0.005	0.008	0.000	0.614								
Dens	0.912	0.901	0.864	−0.466	−0.759							
p-Value	0.000	0.000	0.000	0.108	0.003							
*p*H	−0.737	−0.784	−0.609	0.028	0.625	−0.776						
p-Value	0.004	0.002	0.027	0.927	0.022	0.002						
HI Ad	0.896	0.879	0.809	−0.298	−0.781	0.923	−0.825					
p-Value	0.000	0.000	0.001	0.323	0.002	0.000	0.001					
HI ch	0.912	0.868	0.834	−0.298	−0.801	0.929	−0.831	0.995				
p-Value	0.000	0.000	0.000	0.323	0.001	0.000	0.000	0.000				
TRI Ad	0.736	0.626	0.834	−0.129	−0.878	0.632	−0.611	0.698	0.736			
p-Value	0.004	0.022	0.000	0.674	0.000	0.021	0.027	0.008	0.004			
TRI Ch	0.907	0.890	0.823	−0.298	−0.798	0.907	−0.803	0.995	0.989	0.736		
p-Value	0.000	0.000	0.001	0.323	0.001	0.000	0.001	0.000	0.000	0.004		

Table 4 (continued)

	Pb	Zn	Sand	Silt	Clay	Density	pH	HI Ad	HI ch	TRI Ad	TRI ch	PER
*P*ER	0.878	0.779	0.818	−0.378	−0.807	0.883	−0.763	0.867	0.889	0.757	0.872	
p-Value	0.000	0.002	0.001	0.202	0.001	0.000	0.002	0.000	0.000	0.003	0.000	
*P*LI	0.905	0.861	0.824	−0.272	−0.821	0.900	−0.813	0.977	0.982	0.762	0.982	0.934
p-Value	0.000	0.000	0.001	0.370	0.001	0.000	0.001	0.000	0.000	0.002	0.000	0.000

(especially Cr, Ni and As), therefore being considered as unacceptable for human health (Fig. 5).

The Spearman correlation analysis showed that all metal(loid)s concentrations (Al, As, Cd, Cr, Cu, Fe, Hg, Mn, Ni, Pb and Zn) are positively and strongly correlated to each other, except Al and Mn, where the correlation was not significant (p value = 0.14), Table 4. All metal(loid)s showed a strong positive correlation with sand contents and a strong negative correlation with clay contents, except Al, which showed a moderate positive correlation with sand contents and a moderate negative correlation with clay contents. The soil bulk density showed a strong positive correlation with all metals and the sand content, and a strong negative correlation with the clay content. The pH showed a strong negative correlation with most metals and with sand content. On the other hand, bulk density had a strong positive correlation with trace elements and a negative correlation with clay and pH. Except Al and Mn, all trace elements showed a strong correlation with the HI, TRI, PER and PLI indices. The indices also showed a strong positive correlation with each other and with sand, bulk density and a strong negative correlation with clay and pH.

The PCA biplot two-dimensional has provided a model of component analysis that clearly separates into clusters (Fig. 6). Correlating the variables from environmental and human data with metal(loid)s and 13 study areas, this indicated that about 90% of the data variance can be explained by the main components representing 84% and 6% the secondary components. The reference areas were associated with soil pH, particle size of clay and silt. The areas closest to collapsed dam areas, like Dam 1, Dam 2, Pi, AF, MC and B, were more correlated to the metal concentrations, bulk density and human indexes. Areas that were partially affected by the CFM dam failure such as SJB, F, SJV, Pa and P did not show such an evident correlation with the variables.

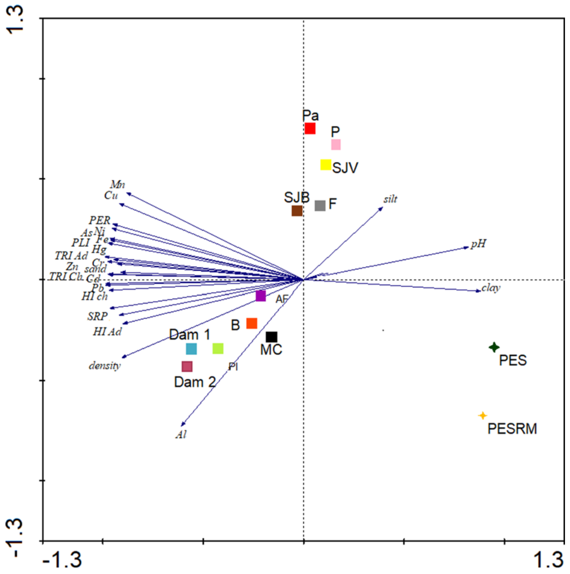

Fig. 6 Bi-dimensional ordination of principal component analyses. Variables such as metal concentrations (Al, As, Cd, Cr, Cu, Fe, Hg, Mn, Ni, Pb and Zn), granulometry (silt, sand, clay), bulk density, pH, PER, PLI, Hazard Index Adult (HI Ad), Hazard Index Children (HI ch), total carcinogenic risk Adult (TRI Ad) and total carcinogenic risk children (TRI ch) were used to assess the potential risk of mine tailings in soils after dam collapse of the Córrego do Feijão Mine, in Brumadinho Municipality (Minas Gerais State, Brazil)

Through cluster correlation analysis, it was possible to verify that the reference areas are significantly different from the areas that were totally and partially affected by the dam failure (Fig. 7). Areas close to the dam failure areas such as Dam 1, Dam 2, Pi, AF, MC and B were significantly similar to each other and were distinct from all the other areas. The partially affected areas (SJB, F, SJV, Pa and P) were similar to each other.

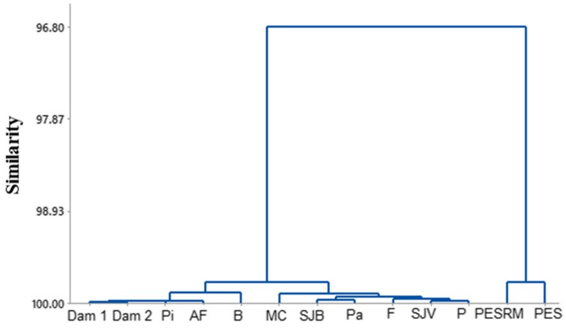

Fig. 7 Dendrograms obtained by cluster analysis, discriminating by the similarity among the 13 study areas after collapse of the Córrego do Feijão Mine, in Brumadinho Municipality (Minas Gerais State, Brazil)

Discussion

The number of catastrophic mine tailing dam failures is increasing globally. This reinforces the urgency in environmental protection actions given the complex interaction between a mine, its local operations and surrounding communities. The absence of public, timely, multi-scale information about the multiple dimensions of this interaction appears to be a common feature in the management of tailings dams by mining companies (Owen et al., 2020, Islam & Murakami, 2021). As with natural tragedies, industrial tragedies involve serious disruption to the functioning of a community or a society due to hazardous events interacting with conditions of exposure, vulnerability and capacity, leading to human, material, economic and/or environmental losses and impacts (UNDRR, 2017; Owen et al., 2020). Currently, there are still few studies about the impact of mine tailings on terrestrial ecosystems (Buch et al., 2020; Davila et al., 2020). A minority of research is tangible to the present study, especially in reference to soils impacted by the CFM dam collapse (Siqueira et al., 2022; Buch et al., 2023). Based on geochemical data reported in this study, the soil properties indicate changes in them and their quality after tailing deposition. Overall, the soil from the areas is very acidic and presents a high soil bulk density (>1.75 g cm^{-3}). This implies a limitation to the root development and implicitly in poor soil conditions for biological development (fauna) hindering, e.g., the passage of air and water favored by rooting.

Mine tailings have a wide variety of mineralogical components rich in trace metals and these can suffer geochemical evolution controlled not only by the original ore paragenesis, but also depending on the mineral processing techniques and weathering conditions (Lemos et al., 2021). Since the eighteenth century, mining activities have been carried out in the Minas Gerais State (Brazil), where the CFM dam is situated. The tailing dispersion over time in this region has attracted strong worldwide attention and aroused great concern for aquatic and terrestrial ecosystems due to metal-pollution and degradation (Buch et al., 2021; Davila et al., 2020). Furthermore, the recurrence in dam failures has caused intensive impacts on local economic-social reflecting at the national level (Buch et al., 2023). In the face still of such situations, the region has strong agricultural and livestock activities, causing concern in relation to the soil quality, which may indirectly affect human health (by the indirect consumption/ingestion) (Bonanomi et al., 2019). In this study, the As and Cd concentrations in samples from Bumadinho areas exceeded the threshold values established by CONAMA and COPAM-MG, which have been analogous to those found in water and sediments (CPRM, 2019b; Vergilio et al., 2020; Thompson et al., 2020), however contrasting to those reported by Siqueira et al., 2022, whose values were below the detection limits. In the areas of total and partial contamination, the same similarity observed in the soils for Cr, Cu, Hg, Ni and Pb values, which were above to those established for soil quality, has also been evidenced in water bodies affected by CFM mud after their collapse (CPRM, 2019b; Teramoto et al., 2021; Pacheco et al., 2022).

The enrichment factor, suggest that in the 11 impacted areas (in partial or total levels), prevails a strong anthropogenic interference due to the CFM dam rupture, despite regional geogenic contributions. In these study areas, the enrichment of Cd, As, Hg, Cu, Pb and Ni in riparian soils was from 2 to 40 times greater than the levels found in years prior to failure. Similarly, anomalous contents of Fe and Mn have also been reported in the literature in sediment and water samples, as a consequence of the CFM dam collapse (Thompson et al., 2020; Pacheco et al., 2022). In almost ubiquitous behavior in riparian soils located around dam failure sites, as well as in floodplain soils and other soil types close to rivers and other river corridors, they tend to mobilize potentially toxic elements, indicating outlier than natural values (Li et al., 2014). Chemically, this can be explained by the higher affinity of the most metal(loid)s with Fe oxyhydroxides and organic material, a component which is very high in these soils, resulting in greater adsorption and cohesion (Grybos et al., 2007; Kabata-Pendias, 2011). The failure of the Pb–Zn mine dam in Chenzhou (China on 25 August 1985) caused by heavy rains inundating the Dong River valley resulted in soils anomalous concentrations of metal(loid)s (Liu et al., 2005). As happened in the region after collapse of CFM dam, a large volume of mine tailings from Chenzhou dam was deposited in soils, strips of farmland 400 m wide along both river banks were covered with a 15-cm-thick layer of mining sludge. Even decades after failure of Chenzhou dam, the unremediated soil showed element concentrations far in excess of the Chinese soil maximum allowable concentration standard (Liu et al., 2005; Lyu et al., 2019). The dispersion of pollutants may be higher and increase its toxicity over time, when it is influenced by highly mobile and variable environmental components such as rainfall. In this context, areas that were partially affected by the collapse of CFM dam, which did not receive direct deposition of tailings, were contaminated due to overflow of the Paraopeba river under intense periods of rain, indicating also the highest metal(loid)s contents in riparian soils when compared with the preexisting concentrations (Mendes et al., 2023). The difficulty measuring the environmental risks after CFM tailings deposition in the Paraopeba River channel due to contamination substantial fluctuates in function of season has been related in the literature (Teramoto et al., 2021; Mendes et al., 2023). It has also been supposed that the total and dissolved concentrations of metals increase during the rainy season by resuspension, associated with larger stream flows that tend to remobilize sediments and tailings from the stream bed, raising the surface area available for interactions, especially chemical and biological (Lebron et al., 2020; Teramoto et al., 2021).

Pollution load and potential ecological risk indices also expressed an analogous tendency to pollution by mine tailings in soils, confirming for the studied areas, the current status of high pollution and of very strong risks to biological communities (local fauna and flora). Generally, sites near to collapsed dam areas are the most polluted both in soils and in water bodies, due to the greater intensity of the spilling tailing waves and to the more concentrated volume of pollutants received (Custodio et al., 2020; Khosravi et al., 2019; Ordóñez et al., 2011). Although it seems obvious that areas closest to sources of pollution tend to be the most affected, over time the opposite can also be expected by particular reasons inherent to local characteristics, e.g., topography, rainfall, moisture, biological activity, anthropic interferences, management, land use and various natural attenuating factors (Buch et al., 2021). Exogenous and intrinsic factors directly influence to chemical species changes in soils the mobility, resulting in distinct effects on mobility, bioavailability and biomagnification of Cd, Pb, Hg, As, Cr and Ni (USEPA, 2001; Kabata-Pendias, 2011; Luo et al., 2012; Buch et al., 2021). Metal ions in soil solution generally are more available for a variety of processes, including plant uptake and transport; however, metal ions in the solid phase may become available if environmental conditions change (Balali-Mood et al., 2021). Mining regions naturally presents metal(loid)s concentrations from the parental lithology and mining activities (e.g., along decades or centuries like this study demonstrates) (Parviainen et al., 2022; Rana et al., 2021). In soils, dissolution kinetics may be one of the main factors controlling the availability of mineral-derived metal ions (Chaturvedi et al., 2007; Rashed, 2010). Generally, the elemental and sulfide forms of a metal are less soluble in biological fluids (e.g., in mammalian species: gastrointestinal tract fluid, sweat, or fluid in the alveoli of the lungs) and hence less bioavailable than the oxide, hydroxide, carbonate, and sulfate forms of the same metal. However, notable exceptions to this rule of thumb exist, such as the elevated pulmonary and dermal bioavailability of elemental mercury; the low solubility of nickel oxides (in the range of nickel sulfide); and the low solubility of chromium hydroxide, the most prevalent form of chromium in soils (Kabata-Pendias, 2011).

Regarding the human health risk assessment found in this research, the values of HQ indices for the metal(loid)s exceed limits values considered safe for the development of non-carcinogenic diseases, especially for the children's group. For this group, the integration and individual interpretation of the three exposure pathways (HIs) indicated the children as target organism more sensitive to effects of mine tailings in soils. Studies evaluating floodplain soils contaminated with iron ore tailings also indicated for metals, HQ's values greater than 1 for children (Davila et al., 2020; Buch et al., 2022). In communities from mine regions, the greater toxicity propensity of children has been observed from 7 to 25 times, when compared to adults groups Qing et al, 2015; Cocãrţã et al., 2016; Adimalla, 2020; Kumar et al., 2022a, 2022b). The possible associated reasons are related to a higher rate of child absorption and sensitivity (e.g., hemoglobin affinity to heavy metals) (Adimalla, 2020; Kumar et al., 2022a, 2022b); greater vulnerability to the ingestion of hazardous materials due to their behavior, since the brain and nervous system are still being formed and do not have sufficient capacity to identify and/or distinguish materials that provide a danger alert (Kumar et al., 2022a, 2022b); and physiological conditions, which tend to be more frail than adults, as well as higher respiration rates per unit of body weight and increased gastrointestinal absorption of some substances (enHealth, 2012; Isley et al., 2022). Regardless of the mining region, anywhere in the world, the people's susceptibility to non-carcinogenic adverse effects is the same, like hyperpigmentation, keratosis and vascular complications (Gore et al., 2022; Kumar et al., 2022a, 2022b).

Under a geochemical point of view, the variability in the hazard index may be associated with inherent factors (such as geogenic and anthropogenic) to the mine types (Li et al., 2014; Liu et al., 2005). In the case of mines from Pb–Zn, Mn and W (tungsten), the HI values greatly exceed 1, whereas from Cu, Au (gold) and Fe the HI values present values up to 1 (Li et al., 2014). Such data underpin our findings about low HI values, once the mining activity of the region is iron ore extraction.

Here, it is necessary to emphasize that HQ and RI indices are separately calculated for each metal. Nevertheless, they are components of a mixture like the mine tailings (whose composition is of extreme variation in terms of quantity and chemical element diversity). Thus, the safety level to health should consider not only the isolated toxicity of a particular element, but also the joint and integrated effect between the components of a mixture (e.g., by aditism and synergism processes) (Zhang et al., 2012; Zhu et al., 2007).

In respect to the carcinogenic risks for adults and children groups, all areas affected by CFM tailings (except reference areas) indicated danger to health due to presence of As, Cd, Cr, Ni and Pb in soils. This propensity is consistent with geochemical studies in mine type different, which evidenced As high contribution to the carcinogenic index, showing an As contribution average of 5.8×10^{-4} $(7.6\times10-4)$ in mine antimony, 1.3×10^{-5} (6.9×10^{-6}) in coal, 4.7×10^{-5} (7.1×10^{-5}) in copper, $1.1\times10-5$ (7.7×10^{-6}) in gold, and 1.7×10^{-4} (2.0×10^{-4}) and in lead–zinc mining areas (Li et al., 2014). Recent studies in three villages close to the Rio Tinto district (Southwest Spain) situated close to a huge Cu mining company (one of the largest in the world) investigated the impact on health of this type of mining. The area faces a complex issue of atmospheric pollution resulting from mining processes, leading to dust containing predominantly As/Pb, alongside noteworthy Cu and Zn concentrations and a significant contribution of the mine to particulate matter pollution together with other external sources. Carcinogenic assessments of metals by inhalation indicated maximum permissible levels of exposure to particulate matter into the atmosphere (Boente et al., 2023). In human and animal organisms, the input of metal(loid)s by direct ingestion has been identified as the main exposure route in soils contaminated by mine tailings (Adimalla, 2020; Buch et al., 2023). However, these elements may be input in organisms via indirect by enriched crop ingestion, which is strongly adsorbed by soil solution, soil particles as well as plant roots and on the surface of leafy vegetables (Buch et al., 2023; Deng et al., 2019; Kabata-Pendias, 2011).

The soil and water contamination by mining waste from more than 160,000 mines abandoned in the Western USA (Arizona, California, Colorado, Idaho, Montana, Nevada, New Mexico, Oregon, South Dakota, Utah, Washington, and Wyoming) has generated along time chronic damage discrepant by metal(loid)s in Native American communities (Lewis et al., 2017). In abandoned areas with gold mine tailings from South Africa, the carcinogenic risks derived the contaminated soils have been considered very high both for children of 3×10^{-2} (As) and 4×10^{-2} (Ni) and for adults of 5×10^{-3} (As) and 4×10^{-3} (Ni) (Ngole-Jeme & Fantke, 2017). In this African region, the aggravating to the high potential toxicity of metals is associated with the nutritional debility and to HIV virus prevalence, contributors to the impairment immune systems (Emmanuel et al., 2018; Ngole-Jeme & Fantke, 2017).

Interactions between metal(loid)s and organisms extra and intracellularly induce adverse effects (Ding et al., 2022). Chemical groups on the cell surface can bind extracellular metal(loid) s, which interfere with the cellular uptake of nutrients, in addition to causing structural damage to cells (Kumar et al., 2022a, 2022b). Assessments in populations living in the Panasqueira mine area of central Portugal found a higher internal dose of elements such as As, Cr, Pb, Mn, Mo, and Zn in exposed individuals (Coelho et al., 2014). Furthermore, metal(loid)-contamination in the Panasqueira mine area induced genotoxic damage in individuals working in the mine or living beings in the area (Coelho et al., 2014). Long-term exposure to metal(loid)s may have toxic effects on various organ systems and cause various clinical symptoms. The target of these elements, such as As and Cd, is bone tissue which results in toxic and chronic effects (Lewis et al., 2017). Various studies from the literature has demonstrated the level of metals in the blood of people from communities that inhabit regions where mining activities predominate (Gil et al., 2006; Lewis et al., 2017; Adimalha and Wang, 2018; Paulelli et al., 2022). Such evidence has been especially noted in the population of Brumadinho-MG, Brazil (in the same region studied of this paper) in blood and urine samples, showing high levels of As, Cd, Hg and Pb (Mota et al., 2022; Paulelli et al., 2022). The intense and continuous disorder caused by metal (loid)s in the biological organism has been widely related to innumerable acute and chronic diseases (Bienert & Tamas, 2018; Rehman et al., 2022). Severe oxidative damage may be induced by metal(loid) s when they enter cells through non-specific chemiosmotic metal uptake systems or certain transporters (Bienert and Tamás, 2018; Kumar et al., 2022a, 2022b). Intracellular metals can replace metal cofactors in active centers of enzymes, causing in denaturation and inactivation of these enzymes (Balistrieri et al., 2018). DNA damage may be induced by intracellular metals decreasing the DNA content and destroying the DNA structure (Ding et al., 2022; Tibane & Mamba, 2022).

There is no doubt that the economic benefit of mining activities is exorbitantly high. However, the risk to human and environmental health accompanies this proportion. Targeted remediation techniques must be implemented urgently and are needed to mitigate carcinogenic potential risks posed by mining tailings to human health and to ecological. Since such "silent," long-term and intensive impacts tend to unbalance populations of animal and plant species, extinguishing them and interfering with the food chain and the functionality of ecosystems (Buch et al., 2021; Mota et al., 2022; Ngole-Jeme & Fantke, 2017; Paulelli et al., 2022).

Conclusion

Indices of human health and geochemical data congruently reinforce evidence of high pollution levels, potential ecological risks and propensities to develop non-cancerous and cancerous diseases for the local community in all areas. Future contributions derived from these data will allow deeper analysis and subjective less of the complex effects of metal(loid)s in different biological species through (eco)toxicological studies (in terrestrial ecosystem dwellers). This information is in process of publication by the same authors of this paper. In addition, it will provide a basis for more specific investigations about certain exogenous and endogenous factors inherent to the synergistic and summative processes that occur in such regions.

In all areas affected by CFM tailings, metal values exceed those established by Brazilian guidelines for soil quality for at least two elements. These values also exceed the maximum values allowed in other countries to maintain soil quality without causing damage to human and ecological health. The findings confirm a substantial anthropogenic contribution to metal enrichment in riparian soils as a consequence of the CFM dam collapse. Spearman correlation matrixes and PCA analysis emphasize serious contamination, especially associated with As, Cd, Cr, Hg, Ni and Pb, suggesting that the areas most compromised by the negative soil quality are those closest to the dam break areas (Dam1, Dam 2, Pi, AF, MC and B). However, this does not disregard or attenuate the observed pollution impacts in the other areas of the study.

It is interesting to point out that even after five years since the CFM dam failure, intensive changes caused in the soil properties still remain unchanged, and natural attenuation has not been evidenced in the investigated areas. Furthermore, so far no interventionist recovery measures in these areas have been employed and implemented.

The propensity for the development of diseases as a consequence of prolonged exposure to environmental matrices contaminated with metals poses risks to human health. Effective policies for the management of mine tailings must be required by the local population to the competent environmental bodies. Thses together with mine companies must be held responsible for the inherent risks, ensuring collective health and environmental preservation/restitution.

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Declarations

Competing interests The authors declare no competing interests.

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